NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL AND CONFERENCE OPINION

Title:	Biological and Conference Opinion on (1) U.S. Navy Northwest Training and Testing Activities (NWTT); and (2) the National Marine Fisheries Service's promulgation of regulations and issuance of a letter of authorization pursuant to the Marine Mammal Protection Act for the U.S. Navy to "take" marine mammals incidental to NWTT activities from November 2020 through November 2027
Consultation Conducted By:	Endangered Species Act (ESA) Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Action Agencies:	United States Navy (Navy) and NOAA's National Marine Fisheries Service, Office of Protected Resources, Permits and Conservation Division
Publisher:	Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Approved:	WIETING.DONNA.S.1365710607 Digitally signed by WIETING.DONNA.S.1365710607 Date: 2020.10.19 17:07:03 -04'00' Donna S. Wieting Director, Office of Protected Resources
Date:	October 19, 2020
Consultation Tracking number:	OPR-2019-00786
Digital Object Identifier (DOI):	https://doi.org/10.25923/7034-6a18

This page left blank intentionally

TABLE OF CONTENTS

Page

1 Introduction 28
1.1 Background 29
1.2 Consultation History 31

2 The Assessment Framework 37
2.1 Evidence Available for this Consultation 39
2.2 Acoustic Effects Analysis for Marine Mammals and Sea Turtles 40
2.2.1 Navy Acoustic Effects Model (NAEMO) 41
2.2.2 Criteria and Thresholds to Predict Impacts to Marine Mammals 44
2.2.3 Criteria and Thresholds to Predict Impacts to Sea Turtles 54
2.3 Criteria and Thresholds to Predict Impacts to Fishes 60
2.3.1 Sonar – Fishes 61
2.3.2 Explosives – Fishes 62
2.4 Species Density Estimates 63
2.4.1 Marine Mammal and Sea Turtle Density Estimates 63
2.4.2 Fish Density Estimates 73

3 Description of the Proposed Action 98
3.1 Primary Mission Areas 99
3.1.1 Air Warfare 100
3.1.2 Anti-Submarine Warfare 100
3.1.3 Electronic Warfare 100
3.1.4 Mine Warfare 101
3.1.5 Surface Warfare 101
3.2 Classification of Sonar and Explosive Sources into Bins 102
3.3 Proposed Training and Testing Activities 107
3.3.1 Training Activities 108
3.3.2 Testing Activities 118
Activity Name 122
Activity Description 122
3.4 Standard Operating Procedures 136
3.4.1 Vessel Safety 136
3.4.2 Weapons Firing Safety 136
3.4.3 Target Deployment and Retrieval Safety 137
3.4.4 Underwater Detonation Safety 137
3.4.5 Towed In-Water Device Safety 137
3.5 Mitigation Measures 138
3.5.1 Procedural Mitigation 138

3.5.2 Mitigation Areas 165
3.6 MMPA Regulations and Issuance of a Letter of Authorization 179

4 Action Area 181
4.1 Offshore Portion of the Action Area 183
4.1.1 Special Use Airspace 183
4.1.2 Sea and Undersea Space 185
4.2 Inland Waters of the Action Area 186
4.2.1 Air Space 186
4.2.2 Sea and Undersea Space 186
4.3 Southeast Alaska Acoustic Measurement Facility 190

5 Potential Stressors 192
5.1 Acoustic Stressors 192
5.1.1 Sonar and other Transducers 192
5.1.2 Explosives 194
5.1.3 Vessel Noise 196
5.1.4 Aircraft Noise 197
5.1.5 Weapons Firing, Launch and Impact Noise 202
5.2 Energy Stressors 204
5.2.1 Electromagnetic Devices 204
5.2.2 Lasers 205
5.3 Physical Disturbance and Strike Stressors 206
5.3.1 Vessel Strike 206
5.3.2 Military Expended Materials 208
5.3.3 In-Water Devices 212
5.3.4 Seafloor Devices 213
5.4 Entanglement Stressors 214
5.4.1 Wires and Cables 215
5.4.2 Decelerators and Parachutes 216
5.4.3 Biodegradable Polymer 218
5.5 Ingestion Stressors 218
5.5.1 Non-Explosive Practice Munitions 218
5.5.2 Fragments from High Explosive Munitions 219
5.5.3 Target Related Materials 219
5.5.4 Chaff 219
5.5.5 Flares 220
5.6 Secondary Stressors 220

6 Species and Designated Critical Habitat that May be Affected 222
6.1 Species Not Likely to be Adversely Affected 225
6.1.1 North Pacific Right Whale 226

6.1.2 Western North Pacific Gray Whale 229
6.1.3 Steller Sea Lion Western Distinct Population Segment (DPS) 233
6.1.4 Ozette Lake Sockeye Salmon 237
6.1.5 Steelhead – Southern California DPS 240
6.2 Status of Species Likely to be Adversely Affected 241
6.2.1 Blue Whale 242
6.2.2 Fin Whale 253
6.2.3 Humpback Whale – Central America and Mexico DPSs 261
6.2.4 Killer Whale – Southern Resident DPS 273
6.2.5 Sei Whale 294
6.2.6 Sperm Whale 301
6.2.7 Guadalupe Fur Seal 309
6.2.8 Leatherback Sea Turtle 315
6.2.9 Chinook Salmon – California Coastal Evolutionarily Significant Unit (ESU) 325
6.2.10 Chinook Salmon – Central Valley Spring-Run ESU 329
6.2.11 Chinook Salmon – Lower Columbia River ESU 333
6.2.12 Chinook Salmon – Puget Sound ESU 337
6.2.13 Chinook Salmon – Sacramento River Winter-Run ESU 344
6.2.14 Chinook Salmon – Snake River Fall-Run ESU 348
6.2.15 Chinook Salmon – Snake River Spring/Summer-Run ESU 352
6.2.16 Chinook Salmon – Upper Columbia River Spring-Run ESU 356
6.2.17 Chinook Salmon – Upper Willamette River ESU 359
6.2.18 Chum Salmon – Columbia River ESU 363
6.2.19 Chum Salmon – Hood Canal Summer-Run ESU 367
6.2.20 Coho Salmon – Central California Coast ESU 372
6.2.21 Coho Salmon – Lower Columbia River ESU 377
6.2.22 Coho Salmon – Oregon Coast ESU 381
6.2.23 Coho Salmon – Southern Oregon and Northern California Coasts ESU 386
6.2.24 Eulachon – Southern DPS 389
6.2.25 Green Sturgeon – Southern DPS 394
6.2.26 Sockeye Salmon – Snake River ESU 403
6.2.27 Steelhead – California Central Valley DPS 406
6.2.28 Steelhead – Central California Coast DPS 410
6.2.29 Steelhead – Lower Columbia River DPS 413
6.2.30 Steelhead – Middle Columbia River DPS 417
6.2.31 Steelhead – Northern California DPS 421
6.2.32 Steelhead – Puget Sound DPS 424
6.2.33 Steelhead – Snake River Basin DPS 429
6.2.34 Steelhead South-Central California DPS 432

6.2.35 Steelhead – Upper Columbia River DPS 434
6.2.36 Steelhead – Upper Willamette River DPS 437
6.2.37 Bocaccio – Puget Sound/Georgia Basin DPS 441
6.2.38 Yelloweye Rockfish – Puget Sound/Georgia Basin DPS 446

7 Environmental Baseline 450
7.1 Global Climate Change 450
7.2 Oceanic Temperature Regimes 455
7.3 Whaling 457
7.4 Unusual Mortality Events 458
7.5 Fisheries Bycatch and Gear Interactions 460
7.5.1 Fisheries Bycatch of Endangered and Threatened Fishes 460
7.5.2 Fisheries Bycatch and Entanglement of Leatherback Sea Turtles 464
7.5.3 Entanglement of Marine Mammals in Fishing Gear 465
7.6 Aquaculture 466
7.7 Hatcheries 467
7.8 Vessel Strike 467
7.9 Water Quality Degradation 470
7.10 Oil Spills 475
7.11 Marine Debris 478
7.12 Anthropogenic Sound 480
7.12.1 Seismic Surveys 481
7.12.2 Active Sonar 483
7.12.3 Vessel Sound and Commercial Shipping 483
7.12.4 Aircraft Noise 488
7.13 Commercial and Private Marine Mammal Watching 489
7.14 Ongoing Military Training and Testing Activities 491
7.14.1 Summary of Northwest Training and Testing Phase II Biological Opinion 492
7.15 Invasive Species 493
7.16 Parasites and/or Disease 493
7.17 Scientific Research and Permits 495
7.18 Impact of the Baseline on ESA-listed Resources 499

8 Effects of the Action 501
8.1 Stressors Not Likely to Adversely Affect ESA-listed Species 512
8.1.1 Marine Mammals 512
8.1.2 Leatherback Sea Turtles 541
8.1.3 Fishes 558
8.2 Stressors Likely to Adversely Affect ESA-listed Species 577
8.2.1 Marine Mammals 577
8.2.2 Leatherback Sea Turtles 692
8.2.3 Fishes – Effects of Explosive Stressors 701

8.3 Designated and Proposed Critical Habitat Effects Analysis 754
8.3.1 Marine Mammals 754
8.3.2 Leatherback Sea Turtle Designated Critical Habitat 773
8.3.3 Fishes 775

9 Cumulative Effects 784

10 Integration and Synthesis 786
10.1 Blue Whale 786
10.2 Fin Whale 788
10.3 Humpback Whale – Central America and Mexico DPSs 790
10.3.1 Central America and Mexico DPSs Proposed Critical Habitat 792
10.4 Southern Resident Killer Whale DPS 793
10.4.1 Inland Designated Critical Habitat 800
10.4.2 Coastal Proposed Critical Habitat 804
10.5 Sei Whale 808
10.6 Sperm Whale 810
10.7 Guadalupe Fur Seal 812
10.8 Leatherback Sea Turtle 813
10.8.1 Designated Critical Habitat 815
10.9 Chinook Salmon 815
10.10 Chum Salmon 827
10.11 Coho Salmon 833
10.12 Steelhead 839
10.13 Sockeye Salmon 848
10.14 Eulachon 851
10.15 Bocaccio – Puget Sound / Georgia Basin DPS 853
10.15.1 Puget Sound / Georgia Basin DPS Designated Critical Habitat 855
10.16 Yelloweye Rockfish – Puget Sound / Georgia Basin DPS 856
10.16.1 Puget Sound / Georgia Basin DPS Designated Critical Habitat 857
10.17 Green Sturgeon – Southern DPS 858
10.17.1 Southern DPS Designated Critical Habitat 862

11 Conclusion 865

12 Incidental Take Statement 866
12.1 Amount or Extent of Take 867
12.2 Effects of the Take 872
12.3 Reasonable and Prudent Measures 873
12.4 Terms and Conditions 873

13 Conservation Recommendations 876

14 Reinitiation Notice 878

15 References 879

LIST OF TABLES

Page

Table 1. Acoustic thresholds identifying the onset of temporary threshold shift (TTS) and permanent threshold shift (PTS) for non-impulsive sound sources by functional hearing group (Navy 2017). 47

Table 2. Onset of TTS and PTS in marine mammals for explosives by functional hearing group. 49

Table 3. Criteria to quantitatively assess marine mammal mortality and non-auditory injury due to underwater explosions. 49

Table 4. Significant behavioral response cutoff distances by species group for moderate source level, single platform sonar events and for high source level[1], multi-platform sonar events (Navy 2017a). 54

Table 5. Behavioral disturbance thresholds for marine mammals from underwater explosives used for the quantitative analysis (Navy 2017a). 54

Table 6. Criteria to quantitatively assess sea turtle non-auditory injury due to underwater explosions. 59

Table 7. TTS and PTS peak pressure thresholds derived for sea turtles exposed to impulsive sounds 60

Table 8. Fish sound exposure criteria for TTS from sonar (Navy 2017). 62

Table 9. Sound exposure criteria used for mortality and injury in fishes. 62

Table 10. Summary of marine mammal and sea turtle density values used for quantitative acoustic effects analysis (Navy 2019f) 65

Table 11. Summary of estimated annual abundance of ESA-listed salmonids. Abundance estimates for each ESU and DPS are divided into natural, listed hatchery intact adipose, and listed hatchery adipose clip. 74

Table 12. Habitat area (distribution) used for each salmonid ESU and DPS in the offshore marine environment. 82

Table 13. Offshore density estimates for ESA-listed salmonids in the NWTT Study Area. 84

Table 14. Summary of estimated adult annual abundance of non-listed Chinook salmon, coho salmon, and chum salmon ESUs included in effects analysis for Southern Resident killer whales prey (Data source: M. Dennis, NMFS personal communication to R. Salz, NMFS, March 12, 2020). 86

Table 15. Adult habitat area (distribution) used in the offshore marine environment by each non-listed salmon ESU included in the effects analysis for Southern Resident killer whale prey. 88

Table 16. Offshore adult density estimates in the NWTT Study Area for non-listed salmon ESUs included in the effects analysis for Southern Resident killer whales prey (includes natural and hatchery origin fish). 89

Table 17. Calculation of Southern DPS green sturgeon acoustic tag detection rates by month and seasonal time frame (rates based on peak 5-day period within each month). 93

Table 18. Description of Navy sonar source bins and comparison of maximum annual activity levels by bin between ongoing (Phase II) activities and the proposed action (Phase III) (Navy 2020e). 103

Table 19. Sonar and transducer sounds sources qualitatively analyzed (Navy 2020e). 106

Table 20. Description of Navy explosive source bins and comparison of the maximum annual number of explosives by bin between ongoing (Phase II) activities and the proposed action (Phase III) (Navy 2020e). 107

Table 21. Navy proposed training activities for NWTT Phase III (Navy 2020e). 109

Table 22. Representative Naval Sea Systems Command testing activities. 119

Table 23. Representative Naval Air Systems Command testing activities. 122

Table 24. Naval Sea Systems Command proposed testing activities for NWTT Phase III (Navy 2020e). 123

Table 25. Naval Air Systems Command proposed testing activities for NWTT Phase III (Navy 2020e). 134

Table 26. Environmental awareness and education procedural mitigation 141

Table 27. Procedural mitigation for active sonar. 143

Table 28. Procedural mitigation for weapons firing noise. 144

Table 29. Procedural mitigation for explosive sonobuoys. 145

Table 30. Procedural mitigation for explosive torpedoes. 149

Table 31. Procedural mitigation for explosive medium-caliber and large-caliber projectiles. 151

Table 32. Procedural mitigation for explosive missiles. 153

Table 33. Procedural mitigation for explosive bombs. 155

Table 34. Procedural mitigation for explosive mine countermeasure and neutralization activities. 156

Table 35. Procedural mitigation for explosive mine neutralization activities involving Navy divers. 159

Table 36. Procedural mitigation for vessel movement. 160

Table 37. Procedural mitigation for towed in-water devices. 161

Table 38. Procedural mitigation for small-, medium-, and large-caliber non-explosive practice munitions. 162

Table 39. Procedural mitigation for non-explosive missiles. 163

Table 40. Procedural mitigation for non-explosive bombs and mine shapes. 164

Table 41. Mitigation areas for seafloor resources. 166

Table 42. Marine species mitigation areas in the offshore NWTT Study Area. 170

Table 43. Marine species mitigation areas in the NWTT Inland Waters. 172

Table 44. Summary of bathymetric features within the offshore portion of the action area. 185

Table 45. Summary of bathymetric features within the Inland Waters of the action area. 188

Table 46. Typical air[1] explosive munitions during Navy activities (Navy 2020e). 196

Table 47. Representative aircraft sound characteristics (Navy 2020e). 199

Table 48. Annual number and location of events including aircraft movement (Navy 2020e). 199

Table 49. Sonic boom underwater sound levels modeled for supersonic flight from a representative aircraft (Navy 2020e). 201

Table 50. Examples of some types of weapons noise from NWTT activities (Navy 2020e). 203

Table 51. Representative vessel types, lengths, and speeds (Navy 2020e). 207

Table 52. Annual number and location of events including vessel movement (Navy 2020e). 208

Table 53. Annual maximum number and location of expended non-explosive practice munitions (Navy 2020e). 209

Table 54. Annual maximum number and location of other expended or recovered items (Navy 2020e). 210

Table 55. Annual maximum number and location of explosive munitions that may result in fragments (Navy 2020e). 211

Table 56. Annual maximum number and location of expended and recovered[1] targets (Navy 2020e). 212

Table 57. Representative types, sizes, and speeds of Navy in-water devices (Navy 2020e). 213

Table 58. Annual number and location of events including in-water devices (Navy 2020e). 213

Table 59. Annual number and location of events including seafloor devices (Navy 2020e). 214

Table 60. Annual number and location of expended wires and cables (Navy 2020e). 215

Table 61. Size categories for decelerators/parachutes expended during training and testing activities (Navy 2020e). 217

Table 62. Annual number and location of expended decelerators/parachutes (Navy 2020e). 217

Table 63. Annual number and location of expended chaff and flares and associated materials (Navy 2020e). 220

Table 64. ESA-listed species and designated (or proposed) critical habitat that may be affected by the proposed action. 222

Table 65. Abundance Estimates for the Ozette Lake ESU of Sockeye salmon (NMFS 2019c). 238

Table 66. Southern Resident killer whale proposed coastal critical habitat specific area descriptions. 293

Table 67. Average abundance for California Coastal Chinook salmon natural-origin spawners (NMFS 2019d). 327

Table 68. Average abundance estimates for Central Valley Spring Run Chinook salmon natural- and hatchery-origin spawners from 2013-2017 (NMFS 2019d). 331

Table 69. Abundance Estimates for the Lower Columbia River ESU of Chinook salmon (NMFS 2020c). 335

Table 70. Average abundance estimates for Puget Sound Chinook salmon natural- and hatchery-origin spawners 2012-2016 (NMFS 2020c). 339

Table 71. Expected 2019 Puget Sound Chinook salmon hatchery releases (NMFS 2020c). 340

Table 72. Average abundance estimates for Sacramento winter-run Chinook salmon natural- and hatchery-origin spawners 2013-2017 (NMFS 2019d). 346

Table 73. Average Abundance Estimates for the Snake River Fall-Run ESU of Chinook salmon from 2015 to 2019 (NMFS 2020c). 350

Table 74. Average Abundance Estimates for the Snake River Spring/Summer-Run ESU of Chinook salmon for 2014-2018 (NMFS 2020c). 354

Table 75. Five Year Average (2015 to 2020) Abundance Estimates for the Upper Columbia River Spring-Run ESU of Chinook salmon (NMFS 2020c). 358

Table 76. Average Abundance Estimates for the Upper Willamette River Spring-Run ESU of Chinook salmon from 2014 to 2018 for Adults and 2015 to 2020 for Juveniles (NMFS 2020c). 362

Table 77. Abundance Estimates for the Columbia River ESU of Chum salmon (NMFS 2019c; Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020). 365

Table 78. Hood Canal summer-run juvenile chum salmon hatchery releases (NMFS 2020c). 369

Table 79. Abundance of natural-origin and hatchery-origin HCS chum salmon spawners in escapements 2013-2017 (NMFS 2020c). 369

Table 80. Average juvenile Central California Coast coho salmon coho salmon hatchery releases (NMFS 2019d). 374

Table 81. Geometric mean abundances of Central California Coast coho salmon spawner escapements by population. Populations in bold font are independent populations (NMFS 2019d). 375

Table 82. Juvenile Abundance Estimates for the Lower Columbia River ESU of Coho salmon (NMFS 2020c). 379

Table 83. Average abundance estimates for Lower Columbia River Chinook salmon natural- and hatchery-origin spawners (NMFS 2020c). 379

Table 84. Average abundance estimates for the Oregon Coast ESU coho salmon natural- and hatchery-origin spawners (NMFS 2020c). 384

Table 85. Average abundance estimates of the natural-origin and hatchery-produced adult Southern Oregon/Northern California Coast ESU coho salmon returning to the Rogue, Trinity, and Klamath rivers (NMFS 2020c). 388

Table 86. Southern DPS eulachon spawning estimates for the lower Fraser River (British Columbia, Canada) and Columbia River (Oregon/Washington states, USA) (NMFS 2020c). 392

Table 87. Physical and biological features of marine and estuarine critical habitat designated for Southern DPS green sturgeon and corresponding species life history events. 400

Table 88. Current Abundance Estimates for Snake River ESU Sockeye salmon (NMFS 2020c). 404

Table 89. Current Abundance Estimates for the California Central Valley ESU of Steelhead (NMFS 2020c). 408

Table 90. Current Abundance Estimates for the California Central Coast ESU of Steelhead (NMFS 2020c). 412

Table 91. Current Abundance Estimates for the Lower Columbia River DPS of Steelhead (NMFS 2019c; Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020). 415

Table 92. Current Abundance Estimates for the Middle Columbia River DPS of Steelhead (NMFS 2020c). 420

Table 93. Current Abundance Estimates for the Northern California DPS of Steelhead (NMFS 2019d). 422

Table 94. Expected 2019 Puget Sound steelhead listed hatchery releases (NMFS 2020c). 426

Table 95. Abundance of PS steelhead spawner escapements (natural-origin and hatchery-production combined) from 2012-2016 (NMFS 2020c). 427

Table 96. Current Abundance Estimates for the Snake River Basin DPS of Steelhead (NMFS 2020c). 431

Table 97. Current Abundance Estimates for the South-Central California Coast DPS of Steelhead (NMFS 2019d). 433

Table 98. Current Abundance Estimates for the Upper Columbia River DPS of Steelhead (NMFS 2020c). 436

Table 99. Current Abundance Estimates for the Upper Willamette River DPS of Steelhead (NMFS 2019c). 439

Table 100. Total expected take of west coast ESA-listed fish species for scientific research and monitoring authorized for 2019. 497

Table 101. NMFS ESA effects determinations by stressor and overall effects determination for each ESA-listed species (LAA = likely to adversely affect; NLAA = not likely to adversely affect; NE = no effect). 503

Table 102. Range to permanent threshold shift for five representative sonar systems. 603

Table 103. Ranges to temporary threshold shift for sonar bin HF4 over a representative range of environments within the action area. 603

Table 104. Ranges to temporary threshold shift for sonar bin LF4 over a representative range of environments within the action area. 604

Table 105. Ranges to temporary threshold shift for sonar bin MF1 over a representative range of environments within the action area. 604

Table 106. Ranges to temporary threshold shift for sonar bin MF4 over a representative range of environments within the action area. 605

Table 107. Ranges to temporary threshold shift for sonar bin MF5 over a representative range of environments within the action area. 605

Table 108. Ranges to a potentially significant behavioral response for sonar bin HF4 over a representative range of environments within the action area. 606

Table 109. Ranges to a potentially significant behavioral response for sonar bin LF4 over a representative range of environments within the action area. 606

Table 110. Ranges to a potentially significant behavioral response for sonar bin MF1 over a representative range of environments within the action area. 607

Table 111. Ranges to a potentially significant behavioral response for sonar bin MF4 over a representative range of environments within the action area. 608

Table 112. Ranges to a potentially significant behavioral response for sonar bin MF5 over a representative range of environments within the action area. 609

Table 113. Estimated impacts on blue whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area. 610

Table 114. Estimated impacts on fin whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area. 612

Table 115. Estimated impacts on humpback whales by DPS over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area. 614

Table 116. Estimated impacts on Southern Resident killer whales over a seven-year period and over a year of maximum activity as a result of the proposed NWTT activities using sonar and other transducers. 616

Table 117. Southern Resident killer whale DPS estimated maximum annual impacts by training activity (BEH = behavioral harassment). 618

Table 118. Southern Resident killer whale DPS estimated maximum annual impacts by testing activity (BEH = behavioral harassment; TTS = temporary threshold shift). 619

Table 119. Estimated impacts on sei whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area..... 621

Table 120. Estimated impacts on sperm whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area..... 623

Table 121. Estimated impacts on Guadalupe fur seals over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area. 625

Table 122. Ranges to Mortality (in meters) for All Marine Mammal Hearing Groups as a Function of Animal Mass. 644

Table 123. Ranges to non-auditory Injury (in meters) for all marine mammal hearing groups. 645

Table 124. Sound Exposure Level (SEL)-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Low-Frequency Cetaceans. 646

Table 125. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Low Frequency Cetaceans. 647

Table 126. Sound Exposure Level (SEL)-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Mid-Frequency Cetaceans. 648

Table 127. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Mid-Frequency Cetaceans. 649

Table 128. Sound Exposure Level (SEL) Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Otariids. 650

Table 129. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Otariids. 651

Table 130. Estimated impacts on blue whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area..... 652

Table 131. Estimated impacts on fin whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area..... 653

Table 132. Estimated impacts on the humpback whale Mexico DPS every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area. 654

Table 133. Estimated impacts on sei whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area. 656

Table 134. Estimated average behavioral disruptions (i.e., TTS or significant behavioral response) from Navy sonar (and other transducers) and explosives per animal of each species/stock/DPS in the action area. These estimates are based on a year of estimated maximum behavioral disruptions during a year of maximum sonar and explosive activity levels. 665

Table 135. Estimated annual number of mortalities and injuries of ESA-listed Chinook salmon ESUs included in effects analysis for Southern Resident killer whales prey resulting from offshore NWTT explosive activities during a maximum year of explosive activity levels. 670

Table 136. Estimated annual number of mortalities and injuries of non-listed Chinook salmon, coho salmon, and chum salmon ESUs included in effects analysis for Southern Resident killer whales prey resulting from offshore NWTT explosive activities during a maximum year of explosive activity levels. 672

Table 137. Anticipated effects of NWTT explosives on Chinook salmon ESUs expressed as a percent of the population (i.e., hatchery[1] and natural populations) killed and injured annually during a maximum year of explosive activity levels. 678

Table 138. Summary of factors considered in determining the number of individuals in each stock potentially struck by a vessel (M/SI – Mortality/Serious Injury). 688

Table 139. Ranges to non-auditory injury[1] (in meters) for sea turtles exposed to explosives as a function of animal mass. 694

Table 140. Ranges to mortality (in meters) for sea turtles exposed to explosives as a function of animal mass[1] 695

Table 141. Peak pressure based ranges to TTS and PTS (in meters) for sea turtles exposed to explosives 696

Table 142. Sound exposure level-based ranges (in meters) to TTS and PTS for sea turtles exposed to explosives 697

Table 143. Ranges to behavioral response for sea turtles exposed to multiple explosions within any given event 698

Table 144. Maximum annual offshore explosive sources that could be used in-water or at the water's surface. 701

Table 145. Maximum annual inland explosive sources that could be used in-water or at the water's surface. 702

Table 146. Sound exposure criteria used to calculate ranges to effects..... 704

Table 147. Ranges to mortality and injury from explosives for fishes with and without a swim bladder. 704

Table 148. Range to TTS for fishes with a swim bladder from explosives. 705

Table 149. Expected co-occurrence of ESA-listed fish species with Navy explosive ordnance activity in the offshore action area. 714

Table 150. Estimated maximum annual number of ESA-listed salmonids (hatchery fish w/adipose fin intact) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 716

Table 151. Estimated maximum annual number of ESA-listed salmonids (hatchery fish w/adipose clip) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 718

Table 152. Estimated maximum annual number of ESA-listed salmonids (natural fish) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 720

Table 153. Estimated maximum annual number of ESA-listed eulachon from the Southern DPS that would die or be injured by explosive activities in the offshore environment. 723

Table 154. Estimated maximum annual number of ESA-listed salmonids that would die or be injured by explosive activities in Puget Sound. -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 735

Table 155. Maximum number of explosive activities during the Navy's proposed NWTT Training and Testing Activities from November 2020 to November 2027. 739

Table 156. Estimated number of ESA-listed salmonids (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this

lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities. 741

Table 157. Estimated number of ESA-listed salmonids (hatchery produced with intact adipose) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities. 744

Table 158. Estimated number of ESA-listed salmonids (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities. 746

Table 159. Estimated number of ESA-listed Southern DPS Eulachon that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates no estimates of ESA-listed Southern DPS Eulachon affected by Navy testing activities. 749

Table 160. Estimated number of ESA-listed salmonids that would be killed or injured in the inland portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities. 749

Table 161. Percent of each Chinook salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 817

Table 162. Percent of each Chinook salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 818

Table 163. Percent of each Chinook salmon ESU (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 819

Table 164. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, -

indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 820

Table 165. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (hatchery produced with adipose fin-clip) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 821

Table 166. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 822

Table 167. Percent of each ESA-listed chum salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 828

Table 168. Percent of each ESA-listed chum salmon ESU (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 828

Table 169. Average annual impacts every seven years to ESA-listed chum salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore and inland portion of the NWTT action area in seven years. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 829

Table 170. Average annual impacts every seven years to ESA-listed chum salmon ESUs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area in seven years. Unless noted otherwise, -indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury. 829

Table 171. Percent of each ESA-listed coho salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 834

Table 172. Percent of ESA-listed coho salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the

NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 834

Table 173. Percent of ESA-listed coho salmon ESU (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 835

Table 174. Average annual impacts every seven years to ESA-listed coho salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 836

Table 175. Average annual impacts every seven years to coho salmon ESUs (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 836

Table 176. Average annual impacts every seven years to coho salmon ESUs (naturally produced) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury. 837

Table 177. Percent of each steelhead DPS (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 840

Table 178. Percent of each steelhead DPS (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 841

Table 179. Percent of each steelhead DPS (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 842

Table 180. Average annual impacts every seven years to ESA-listed steelhead DPSs (hatchery produced with adipose fin intact) killed or injured in the offshore

and inland portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 843

Table 181. Average annual impacts every seven years to ESA-listed steelhead DPSs (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 844

Table 182. Average annual impacts every seven years to ESA-listed steelhead DPSs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area in seven years. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 845

Table 183. Percent of each sockeye salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 848

Table 184. Percent of each sockeye salmon ESU (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 849

Table 185. Average annual impacts every seven years to ESA-listed sockeye salmon (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 849

Table 186. Average annual impacts every seven years to ESA-listed sockeye salmon (naturally produced) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury. 849

Table 187. Percent of each Southern DPS eulachon that would be killed or injured in the offshore of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected. 852

Table 188. Average annual impacts every seven years to Southern DPS eulachon that would be killed or injured in the offshore of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected. 852

Table 189. The estimated total number of lethal and non-lethal takes of threatened and endangered marine mammals, sea turtles, and fish reasonably certain to occur over a seven-year period as a result of the proposed Navy training and testing activities in the NWTT action area. Numbers in parentheses indicate the estimated maximum annual number of takes likely to occur based on the proposed maximum activity level (except in the case of mortality from vessel strike). 869

LIST OF FIGURES

Page

Figure 1. Navy auditory weighting functions for all marine mammal species groups. 45

Figure 2. TTS and PTS exposure functions for sonar and other acoustic sources for marine mammals (Navy 2018b). 46

Figure 3. Behavioral, TTS, and PTS exposure functions for explosives (Navy 2018b). 48

Figure 4. Behavioral response function for odontocetes showing probability of response as a function of received sound pressure level (Navy 2017a). 52

Figure 5. Behavioral response function for mysticetes (Navy 2017a). 53

Figure 6. Behavioral response function for pinnipeds (Navy 2017a). 53

Figure 7. Auditory weighting function for sea turtles (Navy 2017). 55

Figure 8. Sea turtle TTS and PTS exposure functions for sonar and other transducers (Navy 2017). 57

Figure 9. TTS and PTS exposure functions for impulsive sounds - sea turtles. 60

Figure 10. Offshore winter/spring distribution of blue whale (Navy 2019f). 66

Figure 11. Offshore summer/fall distribution of blue whale (Navy 2019f). 66

Figure 12. Offshore winter/spring distribution of fin whale (Navy 2019f) 67

Figure 13. Offshore summer/fall distribution of fin whale (Navy 2019f). 67

Figure 14. Offshore winter/spring distribution of humpback whale (Central American and Mexico DPSs combined) (Navy 2019f). 68

Figure 15. Offshore summer/fall distribution of humpback whale (Central American and Mexico DPSs combined) (Navy 2019f). 68

Figure 16. Offshore annual distribution of sperm whale (Navy 2019f). 69

Figure 17. Offshore annual distribution of Southern Resident killer whale (J Pod) (Navy 2019f). 69

Figure 18. Offshore January–May distribution of Southern Resident killer whale (K & L pods) (Navy 2019f). 70

Figure 19. Offshore June–December distribution of Southern Resident killer whale (K & L pods) (Navy 2019f). 70

Figure 20. Inland waters winter distribution of Southern Resident killer whale (Navy 2019f). 71

Figure 21. Inland waters spring distribution of Southern Resident killer whale (Navy 2019f). 71

Figure 22. Inland waters summer distribution of Southern Resident killer whale (Navy 2019f). 72

Figure 23. Inland waters fall distribution of Southern Resident killer whale (Navy 2019f). 72

Figure 24. Locations of the 21 marine recovery areas (indicated by dark lines) used to estimate distributions (Weitkamp 2010). 79

Figure 25. Recovery patterns for coded-wire-tagged Chinook salmon. Each horizontal bar represents the percentages of recoveries in the 21 marine recovery areas for a single hatchery run type group (Weitkamp 2010). 80

Figure 26. Estimated proportional spatial distribution by season of fall Chinook salmon originating from 11 different regions. Each row represents the proportion of fish from a region present in each ocean region (rows sum to 1) (Shelton et al. 2019). Acronyms are presented in footnote below. 81

Figure 27. Seafloor resource mitigation areas in the NWTT Study Area. 167

Figure 28. Marine species mitigation areas in the NWTT Study Area. 169

Figure 29. Action area for the Northwest Training and Testing (NWTT) Phase III proposed action. 182

Figure 30. Offshore portion of the NWTT Study Area. 184

Figure 31. Inland waters of the action area. 187

Figure 32. Western Behm Canal, Alaska and the Southeast Alaska Acoustic Measurement Facility 191

Figure 33. Map identifying the range of the endangered blue whale. 242

Figure 34. Map identifying the range of the endangered fin whale. 253

Figure 35. Map showing the distribution of the 14 humpback whale Distinct Population Segments (modified from Bettridge et al. 2015). 262

Figure 36. Proposed critical habitat for the humpback whale Central America Distinct Population Segment 271

Figure 37. Proposed critical habitat for the humpback whale Mexico Distinct Population Segment. 272

Figure 38. Map identifying the range of the endangered Southern Resident killer whales distinct population segment of killer whale. Approximate April through October distribution of the Southern Resident killer whales distinct population

segment of killer whale (shaded area) and range of sightings (diagonal lines) (Carretta et al. 2016c). 275

Figure 39. Map identifying inland designated critical habitat for the endangered Southern Resident killer whales DPS of killer whale 291

Figure 40. Existing and proposed critical habitat for Southern Resident killer whales. 292

Figure 41. Map identifying the range of the endangered sei whale. 295

Figure 42. Map identifying the range of the endangered sperm whale. 301

Figure 43. Guadalupe fur seal historic range. 311

Figure 44. Map identifying the range of the endangered leatherback sea turtle. From NMFS http://www.nmfs.noaa.gov/pr/species/turtles/leatherback.html, adapted from (Wallace et al. 2010). 316

Figure 45. Leatherback sea turtle designated critical habitat on the Pacific Coast. 324

Figure 46. Geographic range and designated critical habitat of California coastal ESU Chinook salmon. 326

Figure 47. Geographic range and designated critical habitat of Central Valley spring-run ESU Chinook salmon. 330

Figure 48. Geographic range and designated critical habitat of Lower Columbia River ESU Chinook salmon. 334

Figure 49. Geographic range and designated critical habitat of Puget Sound ESU Chinook salmon. 338

Figure 50. Geographic range and designated critical habitat of the Sacramento River winter-run ESU of Chinook salmon. 345

Figure 51. Geographic range of Snake River fall-run ESU Chinook salmon. 349

Figure 52. Geographic range and major population groups of Snake River spring/summer-run ESU Chinook salmon. 353

Figure 53. Geographic range and designated critical habitat of Chinook salmon, upper Columbia River ESU. 357

Figure 54.Geographic range and designated critical habitat of Chinook salmon, upper Willamette River ESU. 361

Figure 55. Geographic range and designated critical habitat of chum salmon, Columbia River ESU. 364

Figure 56. Geographic range and designated critical habitat of chum salmon, Hood Canal ESU. 368

Figure 57. Geographic range of Coho salmon, Central California Coast ESU. 373

Figure 58. Geographic range and designated critical habitat of Coho salmon, Lower Columbia River ESU. 378

Figure 59. Geographic range and designated critical habitat of coho salmon, Oregon coast ESU. 382

Figure 60. Geographic range of the Southern Oregon/Northern California ESU of Coho salmon. 387

Figure 61. Map identifying the range of the eulachon Southern distinct population segment (NMFS 2016f). 390

Figure 62. Map of designated critical habitat for the eulachon Southern distinct population segment. 393

Figure 63. Geographic range (within the contiguous U.S., not including Alaska range) and designated critical habitat for the green sturgeon Southern DPS. 395

Figure 64. Number of individual green sturgeon detections at receivers off the coast of Washington State from May 2019 to September 2019 (unpublished map provided by Joe Smith, NMFS Northwest Fisheries Science Center, January 2, 2020). 398

Figure 65. Green sturgeon Southern DPS critical habitat. 401

Figure 66. Geographic range of Sockeye salmon, Snake River ESU. 404

Figure 67. Geographic range and designated critical habitat of California Central Valley Steelhead. 407

Figure 68. Geographic range and designated critical habitat of Central California Coast Steelhead. 411

Figure 69. Geographic range and designated critical habitat of Lower Columbia River steelhead. 414

Figure 70. Geographic range and designated critical habitat of Middle Columbia River steelhead. 419

Figure 71. Geographic range and designated critical habitat of Northern California DPS steelhead. 422

Figure 72. Geographic range and designated critical habitat of Puget Sound DPS steelhead. 426

Figure 73. Geographic range and designated critical habitat of Snake River Basin steelhead. 429

Figure 74. Geographic range and designated critical habitat of South-Central California Coast steelhead. 433

Figure 75. Geographic range and designated critical habitat of upper Columbia River steelhead. 435

Figure 76. Geographic range and designated critical habitat of upper Willamette River steelhead. 438

Figure 77. Map identifying the range of the Puget Sound/Georgia Basin distinct population segment bocaccio. 441

Figure 78. Designated critical habitat for ESA-listed rockfishes. 445

Figure 79. Yelloweye Rockfish, Puget Sound/Georgia Basin DPS range and designated critical habitat. 447

Figure 80. Guadalupe fur seal annual strandings in California, 2013 to present. Orange and red bars indicate unusual mortality event years (NMFS 2020a). 459

Figure 81. Guadalupe fur seal annual strandings in Oregon and Washington, 2013 to present. Orange/red – Oregon; Blue/light blue – Washington (NMFS 2020a). 460

Figure 82. Trend in total confirmed whale entanglements per year detected off the U.S. west coast from 2001 to 2016, and estimated humpback whale population size (Santora et al. 2020). 466

Figure 83. Commercial Ship Automated Identification System Position Data for Washington State for January 2009 468

Figure 84. Number of confirmed vessel collisions with ESA-listed species in California from 1986-2019. 470

Figure 85. Sediment contamination levels within Southern Resident killer whale inland critical habitat areas (note: military sites excluded from critical habitat designation are not shown on these maps) (NMFS 2006a). 473

Figure 86. Map of current rail routes, interstate pipelines, and barges transporting crude oil across the West Coast (from: http://oilspilltaskforce.org/wp-content/uploads/2019/07/BCStates_crude_oil_movement_2019_rev4.pdf. Accessed 6 January 2020). 476

Figure 87. Marine oil spill incidents on the Pacific West Coast from 1971-2017 within areas under consideration for Southern Resident killer whale proposed critical habitat designation (NMFS 2019e). 477

Figure 88. Frequency ranges of ESA-listed fishes affected by NWTT activities. Sources: (Hawkins and Johnstone 1978; Popper 2008; Popper et al. 2007a; Popper et al. 2014b; Tavolga and Wodinsky 1963). 566

Figure 89. Two hypothetical temporary threshold shifts. 579

Figure 90. Blue whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 611

Figure 91. Fin whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 613

Figure 92. Humpback whale (Central America and Mexico DPSs) impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 615

Figure 93. Southern Resident killer whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 617

Figure 94. Sei whales impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 622

Figure 95. Sperm whales impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 624

Figure 96. Guadalupe fur seal impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category. 626

Figure 97. Estimated impacts on blue whales per year from explosives used during NWTT by region and activity category. 652

Figure 98. Estimated impacts on fin whales per year from explosives used during NWTT by region and activity category. 653

Figure 99. Estimated impacts on the humpback whale Mexico DPS per year from explosives used during NWTT by region and activity category. 654

Figure 100. Estimated impacts on sei whales per year from explosives used during NWTT by region and activity category. 657

Figure 101. Overlap of proposed humpback whale critical habitat (Units 11 through 15) with the NWTT offshore action area. 755

Figure 102. Overlap of proposed humpback whale critical habitat (Unit 10) and Western Behm Canal portion of the NWTT action area. 756

1 INTRODUCTION

The Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7(a)(2) of the ESA requires Federal agencies to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Federal agencies must do so in consultation with the National Marine Fisheries Service (NMFS) for threatened or endangered species (ESA-listed) or designated critical habitat that may be affected by the proposed action that are under NMFS jurisdiction (50 C.F.R. §402.14(a)). If a Federal action agency determines that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS concurs with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

The Federal action agency shall confer with the NMFS under ESA Section 7(a)(4) for species under NMFS jurisdiction on any action which is likely to jeopardize the continued existence of any proposed species or result in the destruction or adverse modification of proposed critical habitat (50 C.F.R. §402.10). If requested by the Federal agency and deemed appropriate, the conference may be conducted in accordance with the procedures for formal consultation in §402.14.

Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat (together termed ESA-listed resources). If NMFS determines that the action is likely to jeopardize ESA-listed species or destroy or adversely modify critical habitat, in accordance with the ESA Subsection 7(b)(3)(A), NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If an incidental take is expected, section 7(b)(4) requires NMFS to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts and terms and conditions to implement the reasonable and prudent measures. NMFS, by regulation, has determined that an ITS must be prepared when take is "reasonably certain to occur" as a result of the proposed action. 50 C.F.R. §402.14(g)(7).

The action agencies for this consultation are the United States Navy (Navy) and NMFS's Permits and Conservation Division (Permits Division). The Navy proposes to conduct testing and training activities in the Northwest Training and Testing (NWTT) Study Area and the Permits Division proposes to promulgate regulations pursuant to the Marine Mammal Protection Act (MMPA) of 1972, as amended (16 U.S.C. 1361 et seq.) for the Navy to "take" marine mammals incidental to NWTT activities. The regulations propose the issuance of a Letter of Authorization

(LOA) that will authorize the Navy to "take" marine mammals incidental to its proposed action, pursuant to the requirements of the MMPA.

This consultation was completed in accordance with section 7(a)(2) of the statute (16 U.S.C. 1536 (a)(2)), associated implementing regulations (50 C.F.R. Part 402), and agency policy and guidance by the NMFS Office of Protected Resources ESA Interagency Cooperation Division (hereafter referred to as "we" or "us"). This biological and conference opinion (opinion) and ITS were prepared by the NMFS Office of Protected Resources ESA Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 C.F.R. Part 402 and specifically 50 C.F.R. §402.14. This opinion reflects the best available scientific information on the status and life history of ESA-listed species, the stressors resulting from the proposed action, the likely effects of those stressors on ESA-listed species and their habitats, the consequences of those effects to the fitness and survival of individuals, and the risk that those consequences pose to the survival and recovery of the threatened or endangered populations they represent.

Updates to the regulations governing ESA interagency consultation (50 C.F.R part 402) were effective on October 28, 2019 (84 FR 44976). As the preamble to the final rule adopting the regulations noted, "[t]his final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice." We have reviewed the information and analyses relied upon to complete this biological opinion in light of the updated regulations and conclude this opinion is fully consistent with the updated regulations.

This document represents NMFS' opinion on the effects of the proposed NWTT activities and the Permits Division's promulgation of regulations pursuant to the MMPA for the Navy to "take" marine mammals incidental to NWTT activities on endangered and threatened species and critical habitat that has been designated for those species. A complete record of this consultation is on file at the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The Navy proposes to conduct training and testing activities within the NWTT Study Area starting in November 2020 and continuing into the reasonably foreseeable future. These activities are hereafter referred to as "Phase III" activities. Navy training and testing activities have been ongoing in this same general geographic area for several decades and as indicated below, many of these activities have been considered in previous ESA section 7 consultations (i.e., as detailed below, in consultations that considered Phase I and Phase II Navy actions).

Between 2010 and 2012, NMFS issued a series of biological opinions on Navy training and testing activities proposed for the Pacific Northwest Training Range Complex (NWTRC). On November 12, 2010, NMFS issued a biological opinion on the U.S. Navy's military readiness activities for five years and the NMFS's Permits Division's proposed issuance of a one-year

LOA to be valid from November 2010 through November 2011. On November 9, 2011, NMFS issued a biological opinion on the U.S. Navy's military readiness activities and the NMFS's Permits Division's proposed issuance of a one-year LOA to be valid from November 2011 through November 2012. On October 16, 2012, NMFS issued a biological opinion on the U.S. Navy's military readiness activities and the NMFS's Permits Division's proposed issuance of a three-year LOA to be valid from November 2012 through November 2015, covering the remaining effective period of the five-year "take" regulations for the NWTRC. Collectively, NMFS and the Navy referred to the activities that were the subject of these consultations as Phase I.[1]

On September 25, 2013, the United States District Court, Northern District of California, made a ruling (*Intertribal Sinkyone Wilderness Council v. NMFS*, 970 F. Supp. 2d 988 2013) on the October 2012 biological opinion (referred to by the Court as the 2012 LOA Biological Opinion), which the Court subsequently summarized as follows: "First, the court held that it was an abuse of discretion in regard to its duty to use the best scientific data available for the NMFS to fail to consider the 2010 and 2011 dolphin studies [James J. Finneran, et al., *Frequency-dependent and Longitudinal Changes in Noise-induced Hearing Loss in a Bottlenose Dolphin*, 128 J. Acoust. Soc. Am. (2) 567-570 at 657, 568 (2010); James J. Finneran, et al., *Subjective Loudness Measurements and Equal Loudness Contours in a Bottlenose Dolphin*, 130 J. Acoust. Soc. Am. (5) at 3124-3136 (2011)] in the 2012 LOA Biological Opinion. (Doc. 66, 15:28 - 16- 1.). Second, the court held because the NMFS abused its discretion in failing to consider the best scientific information available in its 2012 LOA Biological Opinion, its estimates of the amount of take in the Incidental Take Statement, like its jeopardy analysis, was not based on the best scientific and commercial data available. The court found, therefore, that the NMFS abused its discretion in the issuance of the Incidental Take Statement. *Id*. at 17:17-20. Finally, the court held that it was an abuse of discretion for the NMFS to define the 'agency action' to be reviewed under ESA as the five-year period permitted under the MMPA. *Id*. at 23:17-19. These rulings impose on the NMFS a duty to correct its abuses of discretion. To the extent that the corrections require issuance, re-issuance or amendment of documents under ESA, such issuance, re-issuance or amendment shall be completed no later than August 1, 2014." Nov. 26, 2013 Remand Order at 3-4.

On March 5, 2014, NMFS reinitiated consultation with the U.S. Navy and NMFS's Permits Division to amend the 2012 biological opinion to address the Court's ruling and to assess new scientific information that had become available since issuance of the 2012 opinion. On August 1, 2014, NMFS issued an amended biological opinion that superseded the 2010 programmatic pertaining to the NWTRC and 2012 biological opinions.

During the timeframe the NWTRC consultation was reinitiated, the U.S. Navy separately submitted a request for two LOAs for the incidental taking of marine mammals during the

[1] Note: Since this was the first set of MMPA incidental take regulations, ESA biological opinions, and National Environmental Policy Act Environmental Impact Statements for Navy at-sea training and testing activities, these activities were referred to as Phase I activities. Subsequent phases are referred to as Phase II, Phase III, etc.

conduct of Phase II training activities within the NWTT Study Area, which includes the NWTRC, for the period of August 2015 through August 2020, and requested initiation of section 7 formal consultation on the NWTT action. On November 9, 2015, NMFS issued a biological opinion on the U.S. Navy's military readiness activities and the NMFS's Permits Division's proposed issuance of a five-year LOA to be valid from November 2015 through November 2020. The activities that were the subject of the 2015 opinion are referred to as NWTT Phase II.

The activities considered in these prior consultations were similar to those proposed for NWTT Phase III that are the subject of this consultation and included the use of active sonar, explosives, and vessels. Where incidental take of ESA-listed marine mammals was anticipated, these prior consultations also considered NMFS Permits Division's promulgation of regulations and issuance of letters of authorization pursuant to the MMPA for the Navy to "take" marine mammals incidental to their activities. Each of these previous opinions concluded that the Navy's and NMFS Permits Division's proposed actions would not jeopardize the continued existence of threatened or endangered species or destroy or adversely modify designated critical habitat.

This opinion is based on information provided by the Navy during pre-consultation technical assistance, the Navy's Biological Assessment (BA; version 2, February 26, 2019), draft NWTT Phase III Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS), and supplemental information provided throughout the consultation process. This opinion also considers information provided by NMFS' Permits Division, including its request for section 7 consultation under the ESA, which included the proposed Federal regulations under the MMPA proposing to authorize the incidental take of marine mammals, including ESA-listed marine mammals, specific to the proposed activities (80 FR 31737) and related draft letters of authorization. Also considered were draft or final recovery plans and status reviews for the endangered or threatened species that are considered in this document, and publications that we identified, gathered, and examined from the public scientific literature, including new information that has become available since the issuance of the previous biological opinions including those mentioned above. Collectively, we consider the foregoing to comprise the best scientific information available for the consultation and this opinion.

1.2 Consultation History

Our communication with the Navy and NMFS' Permits Division regarding this consultation is summarized below:

On April 1, 2019, the Navy provided us with the Draft EIS for NWTT.

On June 10, 2019, the Navy provided us with a draft NWTT BA. The draft BA included a section for Essential Fish Habitat consultation under the Magnuson-Stevens Fishery Conservation Management Act.

On July 8, 2019, we provided the Navy with comments on their draft BA.

On September 17, 2019, the Navy provided us with an updated and mutually agreed to timeline (i.e., Navy NWTT stick chart dated 9-13-2019) for the NWTT Phase III EIS/OEIS, NMFS' issuance of an incidental take authorization pursuant to the MMPA, and the ESA section 7 consultation schedule. The revised timeline extended the consultation beyond 135 days with completion of the biological opinion on or before November 1, 2020.

On September 19, 2019, NMFS issued a proposed rule to revise the critical habitat designation for the Southern Resident killer whale Distinct Population Segment (DPS) pursuant to section 4 of the ESA to include six new offshore areas along the U.S. West Coast (84 FR 49214).

On October 9, 2019, NMFS issued a proposed rule to designate critical habitat for the endangered Western North Pacific DPS, the endangered Central America DPS, and the threatened Mexico DPS of humpback whales pursuant to section 4 of the ESA (84 FR 54354).

On October 23, 2019, we received the Navy's revised BA and a request for formal section 7 consultation for proposed NWTT Phase III activities. The Navy's BA did not include a request to conference on proposed critical habitat for the Southern Resident killer whale DPS (84 FR 49214) or the Central America and Mexico DPSs of humpback whales (84 FR 54354).

On October 25, 2019, we emailed the Navy indicating that the NMFS West Coast Regional Office would be conducting the Essential Fish Habitat consultation under the Magnuson-Stevens Fishery Conservation and Management Act for NWTT Phase III. We provided the NMFS West Coast Regional Office contacts for the Navy to reach out to regarding the Essential Fish Habitat consultation.

On November 8, 2019, we provided the Navy with comments on their revised BA.

On November 18, 2019, we held a conference call with the Navy to discuss our comments and questions regarding the Navy's BA.

On November 22, 2019, the Navy sent a comment resolution matrix (spreadsheet) to us addressing, in writing, remaining comments on the revised version of the BA.

On December 12, 2019, the Navy sent us a deconstruction matrix providing additional details on the NWTT Phase III activities.

From December 13, 2019 through January 22, 2020, additional information from the Navy to assist us with our analysis of the effects on explosives on fish for the Phase III proposed action. This included updated tables provided by the Navy for the NWTT Phase II opinion, range to effects from explosives for the injury threshold criteria used by NMFS in other Phase III consultations.

On January 31, 2020, we held a conference call with the Navy to discuss conferencing on proposed critical habitat for Central America DPS and Mexico DPS of humpback whales, and for the Southern Resident killer whale DPS. As a followup to this call, the Navy sent us an email indicating that they would like to conference on proposed critical habitat for these species. The Navy also indicated that they intend to provide a conference package by early April 2020 to facilitate the inclusion of these designation actions in the formal consultation for the NWTT action.

On February 5, 2020, the Navy sent us a revised version of the NWTT BA.

On February 11, 2020 we responded to the Navy indicating that the version of the BA submitted on February 5, 2020 was complete. We indicated that during the consultation process additional information may be requested and additional mitigation measures may be proposed to the Navy to minimize impacts to ESA-listed resources based on our effects analyses. Since the Navy's proposed action is interrelated with the NMFS's Permits Division's proposed issuance of regulations in accordance with the MMPA, initiation of formal consultation would commence once we receive and accept as complete the NMFS's Permits Division's initiation package.

On February 11, 2020 we requested additional information from the Navy to assist with our biological opinion effects analysis. The Navy provided written responses to our questions on February 26, 2020.

On March 13, 2020, we sent Navy a document proposing additional mitigation measures to minimize the adverse effects of NWTT activities on ESA-listed resources. Included in this communication was 1) a proposal to only conduct offshore mine counter-measure and neutralization testing in water depths greater than 650 feet, and 2) a request for the Navy to consider additional mitigation measures to reduce the impacts from several activities that, based on the Navy's quantitative analysis, are anticipated to result in "take" of Southern Resident killer whales.

On March 26, 2020, the Navy provided a response our March 13 document requesting additional mitigation measures. The Navy indicated that restricting offshore mine counter-measure and neutralization testing to water depths greater than 650 feet would not be practical to implement with respect to allowing the Navy to meet mission requirements. As an alternative, the Navy indicated they were formally vetting the following additional mitigation measures: 1) For personnel safety, mine counter-measure and neutralization testing activities will be conducted during daylight hours only and in Beaufort Sea state 3 or less, 2) The Navy will only conduct up to one explosive mine counter-measure and neutralization testing event annually within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area, with the exception of any testing on Quinault Range Site, and 3) Within the Juan de Fuca Whale Feeding Habitat Mitigation Area, the Navy will not conduct explosive mine counter-measure and neutralization testing. For Southern Resident killer whales impacts, the Navy indicated they were formally

vetting two geographic mitigation measures to reduce potential impacts from surface ship hull-mounted mid-frequency 1 (MF1) active sonar and other potential stressors associated with Undersea Warfare Testing.

On April 24, 2020, we held a conference call with the NMFS West Coast Region lead on the Magnuson Steven Act Essential Fish Habitat (EFH) consultation with the Navy on their proposed NWTT Phase III action. The purpose of the call was to share information regarding our analysis of the effects of NWTT explosives on ESA-listed fish species.

On May 27, 2020 we received a request for formal consultation from the NMFS Permits Division, along with a proposed rule, for its promulgation of regulations and issuance of a letter of authorization pursuant to the MMPA for the U.S. Navy to "take" marine mammals incidental to Northwest Training and Testing activities from November 2020 through November 2027.

On June 2, 2020, we initiated formal consultation with the Navy on their proposed NWTT Phase III action and with the NMFS Permits Division on its proposed MMPA action.

On June 4, 2020, we sent the Navy a spreadsheet showing the density calculations used for our analysis of the effects of NWTT explosives on salmonids and eulachon. An updated version of this spreadsheet was sent to the Navy on July 21, 2020.

On June 8, 2020, we sent the Navy a proposal for seasonal/area mitigation restriction on the offshore mine counter-measure and neutralization testing activity. To minimize the impacts of this activity on ESA-listed resources (including many fish species, humpback whales, and Southern Resident killer whales, and critical habitat for several species) we proposed the following conservation measure: *The Navy will not conduct offshore mine countermeasure testing, or any other activity involving the use of in-water explosives, in the offshore portion of the action area from September 1 through June 30 in areas with water depths of less than 100 meters.*

On June 23, 2020, we received an email from the NMFS lead on the EFH consultation on the NWTT action requesting 1) that we review their EFH analysis for errors or inconsistencies with our biological opinion effects analysis, and 2) that we coordinate delivery of their EFH consultation document to the Navy through NMFS Headquarters General Counsel.

On June 25, 2020, the Navy responded to our proposed seasonal/area mitigation restriction on the offshore mine counter-measure and neutralization testing activity. The Navy indicated that our proposed conservation measure would cause more than a minor change to the proposed testing activity and would not be practical for the Navy to implement. The Navy's offered a counter-proposal (pending senior Navy Flag-level approval) as follows: *Within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area, the Navy will conduct a maximum of one explosive mine countermeasure and neutralization testing event <u>from October 1 through</u>*

June 30. In their response the Navy also indicated that mine countermeasure and neutralization testing using explosives within bin E7 would be no closer to shore than six nautical miles in the Quinault Range Site.

On June 26, 2020, NMFS and the Navy held a conference call to discuss the counter-proposal and other NWTT mitigation related topics. During this call NMFS requested the following changes to the proposed language: 1) reword mitigation to include limitation on the number of explosives during certain times of year, rather than just number of events as previously proposed by the Navy, and 2) Add language that the Navy would "to the extent practicable" avoid using explosives within 20 nautical miles (nm) from shore during certain times of year.

On June 29, 2020, we provided the Navy and the NMFS Permits Division with our draft biological opinion on the NWTT Phase III proposed action.

On July 2, 2020, the Navy sent a revised response to our proposed seasonal/area mitigation for the mine countermeasure and neutralization testing activity, which was modified to address the two suggested changes discussed during the June 26, 2020 call (see above).

On July 7, 2020, the Navy submitted an ESA section 7 Conference document for the NWTT proposed action on proposed critical habitat for Southern Resident Killer Whale and Central America DPS and Mexico DPS of humpback whales.

On July 15, 2020, NMFS emailed the Navy regarding their July 2 counterproposal on additional mitigation measures for the NWTT mine countermeasure and neutralization testing activity. We requested the Navy include a seven year limit on the number of explosives that could be used seasonally from October through June (in addition to Navy proposed annual limit). We also reiterated our concerns that the seasonal limit on explosives did not include the month of September and requested the following change in mitigation language: *The Navy will, to the maximum extent practicable, conduct mine countermeasure and neutralization testing from July 1 through August 31 when operating within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area*.

On July 15, 2020, the Navy provided us with their comments on the NWTT Phase III draft biological opinion. In addition to comments within the document itself, the Navy provided a separate document with a review of the green sturgeon impact analysis for mine countermeasure and neutralization activities.

On July 17, 2020, the Navy emailed us indicating that they agreed to change their proposed seasonal mitigation for the mine countermeasure and neutralization testing activity to include a seven year limit on explosives from October through June. The Navy also indicated that they would not change the language in their proposed mitigation to minimize the use of explosives in

the month of September as it would not be practicable to conduct all mine countermeasure and neutralization testing during the short, two month (i.e., July-August) window of time.

On July 17, 2020, the Navy provided us with an update NWTT Phase III consultation timeline indicating a final signed biological opinion date of October 23, 2020.

On July 21, 2020, we held a conference call with the Navy to discuss and address their comments on the draft opinion. From July 23 through August 20, 2020 we continued to work with the Navy on improving our analysis of the effects of explosives on fish.

On July 30, 2020, we reiterated our request for the Navy to provide range to effects from explosives for the injury threshold criteria used by NMFS in other Phase III consultations (i.e., > 207 and > 213 sound pressure level (SPL) peak thresholds for fishes with and without swim bladders). This request was initially made in an email to the Navy on January 22, 2020.

On August 4, 2020, the Navy indicated that the proposed mitigation measures for the mine countermeasure and neutralization testing activity outlined in their July 16, 2020 document (Navy 2020d) had been approved by Navy senior leadership and would be incorporated into the Final SEIS/OEIS. The Navy also provided additional information, as requested by NMFS during the July 21 comment resolution call, on the location of moored mines (used during this activity) in relation to the seafloor.

On August 10, 2020, a conference call was held with the Navy, U.S. Fish and Wildlife Service (USFWS), and NMFS. The USFWS (Washington Fish and Wildlife Office), proposed this joint meeting between all three agencies to discuss and coordinate on mitigation measures specific to the mine countermeasure and neutralization testing activity in the offshore waters of the NWTT Study Area. The USFWS expressed concerns about the effects of this activity on the endangered marbled murrelet and proposed an additional mitigation measure that would restrict the use of E4 explosives within 4.6 nautical miles from shore.

On September 29, 2020, we sent the Navy the draft biological opinion and requested a review limited in scope to identifying errors, misstatements, omissions, or misleading information (i.e., often referred to by the Navy as a "fatal flaw" review). The Navy provided their comments on the draft opinion on October 4, 2020.

2 THE ASSESSMENT FRAMEWORK

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 C.F.R. §402.02).

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of an ESA-listed species (50 C.F.R. §402.02).

An ESA section 7 assessment involves the following steps:

1) We describe the proposed action (Section 3) and the action area (Section 4).
2) We deconstruct the action into the activities such that we can identify those aspects of the proposed action that are likely to create pathways for adverse impacts to ESA-listed species and/or designated critical habitat. These pathways or "stressors" may result in effects on the physical, chemical, and biotic environment within the action area (Section 5). We also consider the spatial and temporal extent of those stressors.
3) We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time (Section 6). During consultation, we determined that some ESA-listed species that occur in the action area were not likely to be adversely affected by the proposed action (Section 6.1). We summarize our findings, and do not carry those species forward in this opinion as species that are not likely to be adversely affected by the proposed action cannot be jeopardized by the proposed action. We then describe the status of those species that are likely to be adversely affected by the proposed action (Section 6.2).
4) We describe the environmental baseline in the action area (Section 7). Environmental baseline refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 C.F.R. §402.02).

5) We evaluate the effects of the action on ESA-listed species and designated critical habitat (Section 8).

   a) During our evaluation, we determined that some stressors were not likely to adversely affect ESA-listed species (or categories of ESA-listed species; e.g., marine mammals) (Section 8.1) and did not carry them forward for further evaluation as they cannot contribute to fitness consequences, population level effects or jeopardy. The stressors that we determined were likely to adversely affect ESA-listed species or critical habitats were carried forward for additional analyses (Section 8.2).

   b) For those stressors likely to adversely affect ESA-listed species (Section 8.2), we identify the number, age (or life stage), and gender if possible and if needed, of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. This is our exposure analysis.

   c) We evaluate the available evidence to determine how individuals of those ESA-listed species are likely to respond given their probable exposure. This is our response analysis.

   d) Our effects analysis for critical habitat (Section 8.3) considers the impacts of the proposed action on the essential habitat features and conservation value of designated (or proposed) critical habitat within the action area.

6) We describe the cumulative effects in the action area (Section 9). "Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02)

7) We integrate and synthesize the above factors (Section 10) by adding the effects of the action and cumulative effects to the environmental baseline and in light of the status of the species and critical habitat, formulate our opinion as to whether the action would reasonably be expected to:

   a) Reduce appreciably the likelihood of both the survival and recovery of the ESA-listed species in the wild by reducing its reproduction, numbers, or distribution; or

   b) Appreciably diminish the value of designated (or proposed) critical habitat as a whole for the conservation of an ESA-listed species.

8) We state our conclusions regarding whether the action is likely to jeopardize the continued existence of ESA-listed species or result in the destruction or adverse modification of designated (or proposed) critical habitat (Section 11).

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated (or proposed) critical habitat, we must identify a reasonable and prudent alternative(s) to the action that would allow the action to proceed in compliance with ESA section 7(a)(2). The reasonable and prudent alternative(s) also must meet other regulatory requirements.

If incidental take of ESA-listed species is expected, section 7(b)(4) of the ESA requires that we provide an ITS (Section 12) that specifies the amount or extent of take, the impact of the take, reasonable and prudent measures that are necessary or appropriate to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures (ESA section 7 (b)(4); 50 C.F.R. §402.14(i)). The ITS must also include measures to ensure the action is carried out in compliance with any incidental take authorization provided under the MMPA, Section 101(a)(5). ESA section (7)(o)(2) provides that compliance by the action agency with the terms and conditions exempts any incidental take from the prohibitions of take in ESA section 9(b) and regulations issued pursuant to ESA section 4(d).

"Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. NMFS has not yet defined "harass" under the ESA in regulation. However, on December 21, 2016, NMFS issued interim guidance on the term "harass," defining it as an action that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering" (NMFS 2016h). For purposes of this consultation, we relied on NMFS' interim definition of harassment to evaluate when the proposed activities are likely to harass ESA-listed species. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity.

Consistent with the ESA section 7(a)(1), we also provide discretionary conservation recommendations (Section 13) that may be implemented by the action agency (50 C.F.R. §402.14(j)). Finally, we identify the circumstances in which reinitiation of consultation (Section 14) is required (50 C.F.R. §402.16).

2.1 Evidence Available for this Consultation

To conduct these analyses and to comply with our obligation to use the best scientific and commercial data available, we considered all lines of evidence available through published and unpublished sources that represent evidence of adverse consequences or the absence of such consequences. We conducted electronic literature searches throughout this consultation, including within the NMFS Office of Protected Resources' electronic library. We examined the Navy's BA (Navy 2020e), the Navy's Draft Environmental Impact Statement (DEIS) (Navy 2019a), the literature that was cited in the Navy's BA and DEIS, and any articles we collected through our electronic searches. We also evaluated the Navy's annual and comprehensive monitoring reports required under the existing MMPA rule and LOAs and the previous biological opinion for current training and testing activities occurring in the same geographic area. These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS' jurisdiction that may be affected by the proposed action to draw conclusions on risks the action may pose to the

continued existence of these species and the value of designated (or proposed) critical habitat for the conservation of ESA-listed species. In addition, we engaged regularly with the Navy to discuss new science and technical issues as part of the ongoing adaptive management program for Navy training and testing and incorporated new information obtained as a result of these engagements in this consultation.

As is evident later in this opinion, many of the stressors considered in this opinion involve sounds produced during Navy training and testing activities. Considering the information that was available, this consultation and our opinion includes uncertainty about the basic hearing capabilities of some marine mammals, sea turtles, and fishes; how these taxa use sounds as environmental cues; how they perceive acoustic features of their environment; the importance of sound to the normal behavioral and social ecology of the different species; the mechanisms by which human-generated sounds affect the behavior and physiology (including the non-auditory physiology) of exposed individuals; and the circumstances that are likely to produce outcomes that have adverse consequences for individuals and populations of exposed species. We relied on conservative assumptions when addressing such uncertainties in our analyses of the potential effects of NWTT training and testing activities on ESA-listed species and their proposed or designated critical habitat in the NWTT action area.

The sections below discuss NMFS' approach to analyzing the effects of sound produced by Navy training and testing activities in the NWTT action area on ESA-listed marine mammals, sea turtles, and fishes. The estimates of the number of ESA-listed marine mammals and sea turtles exposed to sound from Navy training and testing, as well as the magnitude of effects from these exposures (e.g., injury, hearing loss, behavioral response), are from the Navy's acoustic effects analysis described in detail in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018c). NMFS considers the modeling conclusions from the Navy's analysis to represent the best available data on exposure of marine mammals and sea turtles to acoustic stressors from the proposed action.[2] Our analysis of the effects of and potential consequences of such exposures is included in Section 8 of this opinion.

2.2 Acoustic Effects Analysis for Marine Mammals and Sea Turtles

Acoustic stressors include acoustic signals emitted into the water for a specific purpose (e.g., by active sonars and air guns), as well as incidental sources of broadband sound produced as a byproduct of vessel movement, aircraft transits, pile driving and removal, and use of weapons or other deployed objects. Explosives also produce broadband sound but are characterized separately from other acoustic sources due to their unique energetic characteristics. To estimate impacts from acoustic stressors associated with proposed training and testing activities, the Navy performed a quantitative analysis to estimate the number of instances that could affect ESA-

[2] The Navy's acoustic effects analysis did not estimate the number of ESA-listed fish exposed to NWTT acoustic stressors.

listed marine mammals and sea turtles and the magnitude of that effect (e.g., injury, hearing loss, behavioral response). The quantitative analysis utilizes the Navy's Acoustic Effects Model (NAEMO) and takes into account criteria and thresholds used to predict impacts in conjunction with spatial densities of species within the action area.

A summary of the quantitative analysis is provided below. A more detailed explanation of this analysis can be found in the Navy's technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018c). NMFS verified the methodology and data used by the Navy in this analysis and, unless otherwise specified in Section 8 of this opinion, accepted the modeling conclusions on exposure of marine mammals and sea turtles to sound generated by the proposed action. NMFS considers the modeling conclusions from the Navy's analysis to represent the best available data on exposure of marine mammals and sea turtles to acoustic stressors from the proposed action and the estimates of take resulting from this analysis are reasonably certain to occur.

2.2.1 Navy Acoustic Effects Model (NAEMO)

NAEMO calculates sound energy propagation from sonars and other transducers (as well as air guns and explosives) during naval activities and the sound received by animat dosimeters. Animat dosimeters are virtual representations of marine mammals and sea turtles distributed in the area around the modeled naval activity. Each of the animat dosimeters records its individual sound "dose." The model bases the distribution of animats over the action area on the density values (see Section 2.4.1 below) in the Navy Marine Species Density Database (Navy 2019f) and distributes animats in the water column proportional to the known time that species spend at varying depths.

Physical environment data plays an important role in acoustic propagation of underwater sound sources used in the impact modeling process (Navy 2019c). Physical environment parameters that influence propagation modeling include bathymetry, seafloor composition/sediment type, wind speed, and sound speed profiles. NAEMO accounts for environmental variability in sound propagation with both distance and depth, as well as boundary interactions, when computing the received sound level of the animats. The model conducts a statistical analysis based on multiple model runs to compute the potential acoustic effects on animals. The number of animats for which the thresholds of effects is exceeded is tallied to estimate the number of times marine mammals or sea turtles could be affected by the aspects of the proposed activity that generate sound.

Marine mammal and sea turtle data input to the NAEMO include densities (discussed above), group size, depth distribution, and (for mammals) stock breakouts (Navy 2019c). Since many marine mammals are known to travel and feed in groups, species-specific group sizes are incorporated into animat distributions. Species specific group sizes are estimated using literature review, survey data, and density data, and uncertainty of group size estimates are statistically represented by the standard deviation. The model accounts for depth distributions by changing

each animat's depth during the simulation process according to the typical depth pattern observed for each species**.** Depth distribution information was collected by a literature review and is presented as a percentage of time the animal typically spends within various depth bins in the water column. Many marine mammals species are divided into multiple stocks based on life history and genetic stock structure for management purposes. For some stocks there is enough survey information to support stock-specific density models. In these cases, a density layer for the stock is provided and is modeled independently of other stocks. In other cases, predicted impacts were assigned by stock, as opposed to the species as a whole (Navy 2019c).

The model estimates the impacts caused by individual training and testing events. During any individual modeled event, impacts on individual animats are considered over 24-hour periods. The animats do not represent actual animals, but rather allow for a statistical analysis of the number of instances during which marine mammals or sea turtles may be exposed to sound levels resulting in an effect. Therefore, the model estimates the annual number of exposures that may result in each type of effect but does not estimate the number of individual marine mammals or sea turtles that may be affected (Navy 2018d). Some individuals may be exposed more than once per year but the model does not estimate whether a single individual is exposed multiple times.

As described further in Section 3.5.1, the Navy proposes to implement a series of procedural mitigation measures designed to minimize or avoid potentially injurious impacts on marine mammals and sea turtles. The Navy implements mitigation measures during training and testing activities when a marine mammal or sea turtle is observed in the mitigation zone. The mitigation zones encompass the estimated ranges to injury for sonar sources and much of the range to injury for explosives. The Navy designed the mitigation zones for most acoustic and explosive stressors according to its source bins. Sonars and other transducers are grouped into classes that share an attribute, such as frequency range or purpose of use. Classes are further sorted by bins based on the frequency or bandwidth, source level, and when warranted, the application in which the source would be used. Explosives detonated in water are binned by net explosive weight (NEW). Mitigation does not pertain to stressors that would have no effect on an ESA-listed species (e.g., acoustic and explosive sources that do not have the potential to impact ESA-listed marine mammals or sea turtles).

NAEMO does not take into account mitigation measures or animal avoidance behavior when predicting impacts to marine mammals and sea turtles from acoustic stressors. Therefore, to account for the potential for mitigation measures to minimize potential impacts on marine mammals and sea turtles, the Navy quantifies the potential for mitigation to reduce model-estimated permanent threshold shift (PTS) to temporary threshold shift (TTS) for exposures to sonar and other transducers, and to reduce model-estimated mortality due to injury from exposures to explosives. Mitigation effectiveness is quantitatively assessed on a per-scenario basis using four factors: species sightability, observation area, visibility, and positive control of the sound source. Sightability of each species that may be present in the mitigation zone is

determined by species-specific characteristics and the viewing platform. Observation area refers to the extent to which the type of mitigation proposed for a sound producing activity (e.g., active sonar) allows for observation of the mitigation zone prior to and during the activity. Positive control of the sound source is based on the ability to shut down the source in a timely manner to mitigate impacts. Considering these factors, only a portion of injurious exposures are considered mitigable. In the quantitative analysis, consideration of mitigation measures means that, for activities where mitigation is feasible, some model-estimated PTS is considered mitigated to the level of TTS. The impact analysis does not analyze the potential for mitigation to reduce TTS or behavioral effects. In practice, mitigation also protects all unobserved (below the surface) animals in the vicinity, including other species, in addition to the observed animal. However, the analysis assumes that only animals sighted at the water surface would be protected by the applied mitigation. The analysis, therefore, does not capture the protection afforded to all marine species in the vicinity of animals sighted at the ocean surface within the mitigation zone.

The Navy estimated the ability of Navy Lookouts to observe the range to PTS for each training or testing event. The ability of Navy Lookouts to detect protected species in or approaching the mitigation zone is dependent on the animal's presence at the surface and the characteristics of the animal that influence its sightability (such as group size or surface active behavior). The behaviors and characteristics of some species may make them easier to detect. For example, based on small boat surveys between 2000 and 2012 in the Hawaiian Islands, pantropical spotted dolphins and striped dolphins were frequently observed leaping out of the water, and Cuvier's beaked whales and Blainville's beaked whales were occasionally observed breaching (Navy 2019c). These behaviors are visible from a great distance and likely increase sighting distances and detections of these species. Environmental conditions under which the training or testing activity could take place are also considered, such as the sea surface conditions, weather (e.g., fog or rain), and day versus night.

To consider the benefits of procedural mitigation to marine mammals and sea turtles within the ESA impact estimates, the Navy conservatively factors mitigation effectiveness into its quantitative analysis process. The Navy's quantitative analysis assumes Lookouts will not be 100 percent effective at detecting all individual marine mammals and sea turtles within the mitigation zones for each activity. This is due to the inherent limitations of observing marine species and because the likelihood of sighting individual animals is largely dependent on observation conditions (e.g., time of day, sea state, mitigation zone size, observation platform) and animal behavior (e.g., the amount of time an animal spends at the surface of the water). This is particularly true for sea turtles, small marine mammals, and marine mammals that display cryptic behaviors (e.g., surfacing to breathe with only a small portion of their body visible from the surface). Discussions about the likelihood that a Lookout would observe a marine mammal or sea turtle pertain specifically to animals that are available to be observed (i.e., on, above, or just below the water's surface). The benefits of procedural mitigation measures for species that were not included in the Navy's quantitative analysis process (i.e., fish) are discussed qualitatively.

The Navy's quantitative analysis accounts for and quantifies the potential for animals to actively avoid potentially injurious sound sources. Marine mammals and sea turtles often avoid loud sound sources (e.g., those that could be injurious). Because marine mammals and sea turtles are assumed to initiate avoidance behavior when exposed to relatively high received levels of sound within their capacity to detect, an exposed animal could reduce its cumulative sound energy exposure from something like a sonar event with multiple pings (i.e., accumulated sound exposures) by leaving the area. This would reduce risk of both PTS and TTS, although the quantitative analysis only considers the potential to reduce instances of PTS by accounting for marine mammals or sea turtles swimming away to avoid repeated high-level sound exposures. All reductions in PTS sonar impacts from likely avoidance behaviors are considered TTS impacts.

A more detailed description of this process is provided in the Navy's technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018d).

2.2.2 Criteria and Thresholds to Predict Impacts to Marine Mammals

The Navy's quantitative acoustic effects analysis for marine mammals relies on information about the numerical sound and energy values that are likely to elicit certain types of physiological and behavioral reactions. The following section describes the specific criteria developed and applied for each species and sound source associated with Navy training and testing activities.

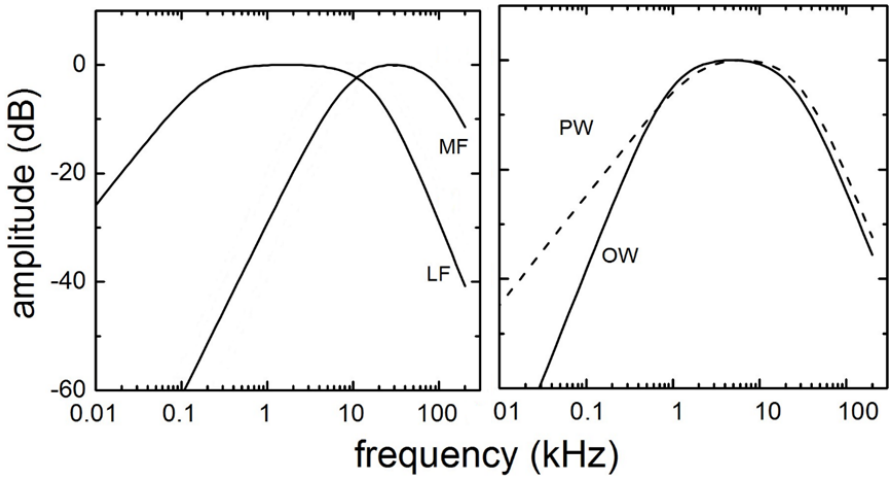
For marine mammals, the Navy, in coordination with NMFS, established acoustic thresholds (for impulsive, non-impulsive sounds, and explosives) using the best available science that identifies the received level of underwater sound above which exposed marine mammals would reasonably be expected to experience a potentially significant disruption in behavior, or to incur TTS or PTS of some degree. Thresholds have also been developed to identify the pressure levels above which animals may incur different types of tissue damage from exposure to pressure waves from explosive detonation. A detailed description of the criteria and threshold development is included in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Impact to Marine Mammals and Sea Turtles* (Navy 2017a). The thresholds used by the Navy were developed by compiling and synthesizing the best available science on the susceptibility of marine mammals to effects from acoustic exposure. NMFS has independently evaluated and adopted the Navy's marine mammal criteria and thresholds for use in this consultation as the best available science on the exposure and response of marine mammals to underwater sound produced by NWTT activities.

2.2.2.1 Marine Mammal Criteria for Hearing Impairment, Non-Auditory Injury, and Mortality

The marine mammal criteria and thresholds for non-impulsive and impulsive sources for hearing impairment, non-auditory injury, and mortality, as applicable, are described below. The Navy's

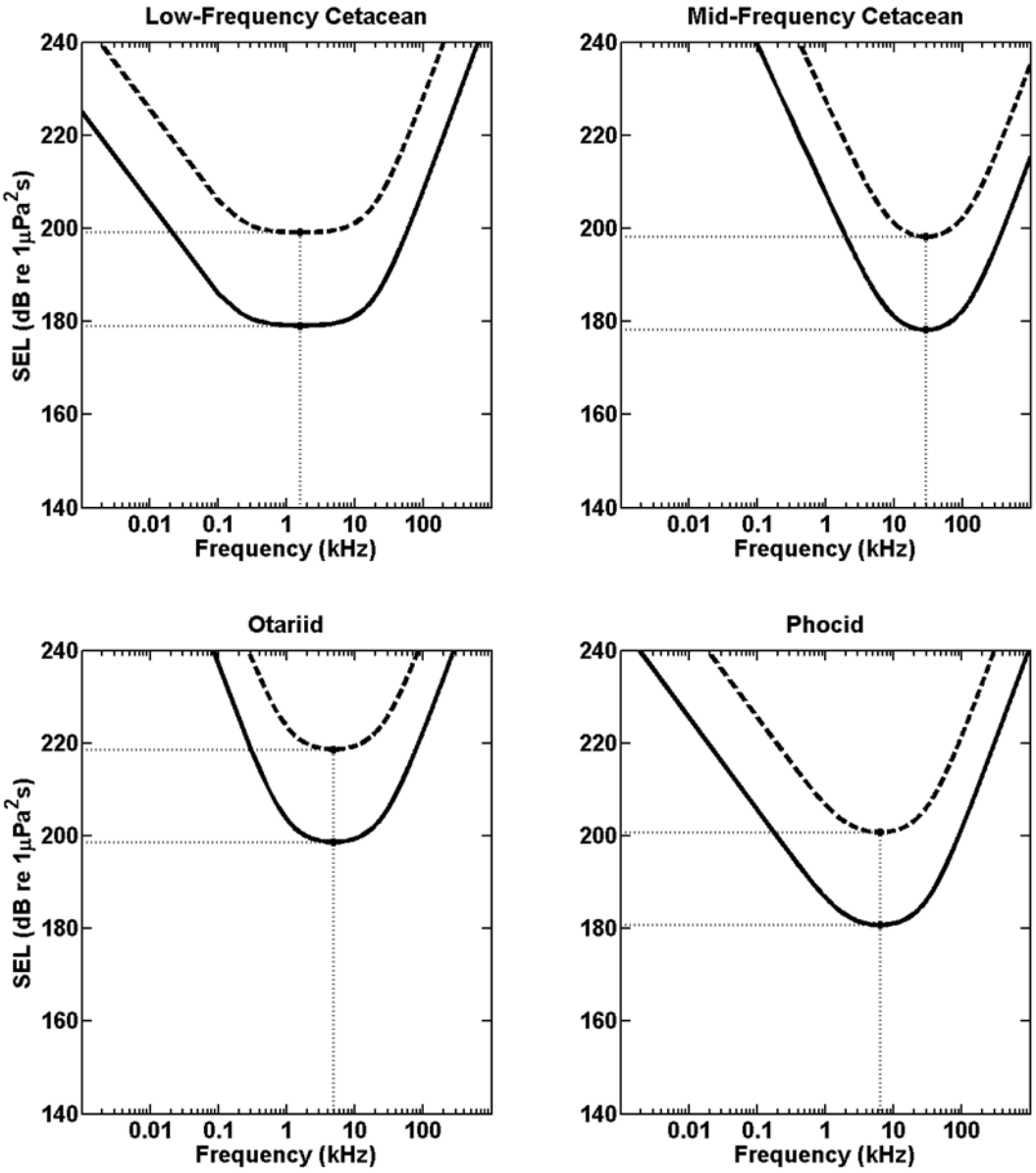
quantitative acoustic effects analysis used dual criteria to assess auditory injury (i.e., PTS) to different marine mammal groups (based on hearing sensitivity) as a result of exposure to noise from two different types of sources: impulsive (explosives) and non-impulsive (sonar and other transducers). The criteria used in the analysis are described in *NMFS' Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing* (NMFS 2018b). The Technical Guidance also identifies criteria to predict TTS.

The Navy used auditory weighting and exposure functions to assess the varying susceptibility of marine mammals to effects from noise exposure. Animals are not equally sensitive to noise at all frequencies. To capture the frequency-dependent nature of the effects of noise, auditory weighting functions were used (Figure 1). Auditory weighting functions are mathematical functions that adjust received sound levels to emphasize ranges of best hearing and de-emphasize ranges with less or no auditory sensitivity. They incorporate species-specific hearing abilities from composite audiograms to calculate a weighted received sound level in units sound pressure level (SPL) or sound exposure level (SEL). For example, the Navy used a mid-frequency cetacean composite audiogram that was consistent with recently published behavioral audiograms of killer whales (Branstetter et al. 2017b) to develop the mid-frequency auditory weighting function. The auditory weighting functions resemble an inverted "U" shape with amplitude plotted as a function of frequency. The flatter portion of the plotted function, where the amplitude is closest to zero, is the emphasized frequency range, while the frequencies below and above this range (where amplitude declines) are de-emphasized.



Note. LF = Low-Frequency Cetacean, MF = Mid-Frequency Cetacean, PW = Phocid (In-water), and OW = Otariid (In-water). ESA-listed phocids (PW) are not present in the action area. For parameters used to generate the functions and more information on weighting function derivation see Navy (2017a).

Figure 1. Navy auditory weighting functions for all marine mammal species groups.



Note: The solid curve is the exposure function for TTS onset and the large dashed curve is the exposure function for PTS onset. Small dashed lines and asterisks indicate the sound exposure level (SEL) threshold for TTS and PTS onset in the frequency range of best hearing. ESA-listed phocids are not present in the action area.

Figure 2. TTS and PTS exposure functions for sonar and other acoustic sources for marine mammals (Navy 2018b).

Table 1. Acoustic thresholds identifying the onset of temporary threshold shift (TTS) and permanent threshold shift (PTS) for non-impulsive sound sources by functional hearing group (Navy 2017).

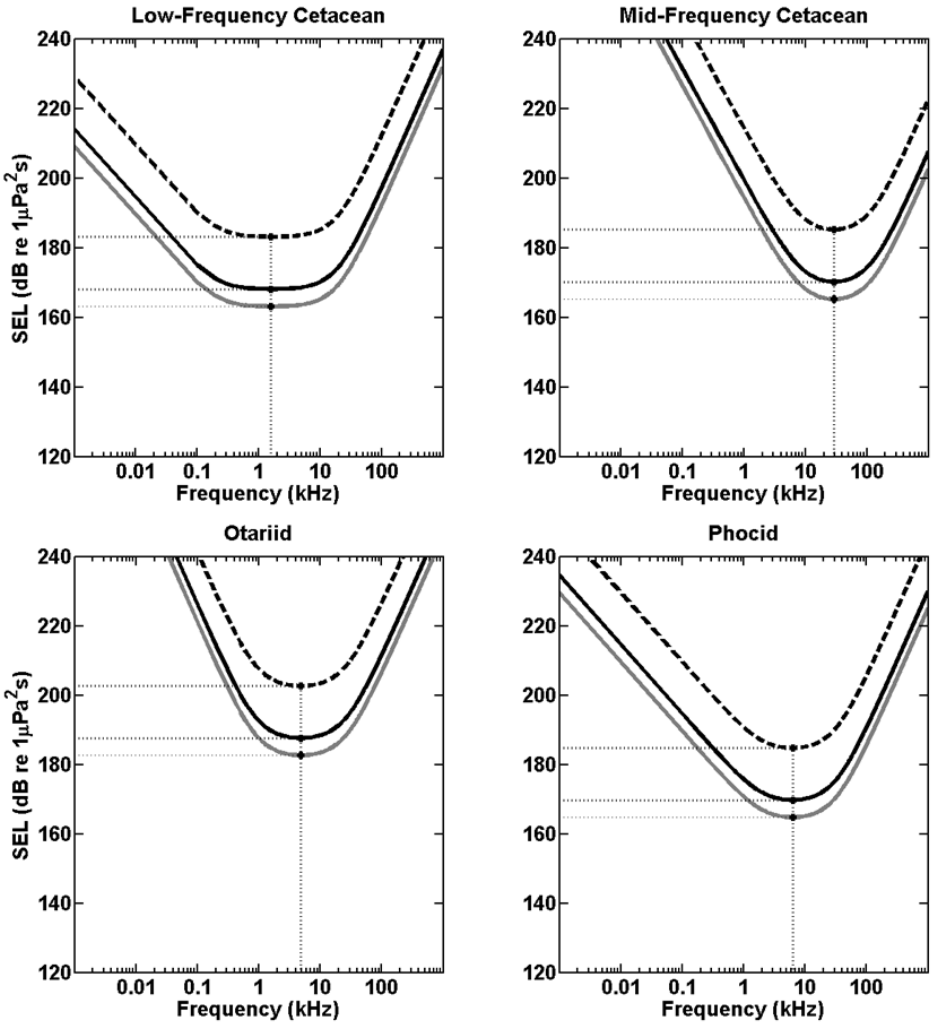
Functional Hearing Group	TTS Threshold (SEL [weighted])	PTS Threshold (SEL [weighted])
Low-Frequency Cetaceans	179	199
Mid-Frequency Cetaceans	178	198
Otariid Pinnipeds (Underwater)	199	219

Sound Exposure Level (SEL) thresholds in dB re 1 µPa2s (decibels referenced to 1 micropascal).

The TTS and PTS exposure functions for marine mammals from non-impulsive sound sources are presented in Figure 2 above. Based on the exposure functions above, the marine mammal thresholds for non-impulsive acoustic sources are summarized in Table 1 above.

For impulsive sources (inclusive of explosives, air guns, and impact pile driving), the behavioral response, TTS and PTS exposure functions for marine mammals are presented in Figure 3. Based on the exposure functions in Figure 3, the thresholds for onset of TTS and PTS for marine mammals from explosive are shown in Table 2 by functional hearing group.

In addition to TTS and PTS, Navy explosives also have the potential to result in non-auditory injury or mortality. Two metrics have been identified as predictive of injury: impulse and peak pressure. Peak pressure contributes to the "crack" or "stinging" sensation of a blast wave, compared to the "thump" associated with received impulse. Two sets of thresholds were used in the non-auditory injury assessment. The exposure thresholds were used to estimate the number of animals that may be affected during Navy training and testing activities. The thresholds for the farthest range to effects are based on the received level at which one percent risk is predicted and are useful for informing mitigation zones (see third column of Table 3). Increasing animal mass (size) and increasing animal depth both increase the impulse thresholds (i.e., decrease susceptibility), whereas smaller mass and decreased animal depth reduce the impulse thresholds (i.e., increase susceptibility). The masses used for impact assessment assume marine mammal populations are 70 percent adult and 30 percent calf/pup. The derivation of these injury criteria and the species mass estimates are provided in the Navy's technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (Navy 2017a).



Note: The dark dashed curve is the exposure function for PTS onset, the solid black curve is the exposure function for TTS onset, and the light grey curve is the exposure function for behavioral response. Small dashed lines indicate the sound exposure level (SEL) threshold for behavioral response, TTS, and PTS onset at each group's most sensitive frequency (i.e., the weighted SEL threshold). ESA-listed phocids are not present in the action area.

Figure 3. Behavioral, TTS, and PTS exposure functions for explosives (Navy 2018b).

Table 2. Onset of TTS and PTS in marine mammals for explosives by functional hearing group.

Functional Hearing Group	Species	Onset TTS	Onset PTS
Low-frequency cetaceans	All mysticetes	168 dB SEL (weighted) or 213 dB Peak SPL (unweighted)	183 dB SEL (weighted) or 219 dB Peak SPL (unweighted)
Mid-frequency cetaceans	All odontocetes	170 dB SEL (weighted) or 224 dB Peak SPL (unweighted)	185 dB SEL (weighted) or 230 dB Peak SPL (unweighted)
Otariid Pinnipeds (Underwater)	Guadalupe fur seal	188 dB SEL (weighted) or 226 dB Peak SPL (unweighted)	203 dB SEL (weighted) or 232 dB Peak SPL (unweighted)

Sound Exposure Level (SEL) thresholds in dB re 1 µPa2s (decibels referenced to 1 micropascal).

Table 3. Criteria to quantitatively assess marine mammal mortality and non-auditory injury due to underwater explosions.

Impact Category	*Impact Threshold*	*Threshold for Farthest Range to Effect*[2]
Mortality[1]	$144M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s	$103M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s
Injury[1]	$65.8M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s	$47.5M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s
	243 dB re 1 µPa SPL peak	237 dB re 1 µPa SPL peak

[1] Impulse delivered over 20 percent of the estimated lung resonance period. See U.S. Department of the Navy (2017a).
[2] Threshold for one percent risk used to assess mitigation effectiveness.
Notes: D = animal depth (m), dB re 1 µPa = decibels referenced to 1 micropascal, M = animal mass (kg), Pa-s = Pascal-second, SPL = sound pressure level

2.2.2.2 Marine Mammal Criteria for Behavioral Response

Though significantly driven by received level, the onset of behavioral disturbance from anthropogenic noise exposure is informed to varying degrees by other factors related to the source (e.g., frequency, predictability, duty cycle), the environment (e.g., bathymetry), and the receiving animal (hearing, motivation, experience, demography, behavioral context) and can be difficult to predict (Ellison et al. 2011; Southall et al. 2007a). Within the Navy's quantitative analysis, many behavioral reactions are predicted from exposure to sound that may exceed an animal's behavioral threshold momentarily but would not constitute a significant disruption of normal behavior patterns or rise to the level of ESA "take." The Navy and NMFS have used the best available science to address the challenging differentiation between significant and non-

significant behavioral reactions, but have erred on the side of caution where uncertainty exists (i.e., counting shorter duration behavioral reactions as take). This may result in some overestimation of the number of significant behavioral disruptions or behavioral harassment takes.

Sonar – Marine Mammals

For Phase III activities, the Navy coordinated with NMFS scientists to develop behavioral harassment criteria specific to the military readiness activities that utilize active sonar. The derivation of these criteria is discussed in detail in the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Impacts to Marine Mammals and Sea Turtles Technical Report* (Navy 2017a). Developing the criteria for sonar involved multiple steps. All available behavioral response studies conducted both in the field and on captive animals were examined in order to understand the breadth of behavioral responses of marine mammals to sonar and other transducers. Marine mammal species were placed into behavioral criteria groups based on their known or suspected behavioral sensitivities to sound. In most cases, these divisions were driven by taxonomic classifications (e.g., mysticetes, odontocetes). The data from the behavioral studies were analyzed by looking for significant disruptions of normal behavior patterns (e.g., breeding, feeding, sheltering), or lack thereof, for each experimental session. Due to the nature of behavioral response research to date, it is not currently possible to ascertain the types of observed reactions that would lead to an abandonment or significant alteration of a natural behavior pattern. Therefore, a methodology was developed to estimate the possible significance of behavioral reactions and impacts on normal behavior patterns.

Behavioral response severity was described herein as "low," "moderate," or "high." These are derived from the Southall et al. (2007a) severity scale. Low severity responses are those behavioral responses that fall within an animal's range of typical (baseline) behaviors and are unlikely to disrupt an individual to a point where natural behavior patterns are significantly altered or abandoned. Low severity responses include an orientation or startle response, change in respiration, change in heart rate, and change in group spacing or synchrony.

Moderate severity responses could become significant if sustained over a duration long enough that they cause variations in an animal's daily behavior outside of normal daily variations in feeding, reproduction, resting, migration/movement, or social cohesion. Based on effects analyses conducted for previous Navy consultations, many of the behavioral responses estimated using the Navy's quantitative analysis would likely be of moderate severity (defined for the purposes of this impact analysis as reaction levels four, five, and six based on the behavioral response severity scale described in Southall et al. 2007a). What constitutes a long-duration response is different for each situation and species, although it is likely also dependent upon the magnitude of the response and species characteristics such as age, body size, feeding strategy, and behavioral state at the time of the exposure. In general, a response could be considered

significant if it lasted for a few tens of minutes to a few hours, or enough time to significantly disrupt an animal's daily routine. Moderate severity responses included:

- alter migration path;
- alter locomotion (speed, heading);
- alter dive profiles;
- stop/alter nursing;
- stop/alter breeding;
- stop/alter feeding/foraging;
- stop/alter sheltering/resting;
- stop/alter vocal behavior if tied to foraging or social cohesion; and
- avoidance of area near sound source.

For the derivation of behavioral criteria, a significant duration was defined as a response that lasted for the duration of exposure or longer, regardless of how long the exposure session may have been. This assumption was made because it was not possible to tell if the behavioral responses would have continued if the exposure had continued. The costs associated with these observed behavioral reactions were not measured so it is not possible to judge whether reactions would have risen to the level of significance as defined above, although it was conservatively assumed to be the case.

Marine mammal species were placed into behavioral criteria groups based on their known or suspected behavioral sensitivities to sound (Figure 4, Figure 5, and Figure 6). These divisions are driven by taxonomic classifications (e.g., odontocetes, mysticetes, pinnipeds).

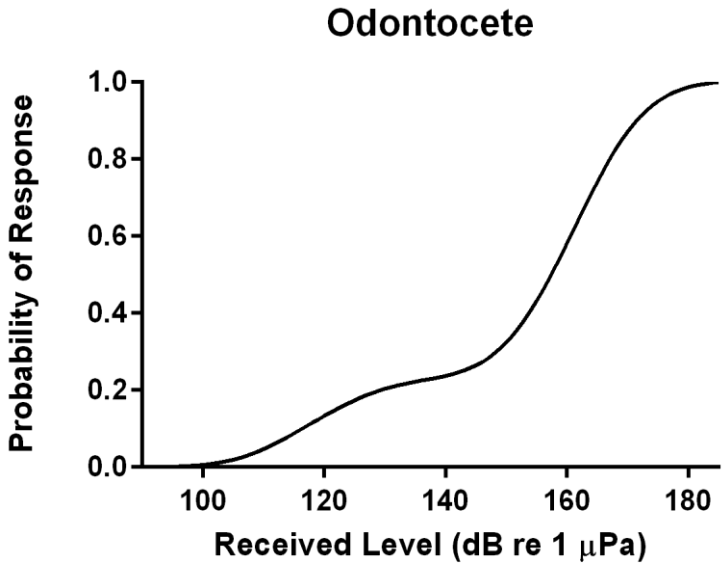


Figure 4. Behavioral response function for odontocetes showing probability of response as a function of received sound pressure level (Navy 2017a).

The analysis for active sonar used cutoff distances beyond which recent research suggests the potential for significant behavioral responses (and therefore harassment under the ESA) is considered to be unlikely (Table 4). For animals within the cutoff distance, a behavioral response function based on a received SPL was used to predict the probability of a potential significant behavioral response. For training and testing events that contain multiple platforms or tactical sonar sources that exceed 215 decibels (dB) re 1 μPa at 1 meter, this cutoff distance is substantially increased (i.e., doubled) from values derived from the literature. The use of multiple platforms (i.e., more than one vessel and/or aircraft) and intense sound sources are factors that are expected to increase responsiveness in marine mammals overall. There are currently few behavioral observations under these circumstances. For this reason, and to be conservative in the analysis of potential effects, the Navy predicted significant behavioral responses at further ranges for these more intense activities.

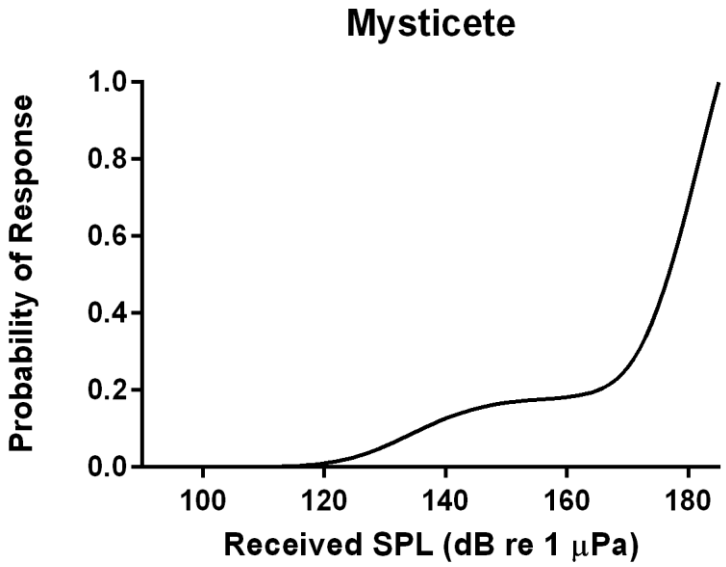


Figure 5. Behavioral response function for mysticetes (Navy 2017a).

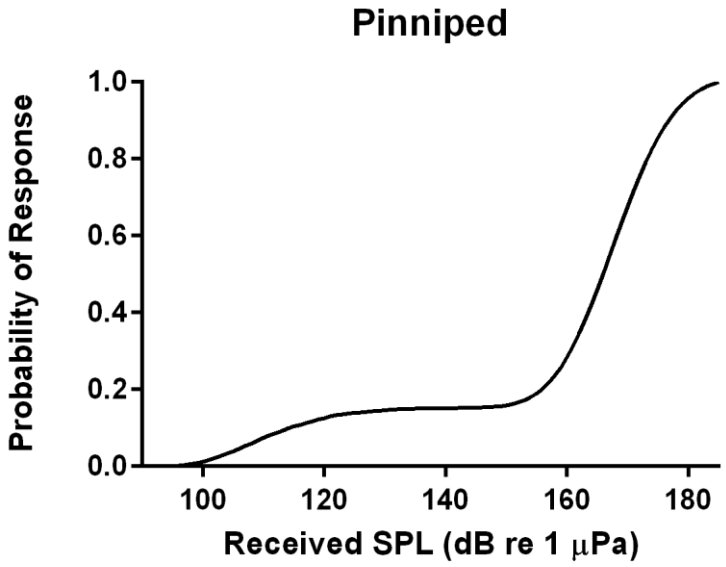


Figure 6. Behavioral response function for pinnipeds (Navy 2017a).

Table 4. Significant behavioral response cutoff distances by species group for moderate source level, single platform sonar events and for high source level[1], multi-platform sonar events (Navy 2017a).

Species Group	Moderate Source Level / Single Platform Cutoff Distance	High Source Level / Multi-Platform Cutoff Distance
Odontocetes	10 km	20 km
Mysticetes	10 km	20 km
Pinnipeds	5 km	10 km

[1] High sources levels are defined as levels at or exceeding 215 dB 1 µPa at 1 meter; km = kilometer.

Explosives Criteria – Marine Mammals

For Phase III consultations, the Navy developed explosive criteria for behavioral thresholds for marine mammals based on the hearing group's TTS threshold minus five dB (See Table 2 above for the TTS thresholds for explosives) for events that contain multiple impulses from explosives underwater (Table 5).

Table 5. Behavioral disturbance thresholds for marine mammals from underwater explosives used for the quantitative analysis (Navy 2017a).

Functional Hearing Group	Sound Exposure Level (weighted)
Low-frequency cetaceans	163
Mid-frequency cetaceans	165
Otariid pinnipeds	183

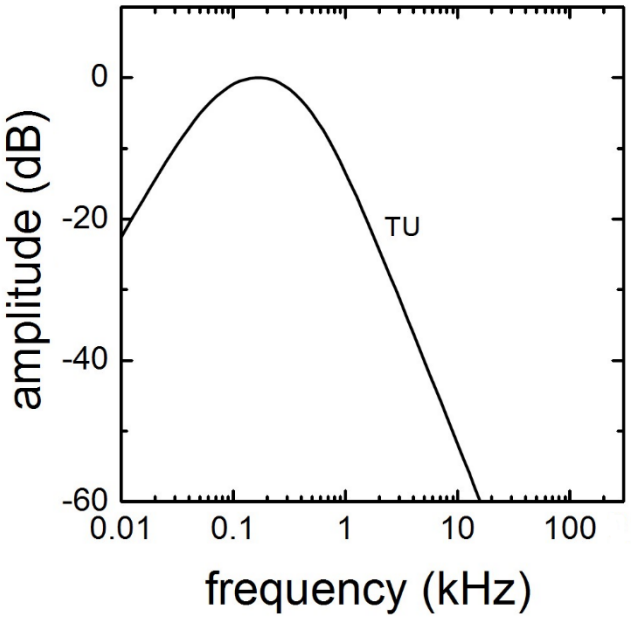
Note: Weighted SEL thresholds in dB re 1 µPa^{2}s underwater.

2.2.3 Criteria and Thresholds to Predict Impacts to Sea Turtles

The Navy's quantitative acoustic effects analysis for sea turtles relies on information about the numerical sound and energy values that are likely to elicit certain types of physiological and behavioral reactions. The following section describes the specific criteria developed and applied for sea turtles and sound sources associated with Navy training and testing activities. The thresholds used by the Navy were developed by compiling and synthesizing the best available science on the susceptibility of sea turtles to effects from acoustic exposure. The Navy provided NMFS with estimated sea turtle impacts using a behavior threshold set by NMFS based on the best available science on the exposure and response of sea turtles to underwater sound produced by NWTT activities. A more detailed description of the criteria and threshold development is included in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Impact to Marine Mammals and Sea Turtles* (Navy 2017a).

2.2.3.1 Auditory Weighting Function – Sea Turtles

In order to develop the hearing thresholds of received sound sources for sea turtles expected to produce TTS and PTS, the Navy compiled all sea turtle audiograms available in the literature to create a composite audiogram for sea turtles as a hearing group (Navy 2020e). Measured or predicted auditory threshold data, as well as measured equal latency contours, were used to influence the weighting function shape for sea turtles. For sea turtles, the weighting function parameters were adjusted to provide the best fit to the experimental data. The same methods were then applied to other species for which TTS data did not exist. However, because these data were insufficient to successfully model a composite audiogram via a fitted curve (as was done for marine mammals) median audiogram values were used in forming the sea turtle hearing group's composite audiogram (Navy 2020e). Based on this composite audiogram and data on the onset of TTS in fishes, an auditory weighting function was created to estimate the susceptibility of sea turtles to hearing loss or damage. This auditory weighting function for sea turtles is shown in Figure 7, and is described in detail in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (Navy 2017a). The frequencies around the top portion of the function, where the amplitude is closest to zero, are emphasized, while the frequencies below and above this range (where amplitude declines) are de-emphasized, when summing acoustic energy received by a sea turtle (Navy 2017a).



Notes: dB = decibels, kHz = kilohertz, TU = sea turtle species group

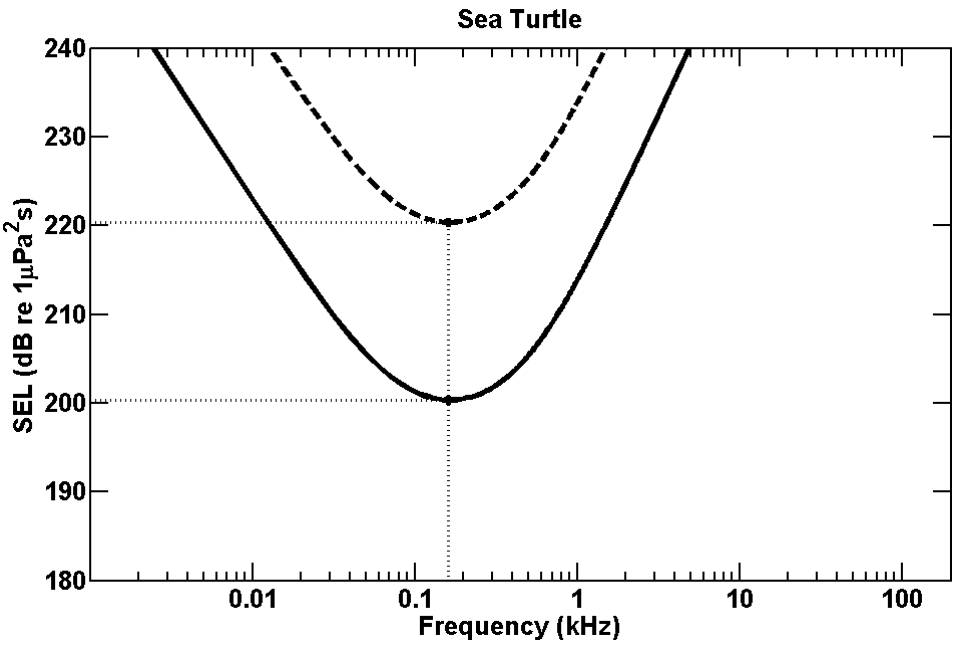
Figure 7. Auditory weighting function for sea turtles (Navy 2017).

2.2.3.2 Sonar Criteria – Sea Turtles

As mentioned above, no studies have been conducted specifically related to sea turtle hearing loss. The Navy evaluated sea turtle susceptibility to hearing loss (from sonar exposure) based upon what is known about sea turtle hearing abilities in combination with non-impulsive auditory effects data from other species such as marine mammals and fishes.

In general, sea turtles appear to be capable of detecting low-frequency sonar (less than 1000 Hz), whereas frequencies for the peak SPL for mid-frequency sonar (2000 to 8000 Hz) appear out of the range of sea turtle hearing sensitivity (Piniak 2012). However, it may be possible for sea turtles to detect high SPLs of mid-frequency sonar at increased sound pressure, but no studies have been conducted to date which expose sea turtles to these levels. Assuming a similar relationship between TTS onset and PTS onset as has been described for humans, and the available data on marine mammals, an extrapolation to PTS susceptibility of sea turtles was made based on the methods proposed by Southall et al. (2007a). Using this approach, dual metric thresholds were established for sea turtles for onset of PTS and TTS. This approach allows for the development of sea turtle exposure functions, shown below in Figure 8. These mathematical functions relate the SELs for onset of PTS or TTS to the frequency of the sonar sound. A full description of how the Navy derived these functions is provided in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (Navy 2017a). Based upon this approach, sea turtle onset of TTS would be expected to occur if received sound levels exceed 200 dB, SEL_{cum} (re: 1 μPa^2-s) and PTS would occur for sounds that exceed 220 dB SEL_{cum} (re: 1 μPa^2-s) at an exposure frequency of 200 Hz.

Because sea turtle hearing is limited to a narrow range of frequencies and thresholds for auditory impacts are relatively high, there are few sonar sources that could result in exposures exceeding the sea turtle TTS and PTS thresholds. The representative bin of LF4 (see Section 3.2) for PTS is zero meters and for TTS is up to five meters for 120 seconds of exposure. Ranges would be greater (i.e., up to tens of meters) for sonars and other transducers with higher source levels (within their hearing range); however, specific ranges could not be provided by the Navy in an unclassified document (Navy 2020e).



Note: dB re 1 μPa²s: decibels referenced to 1 micropascal second squared, kHz = kilohertz. The solid black curve is the exposure function for TTS and the dashed black curve is the exposure function for PTS onset. Small dashed lines and asterisks indicate the SEL thresholds at the most sensitive frequency for TTS (200 dB) and PTS (220 dB).

Figure 8. Sea turtle TTS and PTS exposure functions for sonar and other transducers (Navy 2017).

To date, very little research has been done on sea turtle behavioral responses relative to sonar exposure. The working group that prepared the 2014 American National Standards Institute (ANSI) Guidelines (Popper et al. 2014a) provided descriptors of sea turtle behavioral responses to sonar and other transducers. The working group estimated that the risk of a sea turtle responding to a low-frequency sonar (less than 1 kilohertz (kHz)) is low regardless of proximity to the source, and that there is no risk of a sea turtle responding to a mid-frequency sonar (1 to 10 kHz). For this analysis, similar to impulsive sounds, NMFS requested that the Navy estimate the number of sea turtles that could be exposed to sonar within their hearing range at received levels of 175 dB re: 1 μPa SPL (rms) or greater. This level is based upon work by Mccauley et al. (2000a), described for air guns. Sound levels that exceed this could cause sea turtles to exhibit a significant behavioral response such as erratic and increased swimming rates and avoidance of the sound source. Because data on sea turtle behavioral responses to non-impulsive sounds (such as sonars) is limited, the air gun data set is used to inform potential risk. We recognize this is a conservative approach, and that the relative risk of a sea turtle responding to air guns would likely be higher than the risk of responding to sonar; so it is likely that potential sea turtle behavioral responses to sonar exposures are a sub-set of sea turtles exposed to received levels of 175 dB rms (re: 1 μPa) or greater.

2.2.3.3 Explosives Criteria – Sea Turtles

NMFS and the Navy apply a peak pressure metric criterion to assess the potential onset of sea turtle physical injury and hearing impairment from explosives. Similar to other marine species, the sound pressure or blast wave produced from a detonation does not only affect hearing, but may also induce other physical injuries such as external damage to the carapace, and internally to organs and blood vessels. For sea turtles, the Navy developed criteria to determine the potential onset of hearing loss, physical injury (non-auditory) and non-injurious behavioral response to detonation exposure using the weighting function described above, as well as the impulsive sound threshold criteria recommended by the 2014 ANSI Guidelines (Popper et al. 2014a). The derivation of these injury criteria (and the species mass estimates) are described in the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* technical report (Navy 2017a).

The peak pressure metric criterion thresholds for non-auditory injury for sea turtles and range to farthest effects are shown in Table 6. These thresholds include the farthest range to effect, based on the received level at which a one percent risk is predicted and are useful for assessing the effectiveness of mitigation measures (described in greater detail later). In order to evaluate the degree to which a sea turtle may be susceptible to injury from the blast energy of an explosive detonation, both the size of the sea turtle as well as depth of the animal in the water column at exposure must be considered. This is because a larger sea turtle located deeper in the water column is assumed to be less susceptible to impacts than a smaller sea turtle, located closer to the surface in the water column. In addition, the Navy divided the percentage of the sea turtle populations according to age classes that are most likely to comprise the populations present in the action area for their impact assessment. The Navy assumed five percent of the population would be adult, and the remaining 95 percent of individuals to be sub-adult. This ratio is estimated from what is currently known about the population age structure for sea turtles based upon egg clutch size, early juvenile survival rates, and survival rates for sub-adult and adult turtles. In general, sea turtles typically lay multiple clutches of 100 or more eggs, have low juvenile survival rates, but those that make it past early life stages have increased survival at later life stages.

No studies of hearing loss from explosives have been conducted on sea turtles. Therefore, sea turtle susceptibility to hearing loss due to an acoustic exposure is evaluated using knowledge about sea turtle hearing abilities in combination with non-impulsive auditory effect data from other species (marine mammals and fish). This yields sea turtle exposure functions, shown in Table 6, which are mathematical functions that relate the SELs for onset of TTS or PTS to the frequency of the sonar sound exposure. The derivation of the sea turtle exposure functions are provided in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (Navy 2017a).

Table 6. Criteria to quantitatively assess sea turtle non-auditory injury due to underwater explosions.

Impact Category	*Impact Threshold*	*Threshold for Farthest Range to Effect*[2]
Mortality[1]	$144M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s	$103M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s
Injury[1]	$65.8M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s	$47.5M^{1/3}[1 + (D/10.1)]^{1/6}$ Pa-s
	243 dB re 1 µPa SPL peak	237 dB re 1 µPa SPL peak

[1] Impulse delivered over 20 percent of the estimated lung resonance period. See U.S. Department of the Navy (2017a).
[2] Threshold for one percent risk used to assess mitigation effectiveness.
Notes: D = animal depth (m), dB re 1 µPa = decibels referenced to 1 micropascal, M = animal mass (kg), Pa-s = Pascal-second, SPL = sound pressure level

For impulsive sounds, hearing loss in other species has been observed to be related to the unweighted peak pressure of a received sound. Because this data does not exist for sea turtles, unweighted peak pressure thresholds for TTS and PTS were developed by applying relationships observed between impulsive peak pressure TTS thresholds and auditory sensitivity in marine mammals to sea turtles. This results in dual-metric hearing loss criteria for sea turtles for impulsive sound exposure: the SEL-based exposure functions in Figure 9 and the peak pressure thresholds in Table 7. The derivation of the sea turtle impulsive peak pressure TTS and PTS thresholds are provided in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy 2017a).

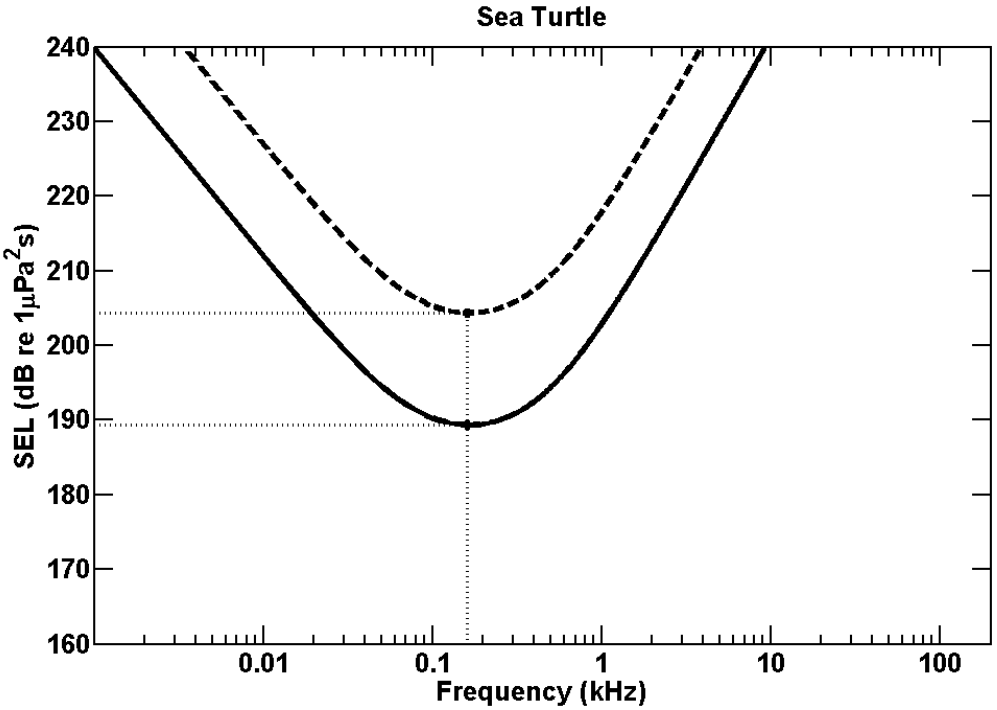


Figure 9. TTS and PTS exposure functions for impulsive sounds - sea turtles.

Notes: kHz = kilohertz, SEL = Sound Exposure Level, dB re 1 µPa^{2}s = decibels referenced to 1 micropascal squared second. The solid black curve is the exposure function for TTS onset and the dashed black curve is the exposure function for PTS onset. Small dashed lines and asterisks indicate the SEL thresholds and most sensitive frequency for TTS and PTS.

Table 7. TTS and PTS peak pressure thresholds derived for sea turtles exposed to impulsive sounds

Auditory Effect	*Unweighted Peak Pressure Threshold*
TTS	226 dB re 1 µPa SPL peak
PTS	232 dB re 1 µPa SPL peak

Notes: dB re 1 µPa = decibels referenced to 1 micropascal, PTS = permanent threshold shift, SPL = sound pressure level, TTS = temporary threshold shift

For behavioral response assessment, NMFS requested that the Navy estimate the number of sea turtles that could be exposed to explosions at received levels of 175 dB rms (re 1 µPa) or greater. This is the level at which Mccauley et al. (2000a) determined sea turtles would begin to exhibit avoidance behavior after multiple firings of nearby or approaching air guns.

2.3 Criteria and Thresholds to Predict Impacts to Fishes

A description of fish hearing is provided in the Effects of the Action section below (see Section 8.1.3.1). For many of the acoustic stressors affecting fishes in the action area during NWTT activities the Navy relied primarily on the recommendations in the 2014 ANSI Guidelines. The

Navy worked with NMFS to develop thresholds or use thresholds developed by others, based on what NMFS considers to be the best available scientific information on the effects of anthropogenic sounds on fishes. None of the studies researching fish hearing have documented PTS impairment from various sound sources. This is attributed to the ability of fish to regenerate inner ear hair cells. Marine mammals and sea turtles are not known to have this ability. Hearing loss in fish is considered recoverable, although the rate of recovery is based upon the degree of the TTS sustained. Thus, auditory impairment in fish is considered recoverable over some duration; and auditory impairment thresholds are based solely on the onset of TTS for fish.

For barotrauma (e.g., physical injuries and mortality) in fish, NMFS and the Navy apply a peak pressure metric criteria. For hearing impairment (i.e., TTS), NMFS and the Navy apply an SEL_{cum} threshold. NMFS also applies an rms threshold for some acoustics sources to assess whether behavioral responses may be elicited during some sound exposures.

2.3.1 Sonar – Fishes

All ESA-listed fishes in the action area have the potential to be exposed to sound from sonar and other transducers during NWTT activities. Direct injury from these sound sources is considered highly unlikely because injury from sound levels produced from sonar has not been documented in fishes (Halvorsen et al. 2012; Kane et al. 2010; Popper et al. 2014a; Popper et al. 2007b; Popper et al. 2013). The sound characteristics (e.g., non-impulsive) of sonar are considered to pose less risk to fish because they have low peak pressures and slow rise times. These non-impulsive sound sources lack the strong shock wave such as that produced from an explosion. The most probable impacts from exposure to sonar and other transducers would be in the form of TTS and would likely occur after a long duration of exposure at low frequencies, longer than most of the sonar exposures created by Navy training and testing activities.

To evaluate the effects of sonar use during Navy activities, NMFS and the Navy use the criteria for sonar and fishes based upon the recommendations provided in the 2014 ANSI Guidelines (Popper et al. 2014a). Only fishes with a swim bladder are likely to develop TTS from low-frequency active sonar exposure. The recommended threshold for onset of TTS in this fish hearing group would be low frequency sonar exposure levels greater than 210 dB SEL_{cum} (re 1 μPa^2-s) (Table 8). TTS has not been observed from exposure to mid-frequency active sonar in fish species with a swim bladder that is not involved in hearing. Fishes within this hearing group, which include all ESA-listed fish considered in this consultation (except Pacific eulachon which lack a swim bladder), do not sense pressure well and typically cannot hear at frequencies above 2 kHz (Halvorsen et al. 2012; Popper et al. 2014a). Therefore, no criteria were proposed for fishes with a swim bladder that is not involved in hearing from exposure to mid-frequency active sonar. Fishes without a swim bladder (e.g., Pacific eulachon) are even less susceptible to noise exposure, therefore TTS is also unlikely to occur, and no criteria are proposed for either mid or low-frequency sonar.

Table 8. Fish sound exposure criteria for TTS from sonar (Navy 2017).

Fish Hearing Group	TTS from Low-Frequency Sonar (SEL_{cum})	TTS from Mid-Frequency Sonar (SEL_{cum})
Fishes without a swim bladder	NC	NC
Fishes with a swim bladder not involved in hearing	> 210	NC

Notes: SEL_{cum} = Cumulative sound exposure level (decibel referenced to 1 micropascal squared seconds [dB re 1 μPa^2-s]), NC = effects from exposure to sonar is considered to be unlikely, therefore no criteria are reported, > indicates that the given effect would occur above the reported threshold.

2.3.2 Explosives – Fishes

For effects of explosives on fish, we used the mortality criteria provided in the 2014 ANSI Guidelines (Popper et al. 2014a), which also divides fish according to presence/absence of a swim bladder and if the swim bladder is involved in hearing (described above). The 2014 ANSI Guidelines do not suggest numeric thresholds for injury or TTS due to explosives (only mortality). Therefore, we used the impact pile driving and air gun injury thresholds suggested by the ANSI Guidelines as surrogates for explosives. These criteria are used for this consultation as numeric thresholds for injury and TTS in fishes with swim bladders. We conservatively assume that the zone of impact would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. For fish with a swim bladder, the onset of the lowest level of injury along the injury continuum in this case would be either greater than 207 dB peak re 1 µPa for injury, or greater than 186 dB SEL_{cum} dB re 1 μPa^2-s for TTS as indicated in Table 9.

Table 9. Sound exposure criteria used for mortality and injury in fishes.

Fish Hearing Group	Onset of Mortality	Onset of Injury	TTS
	SPL_{peak}	SPL_{peak}	(SEL_{cum})
Fishes without a swim bladder	229	220	NC
Fishes with a swim bladder not involved in hearing	229	> 207	> 186

Notes: SEL_{cum} = Cumulative sound exposure level (decibel referenced to 1 micropascal squared seconds [dB re 1 μPa^2-s]), SPL_{peak} = Peak sound pressure level (decibel referenced to 1 micropascal [dB re 1 µPa]), > indicates that the given effect would occur above the reported threshold. Notes: TTS = Temporary Threshold Shift. NC = no criteria, > indicates that the given effect would occur above the reported threshold.

While NMFS uses acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder, the Navy developed its ranges to effects for explosives on fish using an injury threshold of 220 dB peak re 1 µPa. This threshold is based on a compilation of data from a variety of studies on the effects of explosives on fishes with swim bladders (Gaspin 1975a; Gaspin et al. 1976; Hubbs and Rechnitzer 1952b; Settle et al. 2002; Yelverton et al. 1975) and is described in

further detail in the Navy's BA (Navy 2020e). Studies have shown that fish without swim bladders are much less susceptible to injury from explosions than fish with swim bladders (Popper et al. 2014c; Yelverton et al. 1975). Southern DPS eulachon is the only ESA-listed fish species considered in this opinion that does not have a swim bladder Therefore, for eulachon we used the Navy's proposed criteria (220/229 dB peak SPL criteria for injury and mortality) for fish without swim bladders as it is likely conservative for this species.

As discussed in Section 8.2.3.1, the Navy used its exposure criteria (220/229 dB peak SPL criteria for injury and mortality) in addition to NMFS' proposed injury exposure criteria for fishes with swim bladders (>207 dB peak re 1 µPa) to develop ranges to effects for mortality and injury. The Navy provided range to effects for each of the above thresholds, which we used to calculate estimated numbers of mortality and injury for ESA-listed fishes that are likely to be adversely affected by the Navy's proposed NWTT activities.

In addition to sound pressure levels, we also considered effects from particle motion of fish. Fishes with swim bladders within the action area such as salmonids have a swim bladder that is distant from the ear and does not contribute to sound pressure reception. These fishes are primarily particle motion detectors. Particle motion is the back and forth motion of the component particles of the medium, measured as the particle displacement, velocity, or acceleration. While it is clear that the use of particle motion for establishing criteria is something that should be done in the future, the lack of data on how particle motion impacts fishes as well as the lack of easily used methods to measure particle motion currently precludes the evaluation of particle motion in our acoustic effects analysis (Hawkins et al. 2020).

2.4 Species Density Estimates

A quantitative effects analysis requires information on the abundance and density of ESA-listed species in the potentially impacted area. In this section we provide the species density estimates that were used for the quantitative effects analyses in Section 8.2. For marine mammals and sea turtles, density estimates were taken directly from the *U.S. Navy Marine Species Density Database Phase III for the Northwest Training and Testing Study Area* (Navy 2019f). We derived fish density estimates (Section 2.4.2 below) within the action area based on the methodology developed for the 2015 NWTT Phase II biological opinion (NMFS 2015c), and updated as needed to reflect new information since the prior consultation.

2.4.1 Marine Mammal and Sea Turtle Density Estimates

To characterize marine mammal and sea turtle densities in the NWTT Study Area, the Navy compiled data from multiple sources and developed a protocol to select the best available density estimates based on species, area, and season. When multiple data sources were available, the Navy ranked density estimates based on a hierarchal approach to ensure that the most accurate estimates were selected (Navy 2020e). The highest tier included peer-reviewed published studies of density estimates from spatial models, since these provide spatially explicit density estimates with relatively low uncertainty. Other preferred sources included peer-reviewed published

studies of density estimates derived from systematic line-transect survey data, the method typically used for NMFS marine mammal stock assessment reports (SARs). In the absence of survey data, information on species occurrence and known or inferred habitat associations have been used to predict densities using model-based approaches including Relative Environmental Suitability models (Navy 2020e). Because these estimates inherently include a high degree of uncertainty, they were considered the least preferred data source. In cases where a preferred data source was not available, density estimates were selected based on expert opinion from scientists. The resulting Geographic Information System database includes seasonal density values for every marine mammal and sea turtle species present within the action area, and density data are provided as a geographic grid of typically ten kilometers by ten kilometers. This database is described in the technical report titled *U.S. Navy Marine Species Density Database Phase III for the Northwest Training and Testing Study Area*, hereafter referred to as the Density Technical Report (Navy 2019f). These data were used as an input into the NAEMO. Marine mammal and sea turtle density estimates that were used in NAEMO modeling for acoustic effects and our risk analyses on the effects of various stressors from Navy training and testing activities are summarized in Table 10 below. For several species, densities were based on a spatial model resulting in numerous density values throughout different portions of the action area. For these species, modeled densities are displayed spatially in Figure 10 through Figure 23. For Southern Resident killer whales, two separate spatial models were developed for offshore densities to account for differences in the offshore distribution and movement of whales from different groups or 'pods' (i.e., one for K and L pods combined and one for J pod).

Table 10. Summary of marine mammal and sea turtle density values used for quantitative acoustic effects analysis (Navy 2019f)

Species	Location	Spring	Summer	Fall	Winter
Blue whale (Figure 10 and Figure 11)	Offshore	S	S	S	S
	Inland	0	0	0	0
	Western Behm Canal	0	0	0	0
Fin whale (Figure 12 and Figure 13)	Offshore	S	S	S	S
	Inland	0	0	0	0
	Western Behm Canal	0.0001	0.0001	0.0001	0.0001
Humpback whale – Central America and Mexico DPSs combined (Figure 14 and Figure 15)	Offshore	S	S	S	S
	Inland	0.0005 - 0.00058	0.00074 - 0.0027	0.00074 - 0.0027	0.0005 - 0.00058
	Western Behm Canal	0.0180	0.0117	0.0081	0.0081
Sei whale	Offshore	0.00032 - 0.00040	0.00032 - 0.00040	0.00032 - 0.00040	0.00032 - 0.00040
	Inland	0	0	0	0
	Western Behm Canal	0	0	0	0
Sperm whale (Figure 16)	Offshore	S	S	S	S
	Inland	0	0	0	0
	Western Behm Canal	0	0	0	0
Southern Resident killer whale (Figures 16-22)	Offshore	S	S	S	S
	Inland	S	S	S	S
	Western Behm Canal	0	0	0	0
Guadalupe fur seal	Offshore 10 – 200 m	0.0070	0.0153	0.0153	0.0070
	Offshore 200 m – 300 km	0.0078	0.0171	0.0171	0.0078
	Offshore 300 – 700 km	0.0013	0.0029	0.0029	0.0013
	Inland	0	0	0	0
	Western Behm Canal	0	0	0	0
Leatherback sea turtle	Offshore	0.000114	0.000114	0.000114	0.000114
	Inland	0	0	0	0
	Western Behm Canal	0	0	0	0

Notes: Units for numerical values are animals/km^2. 0 = species is not expected to be present. S = spatial model with various density values throughout the range.

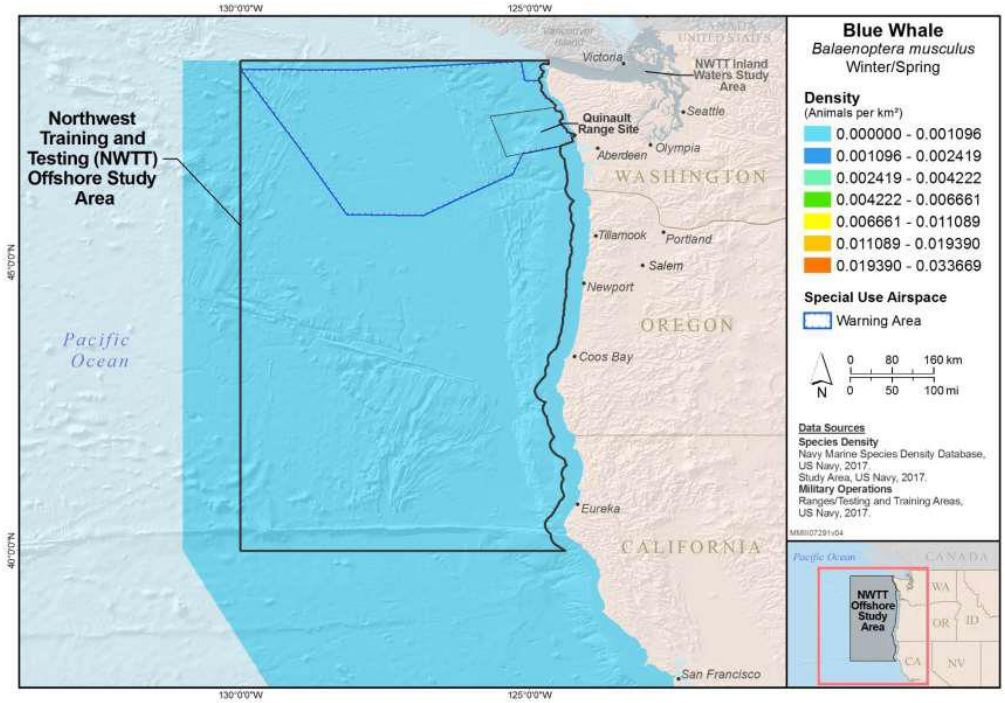


Figure 10. Offshore winter/spring distribution of blue whale (Navy 2019f).

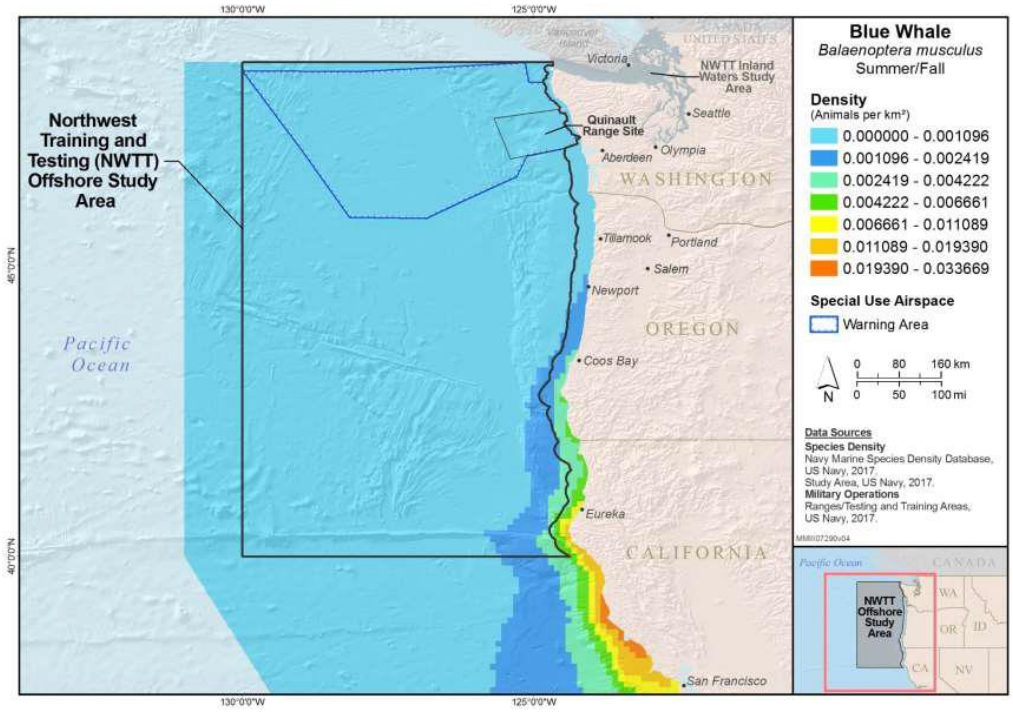


Figure 11. Offshore summer/fall distribution of blue whale (Navy 2019f).

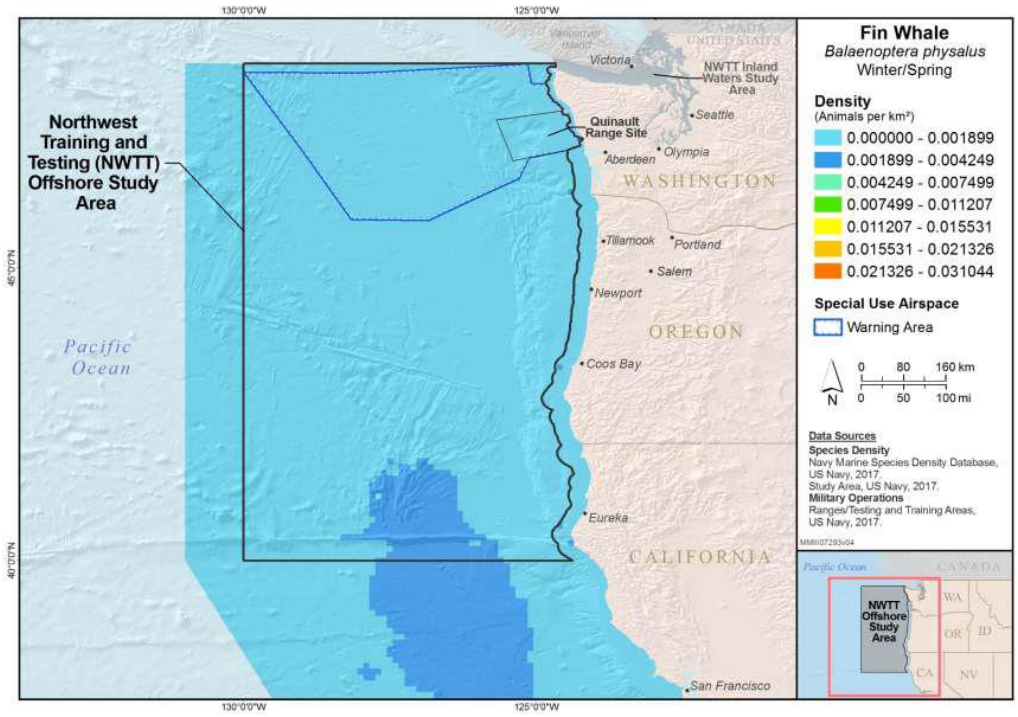


Figure 12. Offshore winter/spring distribution of fin whale (Navy 2019f) .

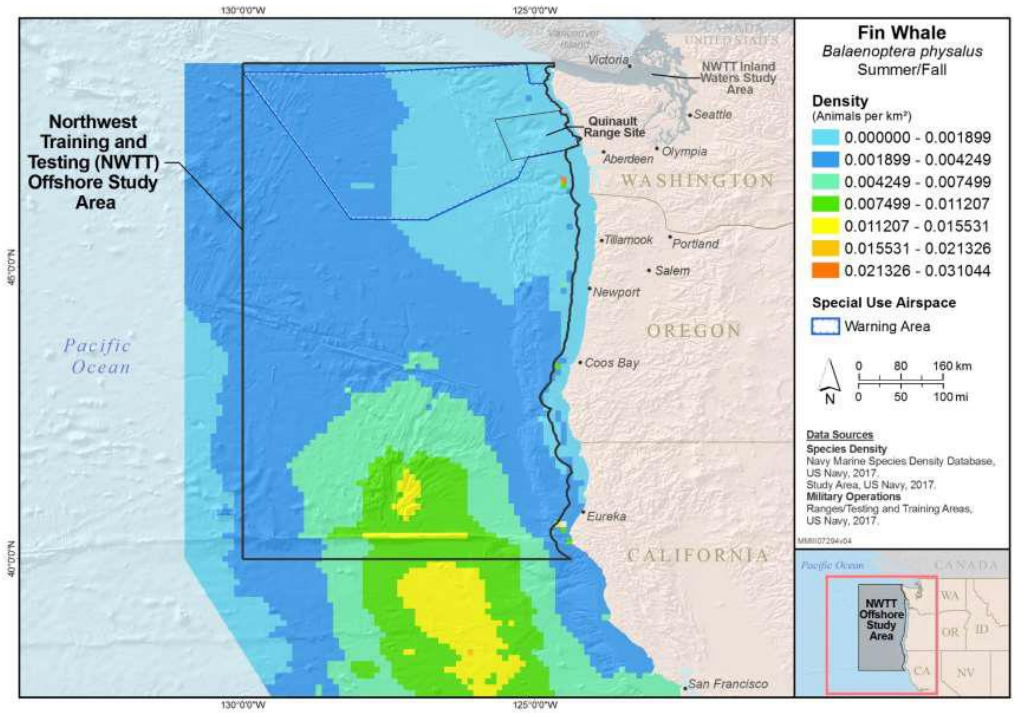


Figure 13. Offshore summer/fall distribution of fin whale (Navy 2019f).

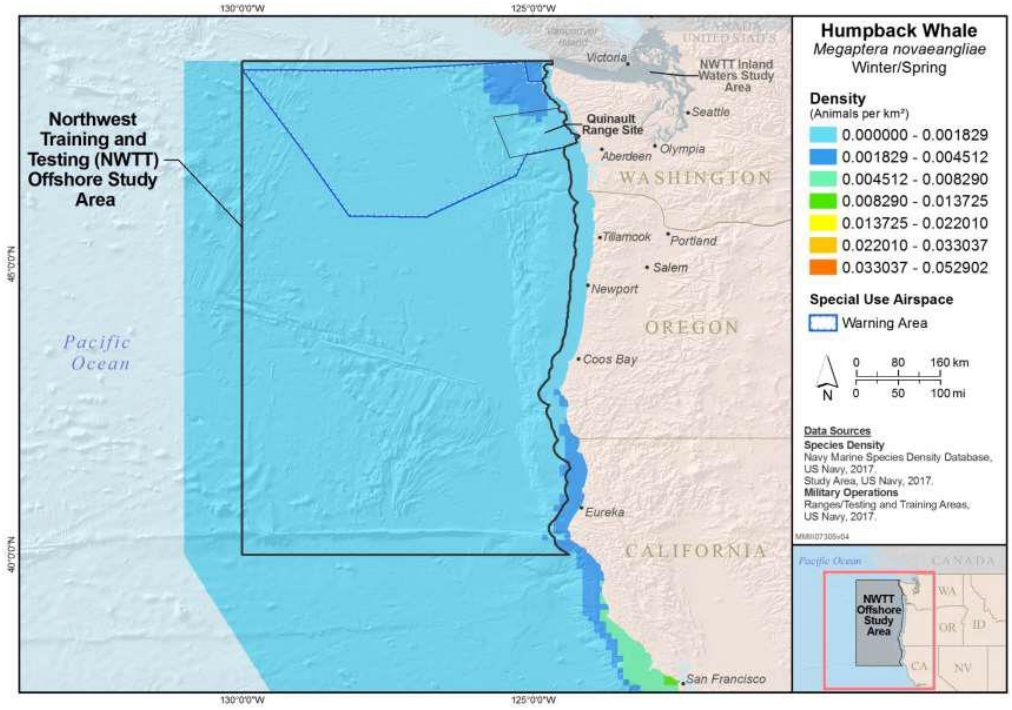


Figure 14. Offshore winter/spring distribution of humpback whale (Central American and Mexico DPSs combined) (Navy 2019f).

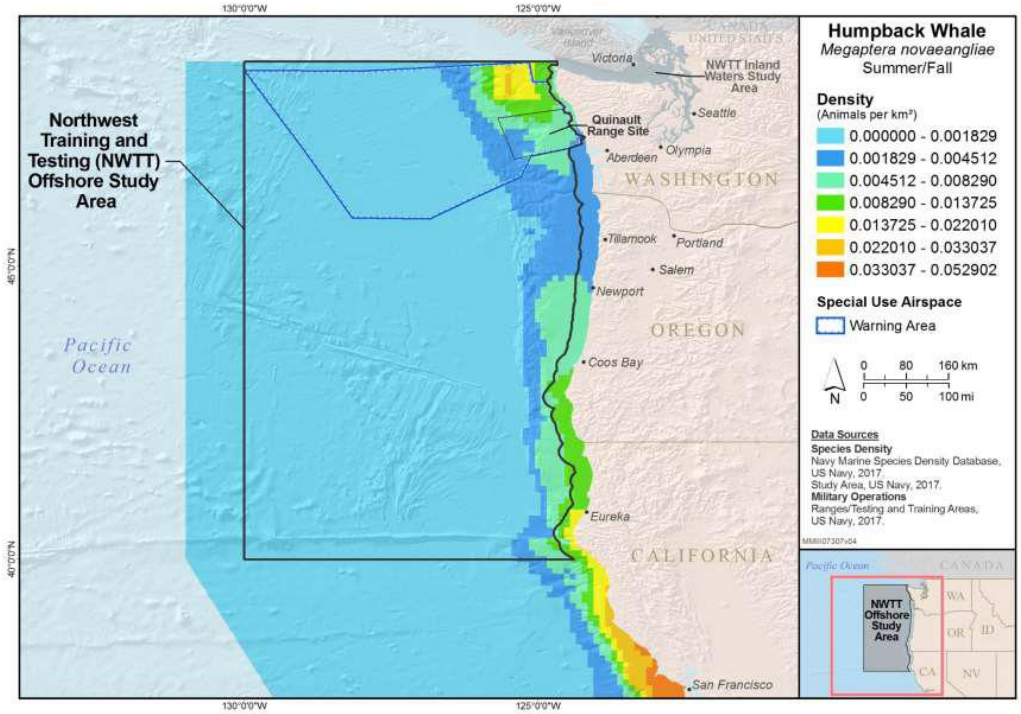


Figure 15. Offshore summer/fall distribution of humpback whale (Central American and Mexico DPSs combined) (Navy 2019f).

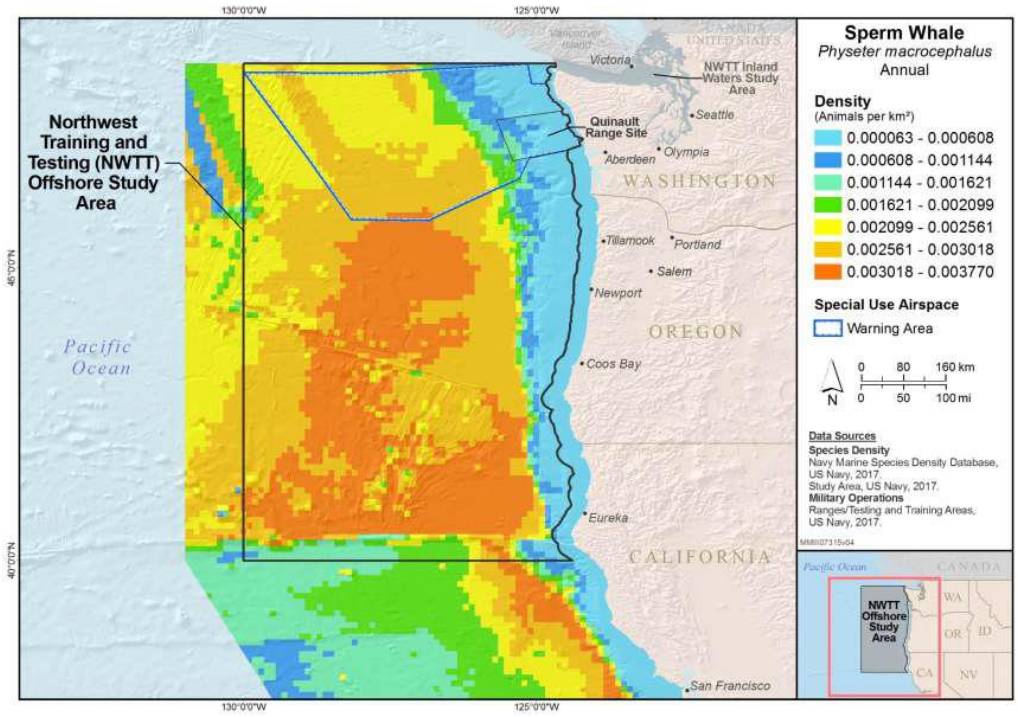


Figure 16. Offshore annual distribution of sperm whale (Navy 2019f).

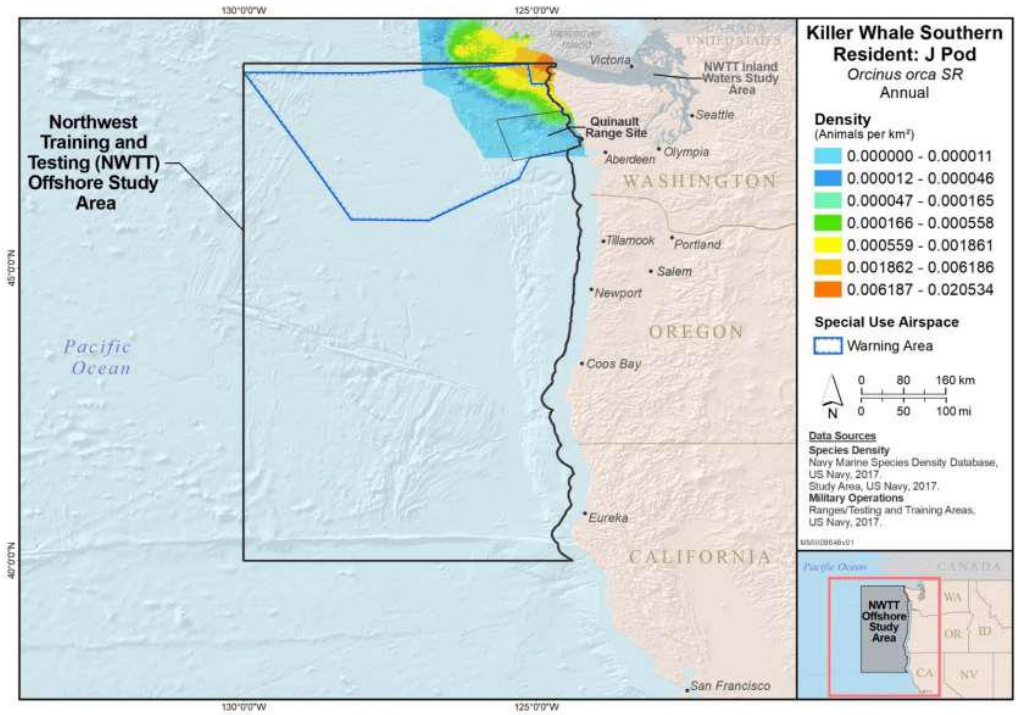


Figure 17. Offshore annual distribution of Southern Resident killer whale (J Pod) (Navy 2019f).

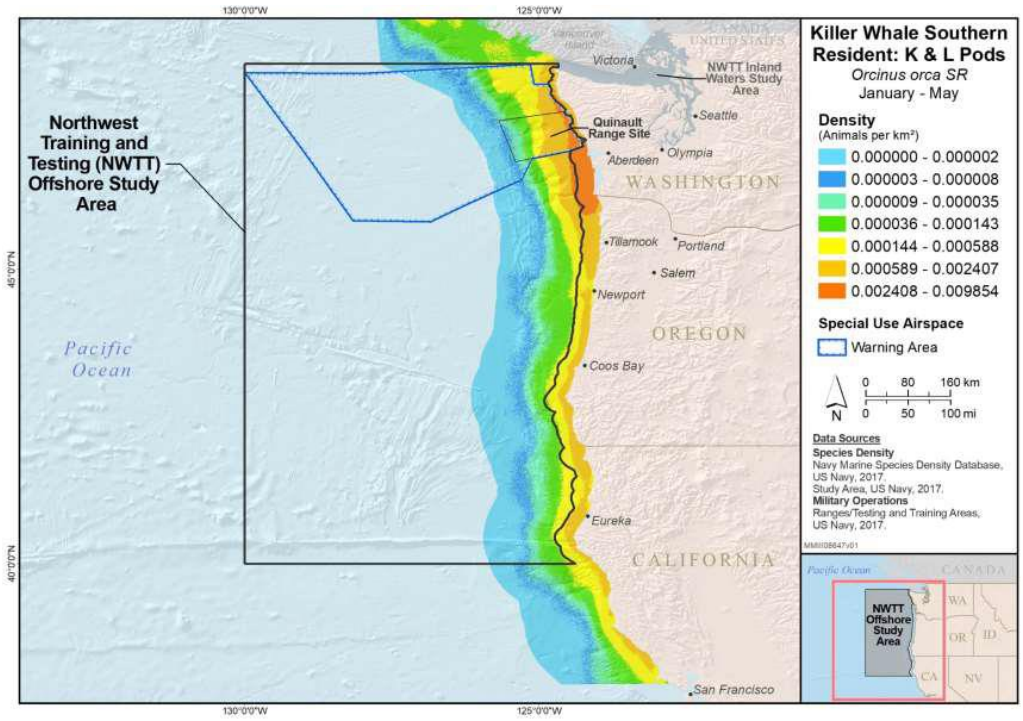


Figure 18. Offshore January–May distribution of Southern Resident killer whale (K & L pods) (Navy 2019f).

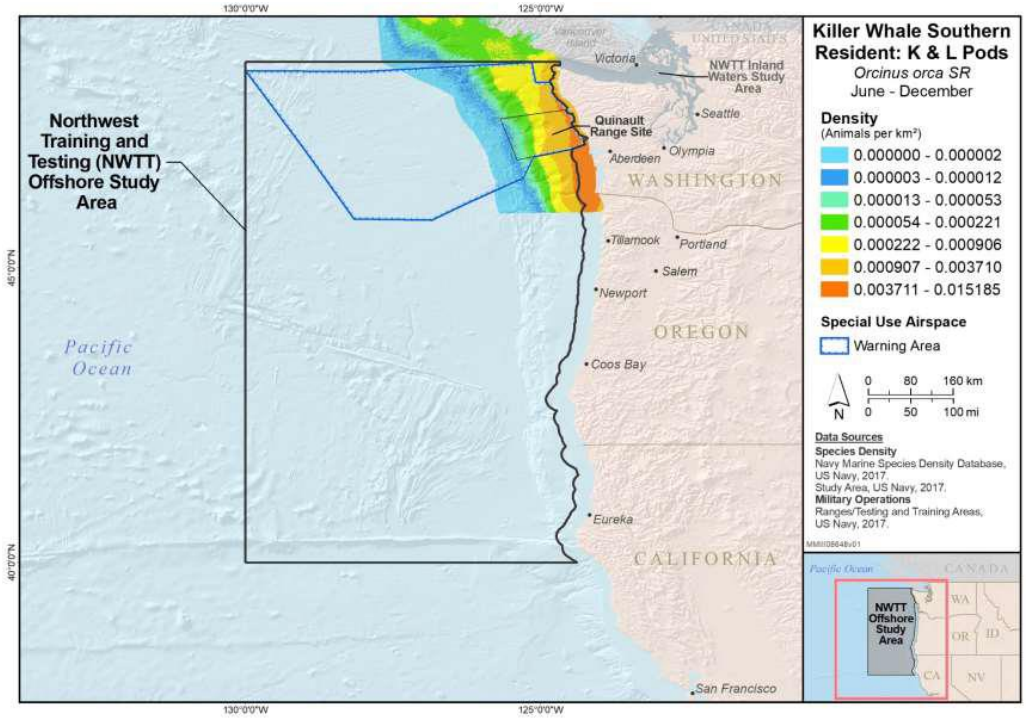


Figure 19. Offshore June–December distribution of Southern Resident killer whale (K & L pods) (Navy 2019f).

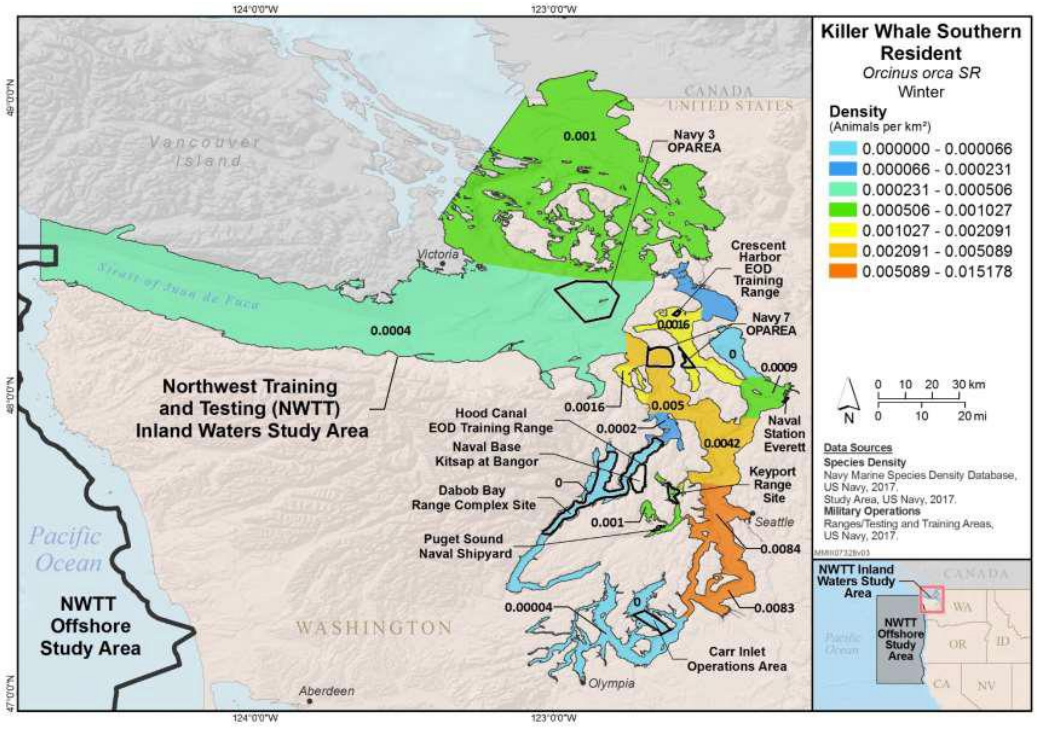


Figure 20. Inland waters winter distribution of Southern Resident killer whale (Navy 2019f).

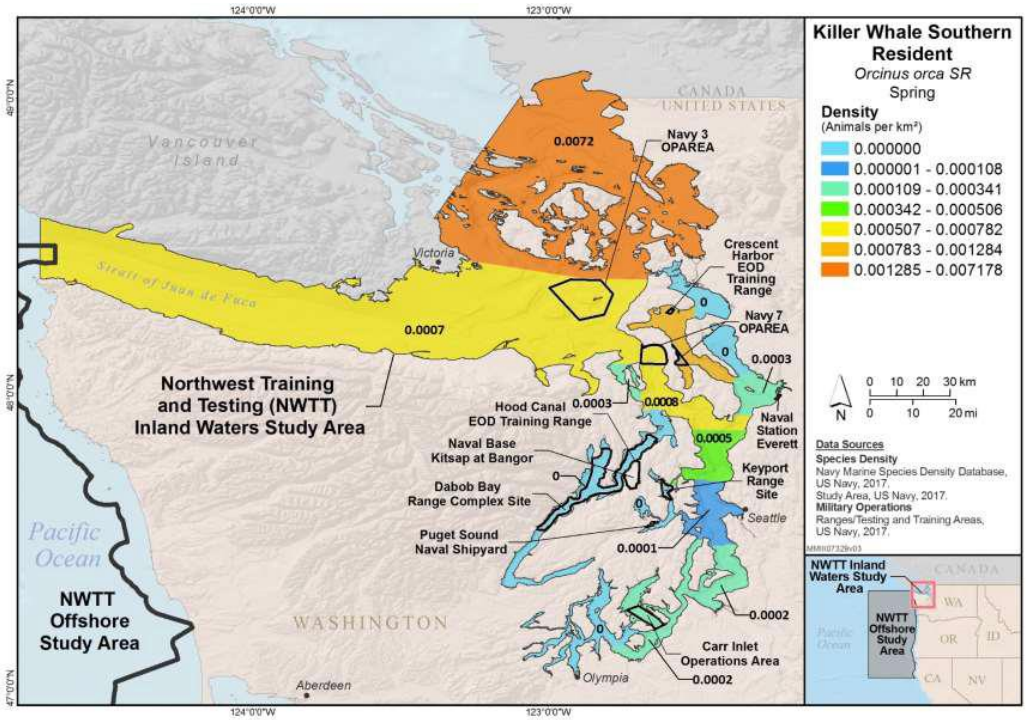


Figure 21. Inland waters spring distribution of Southern Resident killer whale (Navy 2019f).

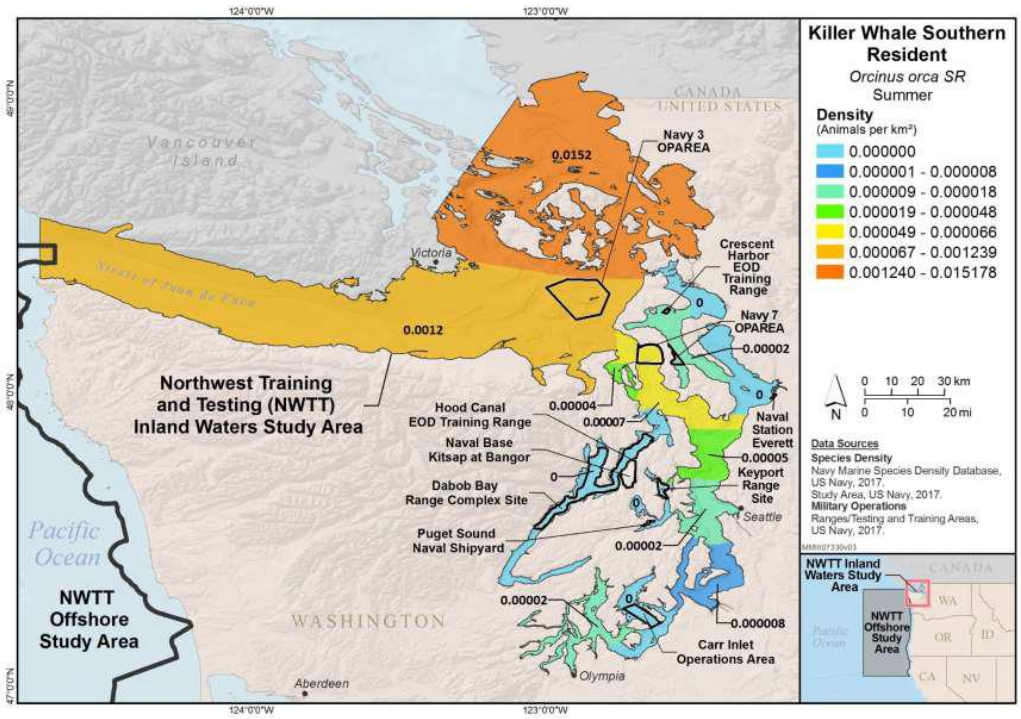


Figure 22. Inland waters summer distribution of Southern Resident killer whale (Navy 2019f).

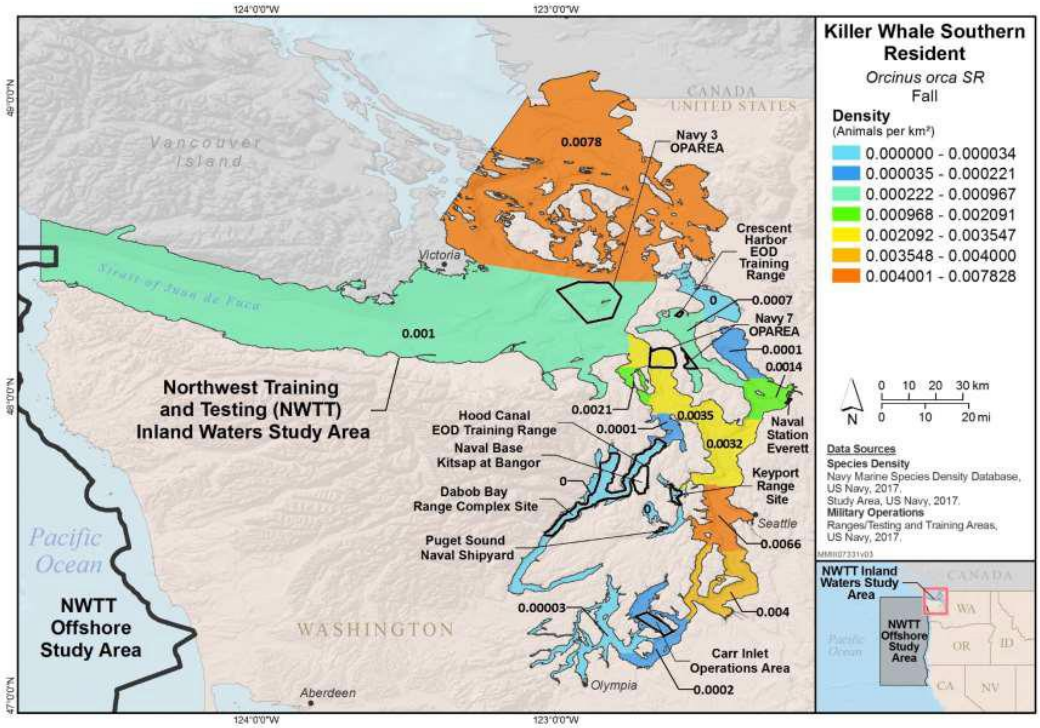


Figure 23. Inland waters fall distribution of Southern Resident killer whale (Navy 2019f).

2.4.2 Fish Density Estimates

The Navy's acoustic effects model did not estimate the number of instances of exposure for ESA-listed fish species. Rather, our exposure analysis for ESA-listed salmon, steelhead and eulachon is based on the methodology developed for the 2015 NWTT Phase II biological opinion. The approach used to derive density estimates for these species is described below. For fish densities, we mainly assume a uniform distribution throughout their area of habitat since we do not have available information on spatial distribution within the action area for calculating spatially explicit densities. However, for Central California Coast coho and several Chinook salmon Evolutionarily Significant Units (ESUs), we developed weighting factors which were applied to the uniform densities based on data indicating the proportion of fish from these ESUs that would likely be found within the NWTT Study Area[3]. During our analysis, we identified catch per unit of effort data for juvenile ESA-listed salmonids in the action area from Teel et al. (2015). While these data indicated variances in juvenile habitat use for several salmonid ESUs within the NWTT Study Area, it did not provide an analogue for density. In order to convert the catch per unit of effort data presented in Teel et al. (2015), multiple assumptions had to be taken into account including 100 percent sampling efficiency. Further, total catch counts (number of fish per trawl) from Teel et al. (2015) were not published. Last, as discussed in Section 3, the Navy's offshore explosive activities can occur at all times of the year in a number of areas within the offshore testing and training sites. Due to the lack of knowledge of when or where these events will occur, combined with a limited amount of information to ascertain offshore salmonid densities from catch per unit of effort data, a uniform density approach was deemed the most appropriate in most cases. However, in instances where information on the proportions of fish in the action area was available, weighting factors were applied to species densities to account for greater or smaller proportions of fish within the action area. Further, separate density estimates, as available, were used for offshore versus inland effects analyses. All data inputs were updated, as needed, to account for newly available information on abundance and distribution since the 2015 opinion. For Puget Sound/Georgia Basin DPS bocaccio (larval and juvenile fish), Puget Sound/Georgia Basin DPS yelloweye rockfish (larval fish), and Southern DPS green sturgeon in the inland portion of the action area, we did not have sufficient information to estimate densities to conduct a quantitative effects analysis.

2.4.2.1 Offshore Density Estimates

ESA-listed Salmonids and Eulachon

For each ESA-listed salmon ESU, eulachon ESU, and steelhead DPS, we estimated a density of animals in the offshore action area based on information regarding the species' distribution and abundance. For abundance estimates, we used data from several different sources (NMFS 2019c; NMFS 2019d; NMFS 2020c; Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020).

[3] Data was derived from Shelton et al. (2019) (see Figure 26 below), and Weitkamp and Neely (2002).

These abundance estimates are presented in Table 11 by lifestage and origin (i.e., natural, hatchery intact adipose-fin, and hatchery adipose clip). Most of these abundance estimates were calculated using the geometric mean. Adult abundance estimates represent the total number of returning spawners, and do not account for all adults in the ocean environment. It should be noted that ESA take prohibitions do not apply to hatchery fish with clipped adipose fins from threatened ESUs or DPSs. Additionally, hatchery fish are generally considered to be of less conservation value than individuals from the natural population (NMFS 2015).

Table 11. Summary of estimated annual abundance of ESA-listed salmonids. Abundance estimates for each ESU and DPS are divided into natural, listed hatchery intact adipose, and listed hatchery adipose clip.

Species	Life Stage	Natural	Listed Hatchery Intact Adipose	Listed Hatchery Adipose Clip
Sacramento River winter-run Chinook salmon	Adult	210	-	2,232
	Smolt	195,354	-	200,000
Central Valley spring-run Chinook salmon	Adult	3,727	-	2,273
	Smolt	775,474	-	2,169,329
California Coastal Chinook salmon	Adult	7,034	-	-
	Smolt	1,278,078	-	-
Snake River fall Chinook salmon	Adult	10,337	13,551	15,508
	Smolt	692,819	2,862,418	2,483,713
Snake River spring/summer Chinook salmon	Adult	12,798	421	2,387
	Smolt	1,007,526	775,305	4,453,663
Lower Columbia River Chinook salmon	Adult	29,469	38,594[1]	-
	Smolt	11,745,027	962,458	31,353,395
Upper Willamette River Chinook salmon	Adult	10,203	31,476[1]	-
	Smolt	1,211,863	157	4,709,045
Upper Columbia River spring Chinook salmon	Adult	2,872	3364	6,226
	Smolt	468,820	368,642	621,759
Puget Sound Chinook salmon	Adult	22,398	15,543[1]	-
	Smolt	3,035,288	7,271,130	36,297,500
Hood Canal summer run chum salmon	Adult	25,146	1,452	-
	Smolt	3,889,955	150,000	-
Columbia River chum salmon	Adult	10,644	426	-
	Smolt	662,6218	601,503	200,000
	Adult	1,932	327	559

Species	Life Stage	Natural	Listed Hatchery Intact Adipose	Listed Hatchery Adipose Clip
Central California Coast coho salmon	Smolt	158,130	165,880	60,000
Southern Oregon/Northern California Coast coho salmon	Adult	9,065	10,934	-
	Parr	2,013,593	575,000	7,287,647
Oregon Coast coho salmon	Adult	94,320	0	-
	Parr	6,641,564	0	-
Lower Columbia River coho salmon	Adult	29,866	8,791	-
	Smolt	661,468	249,784	-
Ozette Lake sockeye salmon	Adult	5,036[2]	0	0
	Smolt	1,037,787	259,250	45,750
Snake River sockeye salmon	Adult	546	-	4,004
	Smolt	19,181	-	242,610
South-Central California steelhead	Adult	695	-	0
	Smolt	79,057	-	0
Central California Coast steelhead	Adult	2,187	-	3,866
	Smolt	248,771	-	648,891
California Central Valley steelhead	Adult	1,686	-	3,856
	Smolt	630,403	-	1,600,653
Northern California steelhead	Adult	7,221	-	-
	Smolt	821,389	-	-
Upper Columbia River steelhead	Adult	1,931	1,163	5,309
	Smolt	199,380	138,601	687,567
Snake River Basin steelhead	Adult	10,547	16,137	79,510
	Smolt	798,341	705,490	3,300,152
Lower Columbia River steelhead	Adult	12,920	22297[1]	-
	Smolt	352,146	9138	1,197,156
Upper Willamette River steelhead	Adult	2,912	-	-
	Smolt	140,396	-	-
Middle Columbia River steelhead	Adult	5,052	112	448
	Smolt	407,697	110,469	444,973
Puget Sound steelhead	Adult	19,313[2]	-	-
	Smolt	2,196,901	112,500	110,000

[1] We do not have separate estimates for fin-clipped and intact adipose fin hatchery fish for the life stage of this DPS/ESU.

[2] Includes estimates for natural and hatchery fish (intact and clipped numbers)

The adult abundances above only presented run-size estimates for fish returning to their natal rivers to spawn as a quantification of adults. The number of returning adults is an underestimate of the number of post-juvenile fish that will occur in the oceanic environment since most Chinook salmon, chum salmon, sockeye salmon, and steelhead spend two to four years foraging and maturing in the ocean environment before returning to spawn. Coho salmon typically return to spawn at age three and thus spend approximately two years at sea, and eulachon typically spend three to five years at sea before returning to freshwater to spawn. Information is not available for all ESA-listed salmon and eulachon ESUs and steelhead DPSs to estimate the total oceanic abundance of these species (PFMC 2015). Therefore, we multiplied the number of returning adults for each ESU or DPS by the average number of years the species spends at sea before returning to spawn, in order to account for all age classes of fish that would be expected in the oceanic environment (i.e., three years for Chinook salmon, chum salmon, sockeye salmon, and steelhead; two years for coho salmon; four years for eulachon). As discussed below, these abundances were then used to estimate densities which were incorporated into our effects analysis (See Section 8.2.3).

We recognize that since this methodology is based on the number of returning adults, it does not account for individuals (i.e., subadults and adults) that die before returning to spawn. However, for our explosives effects analysis below (Section 8.2.3.1), estimated impacts (i.e., injury and mortality) to salmonids are expressed as impacts to adults, and are evaluated in relation to the adult population which is based on returning spawners. That is, in our effects analysis we assess the potential for population level effects by comparing mortality and injury from this action to the number of returning adults (which is generally how salmon, steelhead, and eulachon abundance and trends are tracked). Therefore, not accounting for subadult and adult mortality in our density estimate does not inhibit our ability to accurately conduct a jeopardy analysis and determine whether or not to expect any population level effects from this action.

Once we estimated the ocean abundance of maturing/adult and juvenile fish from each ESU/DPS, we estimated a density based on the expected habitat area (distribution) in the marine environment for each species. After a thorough review of densities used in NMFS (2015a), it was determined that updates to the distributions for ESA-listed Chinook, sockeye, chum, and coho salmon were necessary to account for updates to best available science and spatial data. Updated areas of offshore habitat for ESA-listed Chinook, sockeye, chum, and coho salmon were calculated using ArcPro version 2.3.1. The habitat area (distribution) data used for our density calculations is presented in Table 12, and a description of the geographic data inputs used to calculate the offshore habitat of ESA-listed Chinook, chum, coho, and sockeye salmon is discussed below and in Table 149 of Section 8.2.3.

The north-south oceanic distribution for Chinook salmon was based on results presented in Weitkamp (2010) and Shelton et al. (2019). Weitkamp (2010) used coded-wire-tags to estimate the distribution of Chinook salmon from various recovery areas along the west coast of North America (See Figure 24 and Figure 25). Furthermore, using the recovery areas presented in

Weitkamp (2010), Shelton et al. (2019) constructed the first coastwide state-space model for fall Chinook salmon tagged fish released from California to British Columbia between 1977 and 1990 to estimate seasonal ocean distribution along the west coast of North America (See Figure 26). For coho salmon, the north-south oceanic distribution was based on Weitkamp and Neely (2002).

Since coho salmon primarily reside on the continental shelf we used the shelf break as the westward boundary for all ESUs. The shelf break was defined as the 200 meter depth contour (Landry and Hickey 1989). In addition to coho, the continental shelf break was used as the western boundary for the majority of ESA-listed Chinook salmon ESUs and lifestages. However, there were instances where differences in western boundaries varied. For example, for the Puget Sound ESU of Chinook salmon, we used the 100 meter contour for juveniles in areas south of Cape Flattery, Washington[4]. This is due to trawling data from Teel et al. (2015) which show that juveniles belonging to the Puget Sound ESU of Chinook salmon are mostly confined to the 100 meter depth contour along the Washington and Oregon Coast. Also, coded wire tag data analyzed in Trudel et al. (2009b) showed the majority of juvenile Puget Sound Chinook salmon were caught in shallower waters in the Straight of Juan de Fuca and at the southern end of the west coast of Vancouver Island.

In addition to variations in the western boundary for juvenile Chinook salmon, for adult Chinook salmon ESUs, the offshore distribution areas used in the NWTT Study Area extended past the 200-meter depth contour out past 50 nautical miles from shore for Snake River spring/summer Chinook, Snake River fall Chinook, Upper Columbia River spring Chinook, and Upper Willamette River Chinook salmon based on offshore distribution data presented in Sharma and Quinn (2012). Further, in areas north of Cape Flattery, Washington, the distribution areas for Snake River spring/summer Chinook, Snake River fall Chinook, Upper Columbia River Chinook, Lower Columbia River Chinook, Upper Willamette River Chinook, and Puget Sound Chinook salmon were extended past 50 nautical miles based on data presented in Sharma and Quinn (2012).

For chum and sockeye salmon, the geographic distribution was based on high seas tagging experiments of the species from 1956 to 2004 as presented in Myers et al. (2007). Chum salmon migrate north and west once they leave their river of origin (Byron and Burke 2014; Quinn 2005) and as shown in Figure 2 of Myers et al. (2007), the southern extent of North American chum and sockeye salmon is the Washington Coast. As chum and sockeye salmon migrate to the ocean and travel north they can be found in father offshore waters.

We derived expected distribution data for steelhead from Light et al. (1989) which calculated the area (km^2) of offshore habitat[5]. NMFS (2015a) relied on the geographic ocean distribution of the

[4] In order to account for juvenile Chinook's northwestern migration in the Pacific Ocean Pearcy, W. G., and J. P. Fisher. 1990. Distribution and abundance of juvenile salmonids off Oregon and Washington, 1981-1985. Pages 83p *in.*, the 200 meter depth contour was used for all juvenile Chinook in areas north of Cape Flattery, Washington.
[5] Area of offshore habitat for ESA-list steelhead was calculated using ArcMap version 10.2.1 (ESRI, Redlands, CA)

species described in Light et al. (1989). NMFS (2015a) used the distribution of the species during winter, where they were most geographically condensed. This resulted in a higher species density that would have been estimated in other seasons, and therefore likely overestimates impacts to steelhead for detonations that occur in spring, summer, and fall. As we do not know when Navy detonations will occur, using this distribution is more conservative for the species since we are assessing impacts to the species should a worst-case, but plausible, scenario be implemented (i.e., all detonations occurring in winter when the species is most densely distributed). However, it is more likely that Navy explosive activities would be spread out more evenly throughout the seasons during times when steelhead are also more widely dispersed.

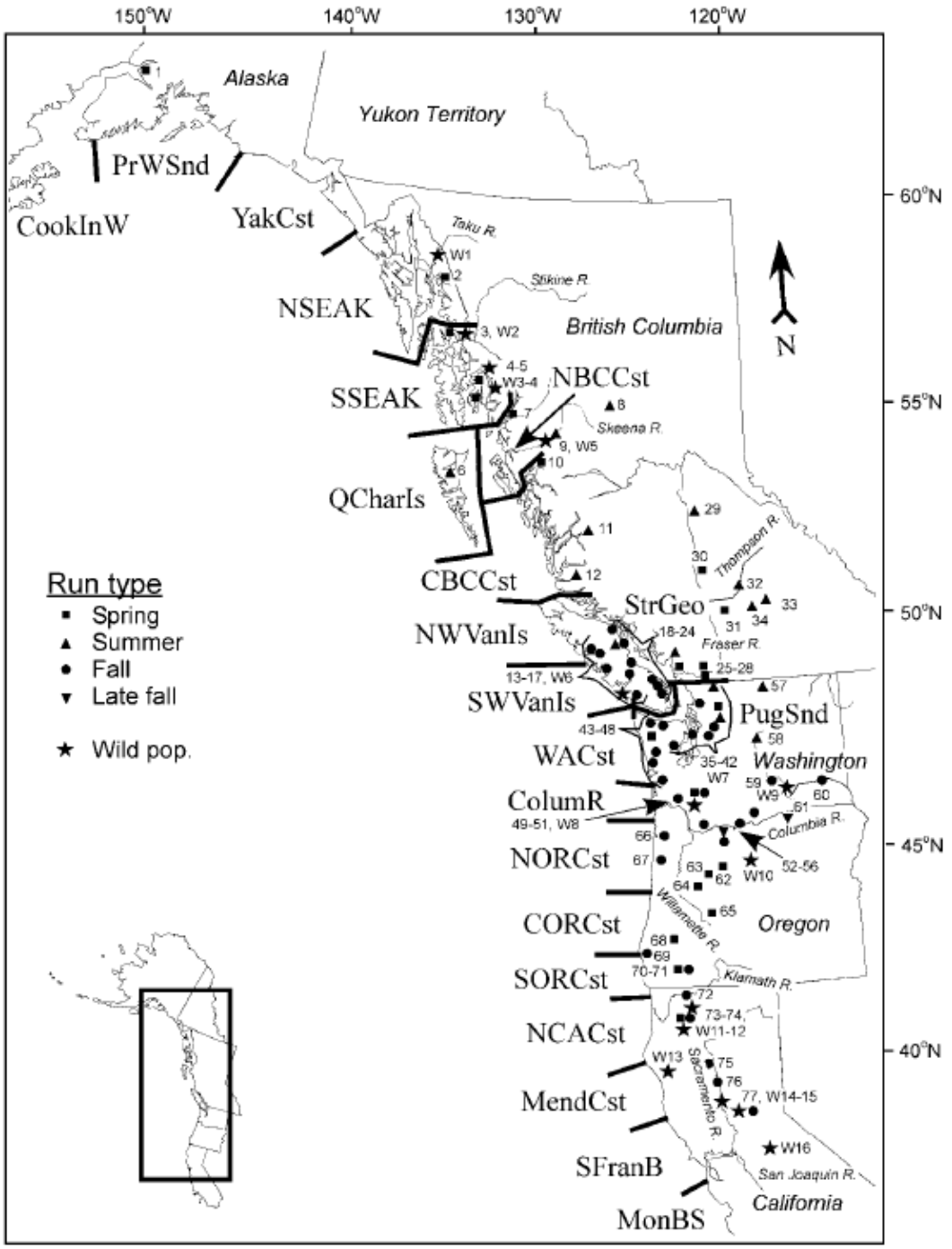


Figure 24. Locations of the 21 marine recovery areas (indicated by dark lines) used to estimate distributions (Weitkamp 2010).

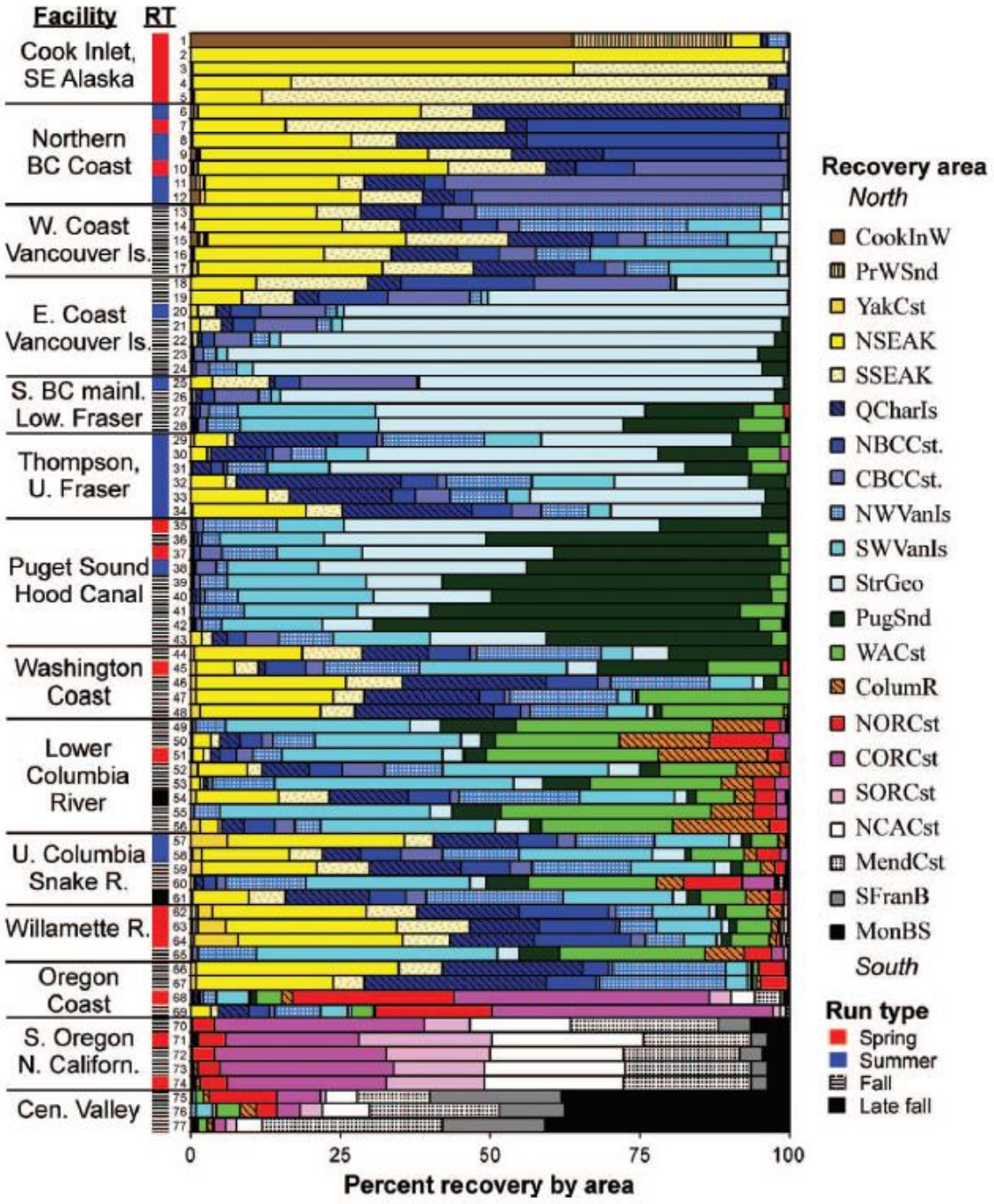


Figure 25. Recovery patterns for coded-wire-tagged Chinook salmon. Each horizontal bar represents the percentages of recoveries in the 21 marine recovery areas for a single hatchery run type group (Weitkamp 2010).

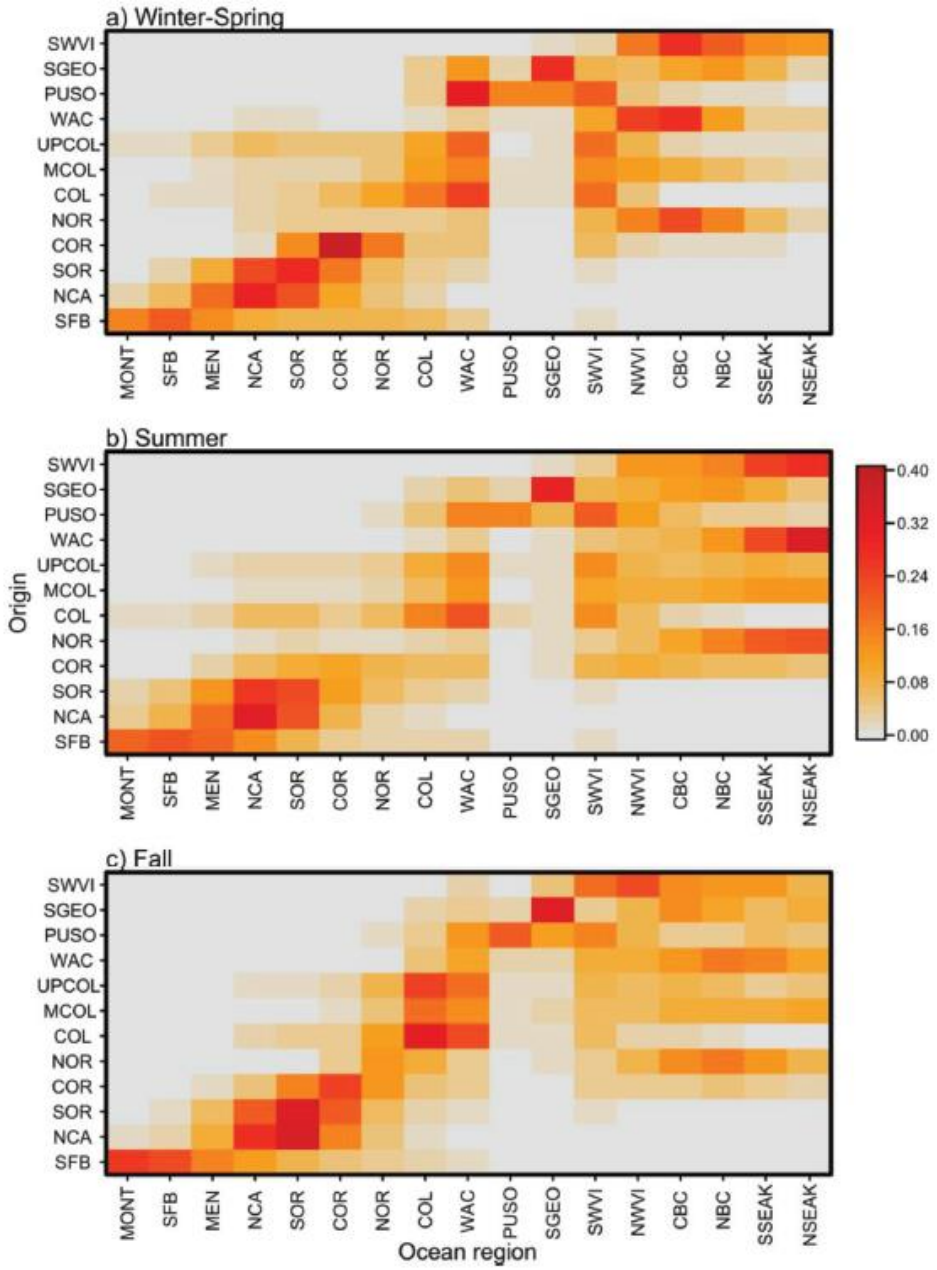


Figure 26. Estimated proportional spatial distribution by season of fall Chinook salmon originating from 11 different regions. Each row represents the proportion of fish from a region present in each ocean region (rows sum to 1) (Shelton et al. 2019). Acronyms are presented in footnote below[6].

[6] The abbreviations of the marine recovery areas are as follows: SWVI = southwest Vancouver Island; SGEO = Strait of Georgia; PUSO = Puget Sound; WAC = Washington coast; UPCOL/MCOL/COL= Columbia River Basin; NOR = north Oregon coast; COR: central Oregon coast; SORC = southern Oregon coast; NCA = northern California coast; SFB = San Francisco Bay.

Table 12. Habitat area (distribution) used for each salmonid ESU and DPS in the offshore marine environment.[7]

DPS/ESU	Marine Habitat Area (km^2)	Northern/Southern Extent	Western Boundary
Sacramento River winter-run Chinook salmon (Adult and Juvenile)	51,554	Washington Coast/Monterey Bay South	Continental Shelf (200 meter depth contour)
Central Valley spring-run Chinook salmon (Adult and Juvenile)	51,554	Washington Coast/Monterey Bay South	Continental Shelf (200 meter depth contour)
California Coastal Chinook salmon (Adult and Juvenile)	51,554	Northern California Coast/Monterey Bay South	Continental Shelf (200 meter depth contour)
Snake River fall Chinook salmon (Adult)	639,642	Yakutat Coast/ Central Oregon Coast	Inside action area: Maximum of 120 nautical miles from shore); North of action area: 145°W(See Figure 5 of Sharma and Quinn (2012))
Snake River fall Chinook salmon (Juvenile)	225,386	Yakutat Coast/ Central Oregon Coast	Continental Shelf (200 meter depth contour)
Snake River spring/summer Chinook salmon (Adult)	639,642	Yakutat Coast/ Central Oregon Coast	Inside action area: Maximum of 120 nautical miles from shore); North of action area: 145°W(See Figure 5 of Sharma and Quinn (2012))
Snake River spring/summer Chinook salmon (Juvenile)	225,386	Yakutat Coast/ Central Oregon Coast	Continental Shelf (200 meter depth contour)
Lower Columbia River Chinook salmon (Adult)	467,536	Northern Southeast Alaska/Central Oregon Coast	Inside action area: Continental Shelf (200 meter depth contour); North of action area: 145°W (See Figure 4 of Sharma and Quinn (2012))
Lower Columbia River Chinook salmon (Juvenile)	198,450	Northern Southeast Alaska/Central Oregon Coast	Continental Shelf (200 meter depth contour)
Upper Willamette River Chinook salmon (Adult)	639,642	Yakutat Coast/ Columbia River	Inside action area: Maximum of 120 nautical miles from shore); North

[7] It is important to note that these distributions are representative of the majority of area a specific ESU/DPS may be found, not inclusive of everywhere where a ESU/DPS has been caught.

			of action area: 145°W(See Figure 5 of Sharma and Quinn (2012))
Upper Willamette River Chinook salmon (Juvenile)	225,386	Yakutat Coast/ Columbia River	Continental Shelf (200 meter depth contour)
Upper Columbia River spring Chinook salmon (Adult)	639,642	Yakutat Coast/ Central Oregon Coast	Inside action area: Maximum of 120 nautical miles from shore); North of action area: 145°W(See Figure 5 of Sharma and Quinn (2012))
Upper Columbia River spring Chinook salmon (Juvenile)	225,386	Yakutat Coast/ Central Oregon Coast	Continental Shelf (200 meter depth contour)
Puget Sound Chinook salmon (Adult)	450,526	Northern Southeast Alaska/Washington Coast	Inside action area: Continental Shelf (200 meter depth contour); North of action area: 145°W(See Figure 6 of Sharma and Quinn (2012))
Puget Sound Chinook salmon (Juvenile)	176,591	Northern Southeast Alaska/Washington Coast	100-meter depth contour
Chum salmon (all ESUs)	4,376,644	North and westward migration; primarily occur north of 48 °N (north of NWTT)	171°E (See Figure 2 of Myers et al. (2007))
Central California Coast coho salmon	38,165	Central Oregon /Monterey South	Continental Shelf (200 meter depth contour)
Southern Oregon/Northern California Coast coho salmon	74,203	Central Oregon /Monterey South	Continental Shelf (200 meter depth contour)
Oregon Coast coho salmon	74,203	Southwest Vancouver Island /Northern California	Continental Shelf (200 meter depth contour)
Lower Columbia River coho salmon	61,827	Southwest Vancouver Island /Northern California	Continental Shelf (200 meter depth contour)
Sockeye salmon (all ESUs)	5,434,790	North and westward migration; primarily occur north of 48°N (north of NWTT)	167°E (See Figure 2 of Myers et al. (2007))
Steelhead (all DPSs)	6,083,400	Southern California/Northern Alaska	161°E (See Figure 16 of Light et al. (1989))

Offshore densities used for ESA-listed salmonids are presented in Table 13. These densities were derived from the population abundances presented in Table 11 and adjusted for adults to account for years at sea, and the marine habitat areas presented in Table 12. An additional density weighting factor was then applied for several Chinook salmon ESUs and Central California Coast coho salmon to account for non-uniform densities within portions of the NWTT Study Area where fish would likely be exposed to explosives. We applied a weighting factor of 0.47 for California Chinook salmon ESUs, 1.7 for Columbia/Snake/Willamette River Chinook salmon ESUs (in waters <50 nautical miles from shore), and 0.33 for Central California Coast coho salmon. These weighting factors were derived from seasonal estimated proportional spatial distribution data for Fall-run Chinook salmon and coho salmon from (Shelton et al. 2019) (see Figure 26) and Weitkamp and Neely (2002). For all other ESA-listed salmonids shown in Table 13, based on the information available, we assumed a uniform density distribution (i.e., density weighting factor of 1.0) within the offshore portions of the action area where exposure to explosives would likely occur. For Columbia/Snake/Willamette River Chinook salmon ESUs impacted by explosives in locations greater than 50 nautical miles, we applied a uniform distribution due to the species having lower densities in waters farther offshore.

Table 13. Offshore density estimates for ESA-listed salmonids in the NWTT Study Area.

Species	Life stage	ESU/DPS	Density weighting factor applied	Density (# fish/km^2)
Chinook salmon	Adult	Sac River winter run	0.47	0.06678861
	Juvenile		0.47	3.604305776
	Adult	Central valley spring run	0.47	0.164099779
	Juvenile		0.47	26.84675117
	Adult	California coastal	0.47	0.192379641
	Juvenile		0.47	11.6517954
	Adult (<50 nm from shore)	Snake River fall	1.7	0.314112582
	Adult (>50 nm from shore)		1.0	0.184772107
	Juvenile		1.7	45.54947956
	Adult (<50 nm from shore)	Snake River spring/summer	1.7	0.124429915
	Adult (>50 nm from shore)		1.0	0.073194068
	Juvenile		1.7	47.03947805
	Adult	Lower Columbia River	1.7	0.73131747
	Juvenile		1.7	377.4426606
	Adult (<50 nm from shore)	Upper Willamette River	1.7	0.335753571
	Adult (>50 nm from shore)		1.0	0.197502101
	Juvenile		1.7	49.25576929
	Adult (<50 nm from shore)	Upper Columbia River spring	1.7	0.099362143
	Adult (>50 nm from shore)		1.0	0.05844832
	Juvenile		1.7	11.00634334
	Adult	Puget Sound	1.0	0.252644686
	Juvenile		1.0	263.9087949
Coho salmon	Adult	Central California coast	0.33	0.039065636

Species	Life stage	ESU/DPS	Density weighting factor applied	Density (# fish/km^2)
	Juvenile		0.33	2.801606184
	Adult	S. Oregon/N. California coast	1.0	0.53903481
	Juvenile		1.0	37.58059647
	Adult	Oregon coast	1.0	2.557282051
	Juvenile		1.0	90.31392262
	Adult	Lower Columbia River	1.0	1.250489268
	Juvenile		1.0	132.6103321
Chum salmon	Adult	Hood Canal summer run	1.0	0.018231778
	Juvenile		1.0	0.923071422
	Adult	Columbia River	1.0	0.007588006
	Juvenile		1.0	1.651429954
Sockeye salmon	Adult	Ozette Lake	1.0	0.002779868
	Juvenile		1.0	0.247072472
	Adult	Snake River	1.0	0.002511597
	Juvenile		1.0	0.048169478
Steelhead	Adult	South-Central California	1.0	0.000342736
	Juvenile		1.0	0.012995529
	Adult	Central California	1.0	0.002985008
	Juvenile		1.0	0.14755926
	Adult	California Central Valley	1.0	0.002733011
	Juvenile		1.0	0.366744912
	Adult	Northern California	1.0	0.003561002
	Juvenile		1.0	0.13502137
	Adult	Upper Columbia River	1.0	0.0041439
	Juvenile		1.0	0.168581385
	Adult	Snake River basin	1.0	0.05236907
	Juvenile		1.0	0.789687182
	Adult	Lower Columbia River	1.0	0.017367097
	Juvenile		1.0	0.25617911
	Adult	Upper Willamette River	1.0	0.001436039
	Juvenile		1.0	0.023078542
	Adult	Middle Columbia River	1.0	0.002767531
	Juvenile		1.0	0.158322484
	Adult	Puget Sound	1.0	0.009524115
	Juvenile		1.0	0.397705395

To determine the average density of southern DPS eulachon in the offshore environment we used a similar methodology as described for estimating salmonid densities above. NMFS (2015a) determined that the southern DPS of eulachon has a marine distribution area of 1,183,304 km^2. The latest estimate of the population abundance of the southern DPS of eulachon was 18,796,090 spawners estimated in the Columbia River and Fraser River from 2014 to 2018. Because we do not have estimates of eulachon abundance in marine waters, the number of spawners in the Columbia River and Fraser River was used as a proxy for abundance in the oceanic environment. We multiplied the number of returning adults by the average number of years the species spends at sea before returning to spawn, in order to account for all age classes of fish that would be expected in the oceanic environment (i.e., four years for eulachon). This method produced a total Southern DPS eulachon density estimate of 63.54 km^2.

Non ESA-listed Salmonids

Density estimates were also produced for salmonid populations that are not listed under the ESA to support our analysis of the effects of the proposed action on Southern Resident killer whales via impacts to their prey. We focused our analysis on the primary target prey species based on the best available information on the Southern Resident killer whale diet composition (see Section 6.2.4 below for details). For non-listed species, this included nine non-listed Chinook salmon ESUs within the action area, Puget Sound/Georgia Strait coho salmon ESU, and Puget Sound/Georgia Strait chum salmon ESU. We estimated a density of animals in the offshore portion of the action area based on information, as available, regarding species' distribution and abundance. For non-listed salmonids we used the same approach as that used to estimate densities of ESA-listed salmonids (as described above).

Abundance data for non-listed salmonids were provided by the NMFS West Coast Regional Office and derived from multiple published and unpublished sources (M. Dennis, NMFS personal communication to R. Salz, NMFS, March 12, 2020). Since only adult abundance estimates were available for non-listed ESUs, we did not estimate juvenile densities for these populations. As was the case for ESA-listed species above, the only data available for non-listed ESUs were run-size estimates of adult fish returning to their natal rivers to spawn, as shown in Table 14.

Table 14. Summary of estimated adult annual abundance of non-listed Chinook salmon, coho salmon, and chum salmon ESUs included in effects analysis for Southern Resident killer whales prey (Data source: M. Dennis, NMFS personal communication to R. Salz, NMFS, March 12, 2020).

Species	ESU	Natural	Hatchery
Chinook salmon	Central Valley Fall-run	68,361	38,538
	Sacramento River Late Fall-run	7,927	---

Species	ESU	Natural	Hatchery
	Klamath River Fall-run	92,006	22,676
	Oregon Coast	175,000	---
	Upper Columbia River Fall-run	264,542	16,201
	Middle Columbia River Spring-run	78,003	---
	Washington coast	35,858	23,093
	Southern Oregon/Northern California Coast (SONCC)	6,809	8,916
	Deschutes River	9,200	171
Coho salmon	Puget Sound/Georgia Strait	283,145	229,863
Chum salmon	Puget Sound/Georgia Strait	373,988	77,243

The average annual number of returning adults is an underestimate of the number of post-juvenile fish that will occur in the oceanic environment since most salmon spend two to four years foraging and maturing in the ocean environment before returning to spawn. Therefore, we multiplied the number of returning adults for each ESU by the average number of years the species spends at sea before returning to spawn, in order to account for all age classes of fish that would be expected in the oceanic environment (i.e., three years for Chinook salmon and chum salmon; two years for coho salmon). We recognize that since this methodology is based on the number of returning adults, it does not account for individuals that die before returning to spawn.

As was done above for ESA-listed ESUs, the north-south oceanic distribution for non-listed Chinook salmon, coho salmon, and chum salmon was based on the results presented in Weitkamp (2010) and Weitkamp and Neely (2002) and we used the shelf break as the westward boundary of these species' distribution. The habitat area used in the offshore marine environment by each non-listed salmon ESU included in our effects analysis for Southern Resident killer whales prey is shown in Table 15. For details, see above approach used for ESA-listed salmonids.

The continental shelf break (i.e., 200-meter depth contour) was used as the western boundary for adult non-listed coho and chum ESUs and for the majority of non-listed Chinook salmon ESUs. The offshore distribution areas used in the NWTT Study Area extended past the 200-meter depth contour out past 50 nautical miles from shore for the Upper Columbia River fall and Deschutes River Chinook salmon ESUs based on offshore distribution data presented in Sharma and Quinn (2012). Further, in areas north of Cape Flattery, Washington, the distribution areas for these two ESUs were extended past 50 nautical miles based on data presented in Sharma and Quinn (2012).

The estimated offshore adult densities for non-listed salmonids, based on the adjusted abundance estimates (i.e., to account for years at sea) and habitat area estimates (from Table 15) are shown in Table 16. An additional density weighting factor was then applied for several Chinook salmon ESUs to account for non-uniform densities within the portions of the NWTT Study Area where fish would likely be exposed to explosives. We applied a density weighting factor of 0.47 for Central Valley Fall-run Chinook salmon, Sacramento River Late Fall-run Chinook salmon, and Klamath River Fall-run Chinook salmon. Also, a weighting factor of 1.7 was used for Upper Columbia River Fall-run Chinook salmon, Middle Columbia River Spring-run Chinook salmon, and Deschutes River Chinook salmon. For the Upper Columbia River fall and Deschutes River Chinook salmon ESUs this weighting factor was not applied to the density estimate for areas greater than 50 nautical miles from shore. These weighting factors were derived from seasonal estimated proportional spatial distribution data for Fall-run Chinook salmon and coho salmon from (Shelton et al. 2019) (see Figure 26) and Weitkamp and Neely (2002). For all other non-listed ESUs shown in Table 16, based on the information available, we assumed a uniform density distribution (i.e., density weighting factor of 1.0) within the offshore portions of the NWTT Study Area where exposure to explosives would likely occur.

Table 15. Adult habitat area (distribution) used in the offshore marine environment by each non-listed salmon ESU included in the effects analysis for Southern Resident killer whale prey.

Non-listed ESU	Marine Habitat Area (km^2)
Central Valley Fall-run Chinook salmon	51,554
Sacramento River Chinook salmon Late Fall-run	51,554
Klamath River Chinook salmon Fall-run	51,554
Oregon Coast Chinook salmon	467,536
Upper Columbia River Chinook salmon Fall-run	639,642

Non-listed ESU	Marine Habitat Area (km^2)
Middle Columbia River Chinook salmon Spring-run	467,536
Washington Coast Chinook salmon	450,526
Southern Oregon/Northern California Coast Chinook salmon	51,554
Deschutes River Chinook salmon	639,642
Puget Sound/Georgia Strait Coho salmon	61,827
Puget Sound/Georgia Strait Chum salmon	4,376,644

Table 16. Offshore adult density estimates in the NWTT Study Area for non-listed salmon ESUs included in the effects analysis for Southern Resident killer whales prey (includes natural and hatchery origin fish).

Non-listed ESU		Density weighting factor applied	Density (# adult fish/km^2)
Central Valley Chinook salmon Fall-run		0.47	2.924
Sacramento River Chinook salmon Late Fall-run		0.47	0.217
Klamath River Chinook salmon Fall-run		0.47	3.137
Oregon Coast Chinook salmon		1.0	1.123
Upper Columbia River Chinook salmon Fall-run	< 50 nm from shore	1.7	2.238
	> 50 nm from shore	1.0	1.317
Middle Columbia River Chinook salmon Spring-run		1.7	0.851
Washington Coast Chinook salmon		1.0	0.393
Southern Oregon/Northern California Coast Chinook salmon		1.0	0.915
Deschutes River Chinook salmon	< 50 nm from shore	1.7	0.075
	> 50 nm from shore	1.0	0.044
Puget Sound/Georgia Strait Coho salmon		1.0	16.595
Puget Sound/Georgia Strait Chum salmon		1.0	0.309

Southern DPS Green Sturgeon

We estimated Southern DPS green sturgeon density in the offshore portion of the NWTT Study Area based on acoustic tag detection data from the NMFS Northwest Fisheries Science Center

and population abundance estimates from Mora et al. (2018). NMFS monitors an array of 115 acoustic receivers off the coast of Washington State, sometimes referred to as the Salmon Ocean Behavior and Distribution acoustic receiver array. The acoustic receivers are spaced approximately 4.5 kilometers apart from each other and each receiver has a detection range area of about one square kilometer. Assuming there is no overlap between receiver detection ranges, we estimate that each receiver has a one square kilometer detection area (e.g., 46 receivers would have a combined detection area of approximately 46 square kilometers). A total of 124 acoustically tagged individual green sturgeon were detected within this array May through September 2019 (see Figure 64 in Section 6.2.25 below), 44 of which were identified as Southern DPS green sturgeon. Detected fish included both subadults and adults, ranging in size from 98 to 170 centimeters fork length (unpublished data from S. Corbett, NMFS to R. Salz, NMFS, provided on March 5, 2020).

While green sturgeon are occasionally found farther offshore (i.e., out to 200 meters) than the NMFS receiver array covers, based on Huff et al. (2011) we anticipate they spend the large majority of their time at depths of 60 meters or shallower when in the offshore portion of the NWTT Study Area (Huff et al. 2011). The only explosives proposed for NWTT Phase III that could occur in these shallower water areas (i.e., within 12 nautical miles from shore) would be from mine countermeasure and neutralization testing (i.e., bins E4 and E7) within the Quinault Range Site. Therefore, our offshore Southern DPS green sturgeon density estimate is based on tag detections at the subset of 46 receivers (out of 124 total) that are within the Quinault Range Site (note: not all 46 receivers were deployed every month, i.e., range from 22 to 46 receivers deployed per month). This conservative approach for estimating density assumes that Navy mine countermeasure and neutralization events would occur within the portion of the Quinault Range (i.e., depths less than 60 meters) with the highest anticipated densities of Southern DPS green sturgeon.

The Navy has proposed up to two mine countermeasure and neutralization events within the Quinault Range Site per year, with each event lasting from one to ten days (see Section 5.1.2 for details on this activity). To minimize impacts to ESA-listed species, the Navy has proposed a seasonal limit on the number of explosives that would be used during mine countermeasure and neutralization events within 20 nautical miles from shore (i.e., within the Marine Species Coastal Mitigation Area) from October 1 through June 30 (i.e., not to exceed 20 E4 and 3 E7 explosives per year; not to exceed 60 E4 and 9 E7 explosives over seven years). This proposed mitigation measure was based on very low Southern DPS green sturgeon acoustic tag detection rates from July through September from unpublished NMFS Northwest Fisheries Science Center data (J.M. Smith, NMFS, pers. comm. to R. Salz, NMFS, February 24, 2020). To account for this mitigation measure in our exposure analysis, we estimate Southern DPS green sturgeon densities for two different seasonal time frames: 1) October through June and 2) July through September.

Tag detection data were compiled by the Northwest Fisheries Science Center for the time period from May 2019 through March 2020, and made available for our analysis. We used the number

of unique Southern DPS green sturgeon detections per acoustic receiver deployed (i.e., detection rate) during a single day (i.e., 24-hour window) to approximate an instantaneous density for estimating the impacts from explosives detonated during the course of each mine countermeasure neutralization event. While in some instances fish may be quickly migrating through the detection area of a receiver, in other cases fish may be spending several hours or even days in the same general vicinity, particularly in areas with high structural seafloor complexity that may provide opportunities for feeding and sheltering (Huff et al. 2011). For purposes of our analysis we conservatively assume that the individual green sturgeon detected across all deployed receivers during a particular 24-hour period could all occur within the detection area simultaneously.

While the mine countermeasure neutralization activity could occur during any month of the year, for personnel safety reasons the Navy has proposed to only conduct this activity in Beaufort Sea States of three or less. Based on information provided by the Navy, the months with sea states that would likely be conducive for this activity are May through November (Navy 2020d). Thus, we based our green sturgeon density estimates on tag detection data only for these seven months (i.e. May through November), when mine countermeasure neutralization testing events with explosives would most likely occur. That is, estimated density for the October through June seasonal time frame was based only on acoustic tag detection data from May, June, October, and November, while estimated density for the July through September seasonal time frame was based on tag detection data from July, August and September. Based on information provided by the Navy, the months when mine countermeasure neutralization testing events would be conducted can vary from year to year depending on a number of factors including platform availability, range availability, and weather (Navy 2020d) . Thus, to arrive at an estimated density for each seasonal time frame we averaged the estimated densities across months.

From the available data, tag detection rates across all receivers combined can vary greatly from one day to the next, indicating frequent movement of fish in and out of the combined acoustic receiver field. For all months, the number of unique Southern DPS green sturgeon detected during a single day (i.e. 24-hour period) across all deployed receivers within the Quinault Range Site ranged from zero to four fish. Although a mine countermeasure neutralization event can last up to a maximum of ten days, we have no specific information regarding the duration of a particular event (i.e., from 1 to 10 days) or the distribution of explosives across the days of an event. For purposes of our analysis, we assume that, on average, each mine countermeasure neutralization event would last five days. Since we have no specific information regarding the particular weeks or days within each month that this activity may occur, we took a conservative approach based on the peak tag detection rates recorded within each month. This approach assumes that Navy mine countermeasure and neutralization events would coincide with the particular days within a given month when the highest densities of Southern DPS green sturgeon could be within the impacted area. Thus, to estimate the peak tag detection rate that could occur for an event during a given month, we calculated the average tag detection rate across the five

consecutive days representing the peak five-day period for acoustic tag detections during that particular month (see Table 17 for calculation details).

As noted above, the next step was to average the estimated five-day peak detection rates across months to arrive at the estimated five-day peak acoustic tag detection rate for each seasonal time frame. Using this approach, we arrive at the following estimated peak tag detection rates for each seasonal time frame: October-June 0.050 fish per receiver; July-September 0.010 fish per receiver. Since each receiver has a detection range of approximately one square kilometer, these tag detection rates can be expressed as the following densities of acoustically tagged Southern DPS green sturgeon: October-June 0.0423 fish per km^2; July-September 0.0058 fish per km^2 (Table 17).

The next step was to extrapolate the estimated densities of acoustically tagged fish (0.0423 and 0.0058 per km^2) up to the entire population of Southern DPS green sturgeon. To do this we needed to determine what proportion of the population were equipped with active (working tags) during the analysis time frame (i.e., May through November, 2019). The fish detected during this time frame were a subset of the 145 Southern DPS green sturgeon fitted with acoustic tags that were projected (by the tag manufacturer) to expire after January 1, 2019. Since there were no fish detected that had tags projected to expire before January 2019, we assume these 145 fish included all the Southern DPS fish with active, working tags during 2019.

Next, we compared tag detection rates between fish with tags projected to expire in 2019 (all of these were from January through May) and fish with tags projected to expire sometime after February, 2020. Sixty-six fish had tag expiration dates between January and May of 2019. Twenty-five of these were detected from May through September, 2019 in the Salmon Ocean Behavior and Distribution acoustic receiver array for a detection rate of 38 percent. By comparison, 78 fish had projected tag expiration dates after February 2020 (one fish did not have an expiration date reported). Nineteen of these were detected from May through September, 2019 in the array for a detection rate of 24 percent. While other variables may be affecting the different detection rates found between these two groups, we found no indication that tags that were projected to expire in early 2019 were less likely to be detected from May through September, 2019 than tags projected to expire in 2020 and beyond (if anything detection rates were higher for the tags that were projected to expire in early 2019). Therefore, we assume that there were 145 Southern DPS fish with active, working tags for the time frame analyzed (i.e., May through November, 2019).

Table 17. Calculation of Southern DPS green sturgeon acoustic tag detection rates by month and seasonal time frame (rates based on peak 5-day period within each month).

Seasonal Time Frame	Months when activity likely to occur[1]	Number of acoustic receivers deployed	Number of fish detected across 5 consecutive peak days of month	Average number of fish detected per day (i.e., 5 day average)	Monthly peak tag detection rate (based on peak 5-day period)
October through June	October	29	day one: 1 fish day two: 0 fish day three: 1 fish day four: 2 fish day five: 1 fish	1.0	0.0345 fish per receiver
	November	29	day one: 1 fish day two: 1 fish day three: 2 fish day four: 1 fish day five: 0 fish	1.0	0.0345 fish per receiver
	May	46	day one: 2 fish day two: 3 fish day three: 1 fish day four: 3 fish day five: 1 fish	2.0	0.0435 fish per receiver
	June	46	day one: 3 fish day two: 3 fish day three: 2 fish day four: 2 fish day five: 3 fish	2.6	0.0565 fish per receiver
	Estimated peak tag detection rate for Oct-Jun seasonal time frame				**0.0423 fish per receiver**
July through September	July	46	day one: 1 fish day two: 2 fish day three: 1 fish day four: 0 fish day five: 0 fish	0.8	0.0174 fish per receiver
	August	46	day one: 0 fish day two: 0 fish day three: 0 fish day four: 0 fish day five: 0 fish	0.0	0.0 fish per receiver
	September	29	day one: 0 fish day two: 0 fish day three: 0 fish day four: 0 fish day five: 0 fish	0.0	0.0 fish per receiver
	Estimated peak tag detection rate for Jul-Sep seasonal time frame				**0.0058 fish per receiver**

[1] The Navy has proposed to only conduct mine countermeasure neutralization with explosives in Beaufort Sea States of three or less. Based on information provided by the Navy, we assume the months with sea states that would likely be conducive for this activity are May through November (Navy 2020d).

The 145 tagged fish ranged in size from 58 to 201 centimeters fork length, at the time of tagging. Of the fish that we have length data for, about 31 percent were adults and 69 percent were subadults, at the time of tagging. Since data were only available in fork length, we used 152 centimeters fork length as the subadult/adult cutoff based on Rien et al. 2001 as cited in Beamesderfer et al. (2007) (i.e., total length = 1.09 fork length). The relative proportion of adults to subadults among tagged fish is somewhat higher than the proportion of adults to subadults for the population as a whole based on Mora et al. (2018) (16 percent adults; 2,106 adults and 11,055 subadults). However, since we do not know the lengths of fish at the time of detection, we do not know the life stages at the time of detection, and therefore cannot compare detection rates between subadults and adults. Therefore, for purposes of our analysis, we combine these two life stages and assume subadults and adults occur within the Quinault Range Site at densities roughly proportional to the estimated relative proportion of each life stage in the population as a whole (i.e., 16 percent adults, 84 percent subadults).

From above, we assume there were 145 Southern DPS fish (adults and subadults combined) with working tags for the time frame May to November, 2019. Based on Mora et al. (2018), the estimated combined adult/subadult population size for this DPS is 13,161 fish. Therefore, we estimate that tagged fish represent 1.1 percent (i.e., 145/13,161) of the total adult/subadult population. We then extrapolate up to the estimated peak density of all Southern DPS green sturgeon (i.e., tagged and untagged fish) for each seasonal time frame as follows:

October through June

0.0423 tagged Southern DPS fish per km^2 X (100/1.1) = 3.85 Southern DPS fish per km^2

July through September

0.0058 tagged Southern DPS fish per km^2 X (100/1.1) = 0.53 Southern DPS fish per km^2

The estimated densities above represents fish in both the subadult and adult life stages. We assume the relative proportion of adults to subadults is the same as for the population based on Mora et al. (2018) (i.e., 16 percent adults, 84 percent subadults).

As discussed above, our approach to estimating Southern DPS green sturgeon densities for use in our explosives effects analysis (Section 8.2.3.1) was based on several conservative assumptions. We recognize that while, in some instances, these assumptions may result in an overestimation of Southern DPS green sturgeon densities, we believe that a conservative approach is warranted here given the lack of specificity regarding the timing and location of the mine countermeasure neutralization testing events, and the limitations associated the tag detection data available for estimating densities. These limitations include: 1) the short time period for which data were available (11 months); 2) the relatively small proportion of fish that have acoustic tags (i.e., about one percent of the entire Southern DPS green sturgeon population); 3) the small detection area (i.e., about 46 km^2) covered by acoustic receivers within the Quinault Range Site relative to the total area within the range where sturgeon could be exposed to explosives; and 4) the unknown tag detection failure rate which may result in an underestimate of detected fish. Despite

these limitations, we believe that the Northwest Fisheries Science Center unpublished green sturgeon acoustic tag detection data represent the best available information for estimating Southern DPS green sturgeon densities in the NWTT offshore portion in areas where this population is most likely to be impacted by NWTT explosive activities.

2.4.2.2 Inland Density Estimates

The following ESA-listed salmonids would likely overlap with Navy activities involving the use of explosives within the Puget Sound inland water Crescent Harbor and Hood Canal Explosive Ordnance Disposal training ranges: Chinook salmon Puget Sound ESU, steelhead Puget Sound DPS, and chum salmon Hood Canal summer-run ESU.

Chinook salmon

To estimate the density of juvenile Puget Sound ESU Chinook salmon in Crescent Harbor we used juvenile density information from unpublished data sampled in southwest Skagit Bay, directly adjacent to Crescent Harbor, by the Northwest Fisheries Science Center. Because we do not know when the Navy will be conducting Explosive Ordnance Disposal activities, we used the average annual maximum densities from 2003 to 2019 equating to 446 unmarked and 406 marked fish/square kilometer (C. Greene, NMFS, personal communication to J. Molineaux, NMFS, August 7, 2020). For adults, we do not have open water density information for the Explosive Ordnance Disposal site in Crescent Harbor or any similar areas adjacent to Crescent Harbor. Adult Chinook salmon density in Crescent Harbor is likely much lower than other areas and habitat types in the Sound (e.g., river mouths, deltas, and estuaries of the Skagit and Snohomish Rivers) because there are no documented spawning rivers in the immediate vicinity of the Explosive Ordnance Disposal site.

In order to derive density estimates for Hood Canal, we divided the number of adult and juvenile fish expected in Hood Canal[8] by the area of the canal (Hood Canal has a surface area of 358.4 square kilometers) resulting in a density of 86.81 per square kilometer juveniles of natural origin, 284.54 per square kilometer juveniles from hatcheries, 1.085 per square kilometer adults of natural origin, and 3.56 per square kilometer adults from hatcheries. Lacking detailed information on residence time of out-migrating juvenile or returning adult Chinook salmon in Hood Canal, and since Chinook salmon are thought to spend significantly more time rearing in estuarine and nearshore waters before migrating into open water habitats (months instead of days), we conservatively assumed all Chinook salmon originating from and returning to Hood Canal rivers could reside in the canal concurrently. This likely overestimates Chinook salmon abundance in the canal since some proportion of juveniles will spend less time in the canal than others and some will die from other causes (e.g., predation) while migrating through the canal. Additionally, adult migration is staggered such that some returning fish will have likely already entered freshwater before others have entered Hood Canal.

[8] These are derived from abundance numbers from all populations of Puget Sound ESU Chinook found in Hood Canal. See Section 8.2.3.1 for more details

Steelhead

We do not have information available to quantitatively estimate the density of juvenile or adult steelhead that may migrate through the Explosive Ordnance Disposal site at Crescent Harbor. To estimate steelhead juvenile density at the Explosive Ordnance Disposal site in Hood Canal, we first determined the average number of fish leaving freshwater from Hood Canal rivers per day by dividing the annual number of outmigrating juveniles[9] by the number of days in the migration period (March through June; 122 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (17 days, as described above). This results in the average number of juvenile Puget Sound steelhead we would expect in the Canal on a given day during the middle portion of the out-migration period (during first and last 17 days of the out-migration period, fewer individual steelhead would be expected in the Canal). We then divide this number by the area of Hood Canal (358.4 km^2) to determine juvenile steelhead density during the outmigration period. We assumed all detonations will occur during the outmigration period as the Navy is unable to predict when detonations will occur. This resulted in a juvenile density of 363.7 per km^2 steelhead. For adults, we used unpublished redd count data shared by the NWFSC as a surrogate to determine the maximum total percentage of adults that may be in Hood Canal at any time (B. Berejikian, NMFS, personal communication to J. Molineaux, NMFS, July 28, 2020). To determine the maximum proportion of adult steelhead in Hood Canal, we used the maximum monthly percentage of redd counts in Hood Canal for years 2007 to 2016 averaged across all major Hood Canal rivers and tributaries. This resulted in a maximum proportion of 51.5 percent of steelhead adults in Hood Canal at any period of time throughout the year. We then multiplied this percentage by the total annual abundance of adult steelhead that could be present in Hood Canal[10] to get a total maximum abundance of adults that could be present in Hood Canal during the year. This number was then divided by the area of Hood Canal to estimate adult density in the Canal on a given day during the run. This resulted in an adult density of 1.6467 per km^2 steelhead.

Chum salmon

To estimate Hood Canal summer-run chum salmon juvenile density at the Explosive Ordnance Disposal site in Hood Canal, we first determined the average number of fish leaving freshwater from adjacent Hood Canal rivers/tributaries per day by dividing the annual number of outmigrating juveniles[11] by the number of days in the migration period (74 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (14 days, as described above). This results in the average number of juvenile Hood Canal summer-run chum salmon that we would expect in the Canal on a given day during the middle portion of the out-migration period (during the first and last 14 days of the out-

[9] These are derived from abundance numbers from all populations of Puget Sound steelhead found in Hood Canal. See Section 8.2.3.1 for more details.
[10] These are derived from abundance numbers from all populations of Puget Sound steelhead found in Hood Canal. See Section 8.2.3.1 for more details.
[11] These are derived from abundance numbers from all populations of Hood Canal summer-run salmon found in the Navy's Hood Canal EOD test site. See Section 8.2.3.1 for more details.

migration period, fewer individual salmon would be expected in the Canal). For adults, we did not have information to estimate the amount of time each fish would be expected to reside in Hood Canal before entering freshwater to spawn. Therefore, we assumed the entire annual summer-run could occur in the Canal on a given day. We then divide this number by the area of Hood Canal (358.4 km^2) to determine summer-run chum salmon density during the outmigration period (for juveniles) and spawning period (for adults). This resulted in a density of 1,651 per km^2 juvenile fishes of natural origin, 115 per km^2 juveniles from hatcheries, 58 per km^2 adults of natural origin, and 4 per km^2 adults from hatcheries.

3 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. "Action area" means all areas to be affected directly or indirectly by the Federal "action" and not merely the immediate area involved in the action. 50 C.F.R. §402.02 (see Section 4 for a description of the action area).

This opinion addresses three corresponding actions conducted by the Navy and NMFS's Permits Division: (1) the Navy's military training and testing activities (i.e., readiness activities) conducted in the NWTT Study Area; (2) NMFS's Permits Division's promulgation of regulations pursuant to the MMPA governing the Navy's "take" of marine mammals incidental to the Navy's military readiness activities from November 2020 through November 2027; and (3) NMFS's Permits Division's issuance of an LOA pursuant to the regulations that authorize the U.S. Navy to "take" marine mammals incidental to military readiness activities in the NWTT Study Area through November 2027.

The Navy proposes to conduct military readiness training and testing ("testing" includes research, development, testing, and evaluation) activities in the NWTT Study Area. These military readiness activities include the use of active sonar and explosives within established operating and warning areas and are representative of training and testing the Navy has been conducting in the NWTT Study Area for decades.

The Permits Division proposes to promulgate regulations pursuant to the MMPA, as amended (16 U.S.C. 1361 et seq.) for the Navy to "take" marine mammals incidental to NWTT activities from November 2020 to November 2027. The regulations propose to authorize the issuance of a LOA that will allow the Navy to "take" marine mammals incidental to their training and testing activities. This consultation considers the MMPA regulations for the Navy to "take" marine mammals incidental to NWTT activities, as modified during ESA consultation.

NMFS recognizes that while Navy training and testing requirements change over time in response to global or geopolitical events and other factors, the general types and tempo of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assumed that the training and testing activities proposed by the Navy during the seven year period of NMFS' proposed incidental take authorization pursuant to the MMPA would continue into the reasonably foreseeable future at levels similar to those assessed in this opinion.

For the training activities considered during consultation, Naval personnel (Sailors and Marines) first undergo entry-level (or schoolhouse) training, which varies according to their assigned warfare community (aviation, surface warfare, submarine warfare, and expeditionary warfare) and the community's unique requirements (Navy 2020e). Personnel then train within their warfare community at sea in preparation for deployment. For the testing activities, the Navy

researches, develops, tests, and evaluates new platforms, systems, and technologies, collectively known as testing. Many tests require realistic conditions at sea and can range from testing new software to complex operations of multiple systems and platforms. Testing activities may occur independent of or in conjunction with training activities (Navy 2020e).

The sections below (Sections 3.1 through 3.3) provide greater detail on the Navy's proposed training and testing activities in the NWTT Study Area. The NMFS Permits Division proposes to promulgate regulations pursuant to the MMPA for the Navy to "take" marine mammals incidental to these activities. We present information on the locations where activities are proposed to occur, describe the specific types of activities proposed, and present information on the levels of activities proposed in the different locations. We then present information on the standard operating procedures (Section 3.4) and mitigation measures (i.e. conservation measures to protect and conserve listed species) (Section 3.5) that will be implemented by the Navy as part of the training and testing activities. We conclude this section by describing the NMFS Permits Division's action under the authority of the MMPA. The primary sources of information for this section were the Navy's NWTT Phase III BA (Navy 2020e), the NWTT DEIS/OEIS (Navy 2019a), and NMFS' proposed rule for its promulgation of regulations and issuance of a letter of authorization pursuant to the MMPA for the U.S. Navy (NMFS 2020e).

3.1 Primary Mission Areas

The Navy categorizes its activities into functional warfare areas called primary mission areas. Activities occurring within the NWTT Study Area generally fall into the following five primary mission areas:

- Air warfare
- Anti-submarine warfare
- Electronic warfare
- Mine warfare
- Surface warfare

Most activities proposed by the Navy are categorized into one of these primary mission areas, though the testing community has three additional categories of activities for vessel evaluation, unmanned systems, and acoustic and oceanographic science and technology. Activities that do not fall within these areas are listed as "other activities" below. Each warfare community (surface, subsurface, and aviation) may train in some or all of these primary mission areas. The research and acquisition community also categorizes most, but not all, of its testing activities under these primary mission areas.

A more detailed description of the sonar, munitions, targets, systems and other material used during training and testing activities within these primary mission areas is provided in Appendix A (Navy Activity Descriptions) of the NWTT DEIS/OEIS (Navy 2019a).

3.1.1 Air Warfare

The mission of air warfare is to destroy or reduce enemy air and missile threats (including unmanned airborne threats). Aircraft conduct air warfare through radar search, detection, identification, and engagement of airborne threats. Surface ships conduct air warfare through an array of modern anti-aircraft weapon systems such as aircraft detecting radar, naval guns linked to radar-directed fire-control systems, surface-to-air missile systems, and radar-controlled cannons for close-in point defense.

3.1.2 Anti-Submarine Warfare

The mission of anti-submarine warfare is to locate, neutralize, and defeat hostile submarine forces that threaten Navy forces. Anti-submarine warfare is based on the principle that surveillance and attack aircraft, ships, and submarines all search for hostile submarines. These forces operate together or independently to gain early warning and detection and to localize, track, target, and attack submarine threats.

Anti-submarine warfare training addresses basic skills such as detection and classification of submarines, as well as evaluating sounds to distinguish between enemy submarines and friendly submarines, ships, and marine life. More advanced training integrates the full spectrum of anti-submarine warfare from detecting and tracking a submarine to attacking a target using either exercise torpedoes (i.e., torpedoes that do not contain a warhead) or simulated weapons. These integrated anti-submarine warfare training exercises are conducted in coordinated, at-sea training events involving submarines, ships, and aircraft.

Testing of anti-submarine warfare systems is conducted to develop new technologies and assess weapon performance and operability with new systems and platforms, such as unmanned systems. Testing uses ships, submarines, and aircraft to demonstrate capabilities of torpedoes, missiles, countermeasure systems, and underwater surveillance and communications systems. Tests may be conducted as part of a large-scale fleet training event involving submarines, ships, fixed-wing aircraft, and helicopters. These integrated training events offer opportunities to conduct research and acquisition activities and to train aircrew in the use of new or newly enhanced systems during a large-scale, complex exercise.

3.1.3 Electronic Warfare

The mission of electronic warfare is to degrade the enemy's ability to use electronic systems, such as communication systems and radar, and to confuse or deny them the ability to defend their forces and assets. Electronic warfare is also used to detect enemy threats and counter their attempts to degrade the electronic capabilities of the Navy. Typical electronic warfare training activities include threat avoidance, signals analysis for intelligence purposes, and use of airborne and surface electronic jamming devices to defeat tracking and communications systems.

Testing of electronic warfare systems is conducted to improve the capabilities of systems and ensure compatibility with new systems. Testing involves the use of aircraft, surface ships, and submarine crews to evaluate the effectiveness of electronic systems. Similar to training activities, typical electronic warfare testing activities include the use of airborne and surface electronic jamming devices, including testing chaff and flares, to defeat tracking and communications systems.

3.1.4 Mine Warfare

The mission of mine warfare is to detect, classify, and avoid or neutralize (disable) mines to protect Navy ships and submarines and to maintain free access to ports and shipping lanes. Mine warfare also includes offensive mine laying to gain control of or deny the enemy access to sea space. Naval mines can be laid by ships, submarines, Navy divers, or aircraft. Mine warfare neutralization training includes exercises in which ships, aircraft, submarines, underwater vehicles, unmanned vehicles, or marine mammal detection systems search for mine shapes. Personnel train to destroy or disable mines by attaching underwater explosives to or near the mine or using remotely operated vehicles to destroy the mine.

Testing and development of mine warfare systems is conducted to improve sonar, laser, and magnetic detectors intended to locate, and record the positions of mines for avoidance or subsequent neutralization. Mine warfare testing and development falls into two primary categories: mine detection and classification, and mine countermeasure and neutralization. Mine detection and classification testing involve the use of air, surface, and subsurface vessels and uses sonar, including towed and side-scan sonar, and unmanned vehicles to locate and identify objects underwater. Mine detection and classification systems are sometimes used in conjunction with a mine neutralization system. Mine countermeasure and neutralization testing include the use of air, surface, and subsurface units to evaluate the effectiveness of tracking devices and countermeasure and neutralization systems to neutralize mine threats. Most neutralization tests use mine shapes, or non-explosive practice mines, to evaluate a new or enhanced capability.

A small percentage of mine warfare tests require the use of high-explosive mines to evaluate and confirm the ability of the system to neutralize a high-explosive mine under operational conditions. The majority of mine warfare systems are deployed by ships, helicopters, and unmanned vehicles. Tests may also be conducted in support of scientific research to support these new technologies.

3.1.5 Surface Warfare

The mission of surface warfare is to obtain control of sea space from which naval forces may operate and entails offensive action against other surface, subsurface, and air targets while also defending against enemy forces. In surface warfare, aircraft use cannons, air-launched cruise missiles, or other precision-guided munitions; ships employ torpedoes, naval guns, and surface-

to-surface missiles; and submarines attack surface ships using torpedoes or submarine-launched, anti-ship cruise missiles.

Surface warfare training includes surface-to-surface gunnery and missile exercises, air-to-surface gunnery and missile exercises, and submarine missile or torpedo launch events, and other munitions against surface targets. Testing of weapons used in surface warfare is conducted to develop new technologies and to assess weapon performance and operability with new systems and platforms, such as unmanned systems. Tests include various air-to-surface guns and missiles, surface-to-surface guns and missiles, and bombing tests. Testing events may be integrated into training activities to test aircraft or aircraft systems in the delivery of ordnance on a surface target. In most cases the tested systems are used in the same manner in which they are used for fleet training activities.

3.2 Classification of Sonar and Explosive Sources into Bins

The Navy developed a series of source classifications, or source bins, in order to better organize and facilitate the analysis of, and implementation of mitigation for, approximately 300 individual sources of underwater sound deliberately employed by the Navy including sonars, other transducers (devices that convert energy from one form to another—in this case, to sound waves), and explosives. Non-impulsive sources are grouped into bins based on the frequency, source level when warranted, and how the source would be used. Low-frequency (LF) sources operate below 1 kHz; mid-frequency (MF) sources operate at or above 1 kHz, up to and including 10 kHz; high-frequency (HF) sources operate above 10 kHz, up to and including 100 kHz; and very high-frequency (VHF) sources operate above 100 kHz, but below 200 kHz. Impulsive bins are based on the NEW of the munitions or explosive devices.

Sonar source bins are described in Table 18, along with a comparison of the maximum annual activity levels between ongoing activities (NWTT Phase II) and the proposed action (NWTT Phase III). In addition to the acoustic sources described above, there are other in-water, active acoustic sources from NWTT activities that were not quantitatively analyzed using NAEMO.

Table 18. Description of Navy sonar source bins and comparison of maximum annual activity levels by bin between ongoing (Phase II) activities and the proposed action (Phase III) (Navy 2020e).

Source Class Category	*Bin*	*Description*	*Unit*[1]	*Training*		*Testing*	
				Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Low-Frequency (LF): Sources that produce signals less than 1 kHz	LF4	LF sources equal to 180 dB and up to 200 dB	H	0	0	110	177
	LF5	LF sources less than 180 dB	H	0	1	71	0–18
Mid-Frequency (MF): Tactical and nontactical sources that produce signals at or above 1 kHz up to and including 10 kHz	MF1	Hull-mounted surface ship sonars (e.g., AN/SQS-53C and AN/SQS-60)	H	166	164	32	20–169
	MF1K	Kingfisher mode associated with MF1 sonars	H	0	0	0	48
	MF2[2]	Hull-mounted surface ship sonars (e.g., AN/SQS-56)	H	0	0	0	32
	MF3	Hull-mounted submarine sonars (e.g., AN/BQQ-10)	H	70	70	145	34–36
	MF4	Helicopter-deployed dipping sonars (e.g., AN/AQS-22)	H	4	0-1	10	41-50
	MF5	Active acoustic sonobuoys (e.g., DICASS)	C	896	918–926	273	300-673
	MF6	Active underwater sound signal devices (e.g., MK 84)	C	0	0	12	60–232
	MF8	Active sources (greater than 200 dB) not otherwise binned	H	0	0	40	0
	MF9	Active sources (equal to 180 dB and up to 200 dB) not otherwise binned	H	0	0	1,183	644–959
	MF10	Active sources (greater than 160 dB, but less than 180 dB) not otherwise binned	H	0	0	1,156	886
	MF11	Hull-mounted surface ship sonars with an active duty cycle greater than 80%	H	16	16	34	48
	MF12	Towed array surface ship sonars with an active duty cycle greater than 80%	H	0	0	24	100
High-Frequency (HF): Tactical and non-tactical sources that produce signals greater than 10	HF1	Hull-mounted submarine sonars (e.g., AN/BQQ-10)	H	48	48	161	10
	HF3	Other hull-mounted submarine sonars (classified)	H	0	0	145	1–19

Source Class Category	*Bin*	*Description*	*Unit*[1]	*Training*		*Testing*	
				Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
kHz up to and including 100 kHz	HF4	Mine detection, classification, and neutralization sonar (e.g., AN/SQS-20)	H	384	0–65	0	1,860–1,868
	HF5	Active sources (greater than 200 dB) not otherwise binned	H	0	0	360	352–400
	HF6	Active sources (equal to 180 dB and up to 200 dB) not otherwise binned	H	192	0	2,099	1,705–1,865
	HF8	Hull-mounted surface ship sonars (e.g., AN/SQS-61)	H	0	0	0	24
	HF9	Weapon-emulating sonar source	H	0	0	0	257
Very High-Frequency (VHF): Tactical and non-tactical sources that produce signals greater than 100 kHz but less than 200 kHz	VHF1	Active sources greater than 200 dB	H	0	0	0	320
	VHF2	Active sources with a source level less than 200 dB	H	0	0	35	135
Anti-Submarine Warfare (ASW): Tactical sources (e.g., active sonobuoys and acoustic countermeasures systems) used during ASW training and testing activities	ASW1	MF systems operating above 200 dB	H	0	0	16	80
	ASW2	MF Multistatic Active Coherent sonobuoy (e.g., AN/SSQ-125)	H	0	0	64	0
			C	720	350	170	240
	ASW3	MF towed active acoustic countermeasure systems (e.g., AN/SLQ-25)	H	78	86	444	487–1,015
	ASW4	MF expendable active acoustic device countermeasures (e.g., MK3)	C	0	0	1,182	1,349–1,389
	ASW5[2]	MF sonobuoys with high duty cycles	H	0	50	0	80
Torpedoes (TORP): Source classes associated with the active acoustic signals produced by torpedoes	TORP1	Lightweight torpedo (e.g., MK 46, MK 54, or Anti-Torpedo Torpedo)	C	0	16	315	298–360
	TORP2	Heavyweight torpedo (e.g., MK 48)	C	0	0–2	299	332–372
	TORP3	Heavyweight torpedo (e.g., MK 48)	C	0	0	0	6
Forward Looking Sonar (FLS): Forward or	FLS2	HF sources with short pulse lengths, narrow beam	H	0	240	0	24

Source Class Category	*Bin*	*Description*	*Unit*[1]	*Training*		*Testing*	
				Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
upward looking object avoidance sonars used for vessel navigation and safety		widths, and focused beam patterns					
Acoustic Modems (M): Systems used to transmit data through the water	M3	MF acoustic modems (greater than 190 dB)	H	0	30	1,519	1,088
Synthetic Aperture Sonars (SAS): Sonars in which active acoustic signals are post-processed to form high-resolution images of the seafloor	SAS2	HF SAS systems	H	0	0–561	798	1,312
Broadband Sound Sources (BB): Sonar systems with large frequency spectra, used for various purposes	BB1	MF to HF mine countermeasure sonar	H	0	0	0	48
	BB2	HF to VHF mine countermeasure sonar	H	0	0	0	48
Swimmer Detection Sonar (SD): Used to detect divers and submerged swimmers	SD1	HF and VHF sources with short pulse lengths, used for the detection of swimmers and other objects for the purpose of port security	H	0	0	757	0

[1] H = hours; C = count (e.g., number of individual pings or individual sonobuoys)

[2] Formerly ASW2 (H) in 2015 BO.

Notes: dB = decibel(s), kHz = kilohertz

Orange cells indicate an increase in hours or counts, and Green cells indicate a decrease in hours or counts.

Table 19. Sonar and transducer sounds sources qualitatively analyzed (Navy 2020e).

Source Class Category	*Bin*	*Characteristics*
Doppler Sonar/Speed Logs (DS): High-frequency/very high-frequency navigation transducers	DS3–DS4	*Required for safe navigation* • downward focused • narrow angle or beam width • very short pulse lengths
Fathometers (FA): High-frequency sources used to determine water depth	FA1–FA4	*Required for safe navigation* • downward focused directly below the vessel • narrow beam width (typically much less than 30°) • short pulse lengths (less than 10 milliseconds)
Imaging Sonar (IMS): Sonars with high or very high frequencies used to obtain images of objects underwater	IMS2–IMS3	• High-frequency or very high-frequency • downward directed • narrow beam width • very short pulse lengths (typically 20 milliseconds)
High-Frequency Acoustic Modems (M): Systems that send data underwater **Tracking Pingers (P):** Devices that send a ping to identify an object location	M1, M2, M3, M4 P1–P4	• low duty cycles (single pings in some cases) • short pulse lengths (typically 20 milliseconds) • low source levels
Acoustic Releases (R): Systems that ping to release a bottom-mounted object from its housing in order to retrieve the device at the surface	R2	• typically emit only several pings to send release order
Side-Scan Sonars (SSS): Sonars that use active acoustic signals to produce high-resolution images of the seafloor	SSS1–SSS2	• downward-directed beam • short pulse lengths (less than 20 milliseconds)

Notes: ° = degree(s)

Explosive source bins are described in Table 20, along with a comparison of the maximum annual activity levels between ongoing activities (NWTT Phase II) and the proposed action (NWTT Phase III). This table shows the number of explosive items that could be used in any year for training and testing activities. A range of annual bin use indicates that use of that bin is anticipated to vary annually, consistent with the variation in the number of annual activities described above.

In addition to the explosives (E) quantitatively analyzed for impacts to ESA-listed species shown in Table 20, the Navy uses some very small impulsive sources (less than 0.1 pounds NEW), categorized in bin E0, that were not quantitatively analyzed by the Navy for potential exposure to marine mammals and sea turtles. These E0 charges are qualitatively analyzed in our effects

analysis for these species groups. The E0 charges were included in our quantitative analysis of the effects of explosives on ESA-listed fishes.

Table 20. Description of Navy explosive source bins and comparison of the maximum annual number of explosives by bin between ongoing (Phase II) activities and the proposed action (Phase III) (Navy 2020e).

Bin	*Net Explosive Weight[1] (lb.)*	*Example Explosive Source*	*Training*		*Testing*	
			Ongoing	*Proposed Action[2]*	*Ongoing*	*Proposed Action[2]*
E0	< 0.1	LIMPET	18	18	0	0
E1	0.1–0.25	Medium-caliber projectiles	48	60–120	0	8
E2	> 0.25–0.5	Medium-caliber projectiles	0	65–130	0	0
E3	> 0.5–2.5	Explosive Ordnance Disposal Mine Neutralization Training	6	6	72	72
E4	> 2.5–5	Mine Countermeasure and Neutralization	0	0	70	36
E5	> 5–10	Large-caliber projectile	80	56–112	0	0
E7	> 20–60	Mine Countermeasure and Neutralization	0	0	0	5
E8	> 60–100	Lightweight torpedo	0	0	3	4
E10	> 250–500	1,000 pounds bomb	4	0–4	0	0
E11	> 500–650	Heavyweight torpedo	0	0	3	4

Note: Orange cell indicates proposed increase, green cell indicates proposed decrease, and no shading indicates proposed no change from on-going activities.

[1] Net Explosive Weight refers to the amount of explosives; the actual weight of a munition may be larger due to other components.

[2] Expected annual use may vary per bin because the number of events may vary from year to year.

3.3 Proposed Training and Testing Activities

The Navy has been conducting military readiness activities in the NWTT Study Area for decades. The tempo and types of training and testing activities have fluctuated because of the introduction of new technologies, the evolving nature of international events, advances in warfighting doctrine and procedures, and changes in force structure (organization of ships, weapons, and personnel). Such developments influence the frequency, duration, intensity, and location of required training and testing activities. The types and numbers of activities proposed by the Navy reflect the most up-to-date compilation of training and testing activities deemed necessary to accomplish military readiness requirements and account for fluctuations in training and testing in order to meet evolving or emergent military readiness requirements. The proposed training and testing activities are detailed in the following sections.

3.3.1 Training Activities

The Navy proposes to conduct military readiness training activities into the reasonably foreseeable future, as necessary to meet current and future readiness requirements. These military readiness training activities include new activities, as well as activities that are currently ongoing and have historically occurred in the NWTT Study Area. For the purposes of this consultation and for the proposed MMPA rule, the Navy identified the number and duration of training activities that could occur over any seven year period, beginning in November 2018. The proposed activity levels consider fluctuations in training cycles and deployment schedules that do not follow a traditional annual calendar but instead are influenced by in-theater demands and other external factors.

Table 21 (below) provides a summary of NWTT training activities. These tables include the event location, source bins used, number of events per year, and maximum levels of sonar (hours) and ordnance used (by bin type), if any. They also compare ongoing maximum annual NWTT Phase II activity levels with the Navy's proposed activity levels for NWTT Phase III (note: green shading indicates decrease from previous levels; orange shading indicates increase from previous levels; no shading indicates no proposed change from on-going activities).

Table 21. Navy proposed training activities for NWTT Phase III (Navy 2020e).

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
Air Warfare								
Air Combat Maneuver[2] (ACM)	None	Offshore Area (W-237)	550[2]	None	126[2]	None	None	Occurs year-round, 100% daytime in W-237, 96% day and 4% night in the Olympic MOA. 232 sorties per year in W-237, 808 sorties per year in Olympic MOA. For flights over land in the Olympic MOA, minimum aircraft altitude is 6,000 feet above sea level, but never less than 1,200 feet above ground level to account for variations in terrain. 70% of all Navy flights in the MOA are above 20,000 feet above sea level and 95% of all activities are above 10,000 ft.
		Offshore Area (Olympic MOA)			574[2]	None		
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A])	None3	Offshore Area (W-237)	160	Lg-cal rounds 230 explosive	125	Lg-cal rounds 0 explosive	None.	Year-round, day only. All large-caliber non-explosive events occur greater than 20 nm from shore, and all other non-explosive rounds are used 12 nm or greater from shore. The target is a fiberglass finned target that is towed approximately 3 nm behind the towing aircraft, at an altitude of 1,000 feet or greater.
				Lg-cal rounds 80 NEPM		Lg-cal rounds 80 NEPM	None	
				Med-cal rounds (6,320 explosive, 9,672 NEPM)		Med-cal rounds (9,660 NEPM)	None.	

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	None	Offshore Area (W-237)	24	AIM-7/9/120 (15 explosive, 15 NEPM)	0–4	AIM-7/9/120 (4 explosive, 4 NEPM)	4 in-air detonations [3]	Year-round, day only. 4 Training events per year with 4 high explosive (HE) warheads, 4 non-explosive practice munitions (NEPM) warheads. Assume 1 flare per Missile Exercise event. All events occur greater than 50 nm from shore and above 15,000 feet altitude. All propellant and explosives are consumed.
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	None	Offshore Area (W-237)	4	RIM-7/116 (8 explosive)	0–4	RIM-7/116 (8 explosive)	8 in-air detonations [3]	Year-round, day only. Assumes that all surface-to-air missiles are high explosive. All events occur greater than 50 nm from shore and missile explosions occur above 500 feet altitude. All explosive and propellant are consumed.
Anti-submarine Warfare								
Torpedo Exercise – Submarine (TORPEX – Sub)	Acoustic	Offshore Area	Not previously Analyzed[4]		0–2	2 MK-48 Torpedoes (non-explosive)	TORP2 (2 torpedoes) MF3 (1 hour) HF1 (0.5 hours)	Exercise non-explosive practice torpedoes are recovered. Guidance wire has a low breaking strength and breaks easily. Weights and flex tubing sink rapidly. All events would occur 12 nm or more from shoreline.

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[(1)]	Max. no. of ordnance per year		
Tracking Exercise – Helicopter (TRACKEX – Helo)	Acoustic	Offshore Area	4	None	0–2	None	MF4 (0.5 hours) MF5 (8 sonobuoys)	Year-round, day and night. Submarines may provide service as the target. All events are conducted in water greater than 600 feet in depth and further than 12 nm from shore. Notes: Sonar reduced from 4 hours MF4 and 16 MF5 sonobuoys
Tracking Exercise – Maritime Patrol Aircraft (TRACKEX – MPA)	Acoustic	Offshore Area	324	None	373	16 Torpedoes (non-explosive)	ASW2 (350 sonobuoys) ASW5 (50 hours) MF5 (918 sonobuoys) TORP1 (16 torpedoes)	Year-round, day and night. Submarine may provide service as the target. If target is air dropped, one parachute per target. All events are conducted in water greater than 600 feet in depth and further than 12 nm from shore. Notes: Sonar decreased from 720 ASW2 sonobuoys; increased from 0 ASW5 hours, 880 MF5 sonobuoys, and 0 TORP1 torpedoes
Tracking Exercise – Ship (TRACKEX – Ship)	Acoustic	Offshore Area	65	None	62	None	ASW3 (86 hours) HF6 (0 hours) MF1 (117 hours) MF11 (16 hours)	Year-round, day and night. A submarine may provide service as the target. All events are conducted in water greater than 600 feet in depth and further than 12 nm from shore. Notes: Sonar increased from 78 ASW3 hours; decreased from 80 HF6 hours, and 140 MF1 hours

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
Tracking Exercise – Submarine (TRACKEX – Sub)	Acoustic	Offshore Area	100	None	75–100	None	HF1 (45.5 hours) HF6 (0 hours) MF3 (47 hours)	Year-round, day and night. All events are conducted in water greater than 600 feet in depth and further than 12 nm from shore. Notes: Sonar reduced from 112 HF6 hours
Electronic Warfare								
Electronic Warfare Training – Aircraft (EW Training)[2]	None	Offshore Area (W-237)	1,062[5]	None	1,062[5]	None	None	Year-round, day and night, 2,172 aircraft sorties per year.
		Offshore Area (Olympic MOA)	3,938[5]		3,938			
Electronic Warfare Training – Ship (EW Training)	None	Offshore Area (W-237), Inland Waters	275	None	220	None	None	Year-round, day and night.
Mine Warfare								
Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises	Acoustic	Inland Waters (NS Everett, NBK Bangor, NBK Bremerton, Manchester Fuel Pier, Port Angeles, Port of Seattle)	Every other year (three in 5 years)	None	0–1	None	HF4 (65 hours) SAS2 (561 hours)	Year-round, day and night. Non-permanent mine shapes will be laid in various places on the bottom and will be retrieved. Shapes are varied, from about 1 m circular to about 2.5 m long by 1 m wide. They will be recovered using normal assets, with diver involvement. Assume one event will occur every other year with a total of three per 5-year period. Notes: Sonar reduced from 384 HF4 hours, increased from 0 SAS2 hours
Mine Neutralization –	Explosive	Inland Waters (Crescent	3	3, 2.5-lb. charges	3	3, 2.5-lb. charges	Up to E3 (total of 3)	Mine shapes will be recovered. Year-round, day only.

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
Explosive Ordnance Disposal (EOD) Training		Harbor EOD Training Range)	3	18, <0.1 pounds charges	3	18, <0.1lb. charges	Up to 18 E0 explosive charges	Year-round, day only.
		Inland Waters (Hood Canal EOD Training Range)	3	3, 2.5-lb. charges	3	3, 2.5-lb. charges	Up to E3 (total of 3)	Mine shapes will be recovered. Year-round, day only.
			3	18, <0.1 pounds charges	3	18, <0.1 pounds charges	Up to 18 E0 explosive charges	Year-round, day only.
Submarine Mine Exercise	None	Offshore Area	8	None	None	None	None	Discontinued Activity.
Surface Warfare								
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Explosive	Offshore Area (W-237)	30	BDU-45, MK-84 bombs (10 explosive, 110 NEPM)	0–28	BDU-45 bombs (84 NEPM)	None.	Year-round, day only. Approximately 90 percent of non-explosive bombs are the sub-scale bombs such as the MK-76 and BDU-48. Non-explosive bombing events occur greater than 20 nm from shore. Air-to-surface bombing is not authorized in the Olympic Coast National Marine Sanctuary.
					0–2	MK-80 bombs (2 explosive)	E10 [2]	Year-round, day only. All explosive bombing events occur greater than 50 nm from shore. Air-to-surface bombing is not authorized in the Olympic Coast National Marine Sanctuary.
Gunnery Exercise (Surface-to-Surface) – Ship	Explosive	Offshore Area	200	Sm-cal rounds (121,200 NEPM)	100–200	Sm-cal rounds (121,000 NEPM)	None	Year-round, day only. Small- and medium-caliber NEPM activity always

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
(GUNEX [S-S] – Ship)				Med-cal rounds (48 explosive)	(up to 34 explosive events)	Med-cal rounds (250 explosive)	E1 (120 rounds) E2 (130 rounds)	occurs 12 nm or more from the shoreline. Large-caliber NEPM activity always occurs 20 nm or more from shoreline. Medium- and large-caliber explosive munitions activity always occur 50 nm or more from shore. For analytical purposes assume all high explosive rounds are fused to detonate upon impact with water surface or target. After impacting the water, the high explosive rounds are expected to detonate within 3 feet of the surface. Non-explosive rounds and fragments from the high explosive rounds will sink to the bottom of the ocean.
				33,492 NEPM		16,750 NEPM	None	
				Lg-cal rounds 80 explosive		Lg-cal rounds 112 explosive	E5 (112 rounds)	
				2,720 NEPM		2,720 NEPM		
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	Explosive	Offshore Area (W-237)	4	AGM-84 (4 explosive missiles)	0–2	AGM-84 (2 explosive missiles)	E10 (2 missiles)	Year-round, day only. Assume one target per event. Most missiles are non-firing. Some missiles are live missiles with HE warhead (2 HE missiles per year). All events occur greater than 50 nm from shore.
Other Training								

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[(1)]	Max. no. of ordnance per year		
Intelligence, Surveillance, Reconnaissance (ISR)	None	Offshore Area	200	None	200	None	None	Year-round, day and night. ISR training is conducted by Maritime Patrol Aircraft and unmanned aerial systems in W-237 and the Pacific Northwest Operating Area. Activities typically last 6 hours. P-8A aircrews use a variety of intelligence gathering and surveillance methods, including visual, infrared, electronic, radar, and acoustic. EP-3 and EA-18G crews conduct ISR training as well, but to a lesser extent than P-8A crews.
Maritime Security Operations	None	Inland Waters (NBK Bremerton, Hood Canal/DBRC, NS Everett, Puget Sound, Strait of Juan de Fuca)	286	1,320 small-caliber rounds (all blanks)	220	1,320 small-caliber rounds (all blanks)	None	Year-round, day only.
Personnel Insertion/ Extraction – Non-Submersible	None	Inland Waters (Crescent Harbor EOD Range, NAVY 7 OPAREA)	10	None	6	None	None	Year-round, day and night.
Personnel Insertion/ Extraction – Submersible	None	Inland Waters	35	None	0 (6)	None	None	
Precision Anchoring	None	Inland Waters (see Figure 2-3 of (Navy 2020e))	10	None	30–40	None	None	Year-round, day and night.

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[(1)]	Max. no. of ordnance per year		
Search and Rescue	None	Inland Waters (Crescent Harbor EOD Range, R-6701)	100	None	80	None	None	Year-round, day and night. This activity involves a helicopter landing and simulated extraction of a survivor. The search and rescue helicopter, an H-60, approaches the survivor, hovers, recovers the survivor, and then departs the area with the survivor onboard.
Small Boat Attack Exercise	None	Inland Waters (NS Everett, NBK Bangor, NBK Bremerton)	1	None	1	None	None	Year-round, day only. At locations where a security barrier is present, and sea lions may be hauled out on the barrier, the security barrier will be pulled fully open to remove haul out opportunities. During Day 1 training, all firing will occur at least 250 feet away from the security barrier.
Submarine Sonar Maintenance	Acoustic	Inland Waters (NBK Bangor, NBK Bremerton), Offshore Area	22	None	26	None	LF5 (1 hour) MF3 (22 hours; 20 hours Inland Waters, 2 hours Offshore Area) HF1 (2 hours Offshore Area)	Year-round, day and night.
Surface Ship Sonar Maintenance	Acoustic	Inland Waters (NBK Bremerton, NS Everett), Offshore Area	13	None	25	None	MF1 (47 hours total; 44 hours Offshore Area, 3 hours Inland Waters)	Year-round, day and night.

Range activity	In-water acoustic/ explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events per year[1]	Max. no. of ordnance per year		
Unmanned Underwater Vehicle Training	Acoustic	Inland Waters (Crescent Harbor EOD Range, DBRC, NBK Bangor, NBK Bremerton, Kitsap Range Site, Manchester Fuel Pier, NAVY 3 & 7 OPAREAs) Offshore Area (QRS)	Not previously analyzed as a training activity [7]	None	60	None	FLS2 (240 hours) M3 (30 hours)	2 events in the Offshore Area; 58 events at the Inland Waters. Potential specific locations for this activity include Northwest Training Range Complex Dabob Bay Range Complex, Hood Canal Sinclair Inlet, NBK Bangor, NBK Keyport, and Manchester Fuel Pier.

[1]For activities where the maximum number of events varies between years, a range is provided to indicate the "representative–maximum" number of events. For activities where no variation is anticipated, only the maximum number of events within a single year is provided.
[2]These events typically involve two aircraft; however, based upon the training requirement, events may involve more than two aircraft.
[3] In-air detonations only.
[4]The TORPEX – SUB activity was analyzed in 2015 as part of the Sinking Exercise. The Sinking Exercise is no longer conducted in the NWTT Study Area, and the TORPEX – SUB activity is now a separate activity.
[5]Multiple Air Combat Maneuver and Electronic Warfare aircraft events occur during a single aircraft training flight (sortie). On average, two events occur per sortie.
[6]This activity is covered under a separate analysis (2018 Final Environmental Assessment for Naval Special Operations in Western Washington State).
[7]Unmanned underwater vehicles were analyzed in 2015 as a testing activity.
Note: Orange cell indicates proposed increase, green cell indicates proposed decrease, and no shading indicates proposed no change from on-going activities.
Acronyms: EOD = Explosive Ordnance Disposal; med-cal = medium caliber; MOA = Military Operations Area; NBK = Naval Base Kitsap; NEPM = Non-Explosive Practice Munitions; NS = Naval Station; oz. = ounce(s); QRS = Quinault Range Site; R- = Restricted Area; sm-cal = small caliber.

3.3.2 Testing Activities

The Navy's research and acquisition community engages in a broad spectrum of testing activities in support of the fleet. These activities include, but are not limited to, basic and applied scientific research and technology development; testing, evaluation, and maintenance of systems (missiles, radar, and sonar), and platforms (surface ships, submarines, and aircraft); and acquisition of systems and platforms to support Navy missions and give a technological edge over adversaries.

The Navy operates in an ever-changing strategic, tactical, funding, and time-constrained environment. Testing activities occur in response to emerging science or fleet operational needs. Following identification of future needs, new systems are developed or existing systems are modified. These systems—whether new or modifications of existing systems—must be tested in the field to ensure they meet fleet needs and requirements.

Some testing activities are similar to training activities conducted by the fleet. For example, both the fleet and the research and acquisition community fire "test" torpedoes. While the firing of a torpedo might look identical to an observer, the difference is in the purpose of the firing. The fleet might fire the torpedo to practice the procedures for such a firing, whereas the research and acquisition community might be assessing a new torpedo guidance technology or ensuring that the torpedo meets performance specifications and operational requirements.

3.3.2.1 Naval Sea Systems Command Testing Activities

Naval Sea Systems Command activities are generally aligned with the primary mission areas (Section 3.1) used by the fleets. Additional activities include, but are not limited to, vessel evaluation, unmanned systems, and other testing activities. Testing activities are conducted throughout the life of a Navy ship, from construction through deactivation from the fleet, as part of verification of performance and mission capabilities. Activities include pierside and at-sea testing of ship systems, including sonar, acoustic countermeasures, radars, weapons, unmanned systems, and radio equipment; tests to determine how the ship performs at sea; development and operational testing and evaluation programs for new technologies and systems; and testing on all ships and systems that have undergone overhaul or maintenance. Table 22 provides descriptions of the Naval Sea Systems Command testing activities included in the NWTT Study Area.

Table 22. Representative Naval Sea Systems Command testing activities.

Activity Name	*Activity Description*	*2015 NWTT Final EIS/OEIS Activity Name*
Anti-Submarine Warfare		
Anti-Submarine Warfare Testing	Ships and their supporting platforms (rotary-wing aircraft and unmanned aerial systems) detect, localize, and prosecute submarines.	Anti-Submarine Warfare Mission Package Testing
		Anti-Submarine Warfare Testing
At-Sea Sonar Testing	At-sea testing to ensure systems are fully functional in an open ocean environment.	*[Similar activity previously analyzed in the 2015 NWTT Final EIS/OEIS under Training (Table 2.8-1: Tracking Exercise – Surface)]*
Countermeasure Testing	Countermeasure testing involves the testing of systems that will detect, localize, and track incoming weapons, including marine vessel targets. Countermeasures may be systems to obscure the vessel's location or systems to rapidly detect, track, and counter incoming threats. Testing includes surface ship torpedo defense systems and marine vessel stopping payloads.	Countermeasures Testing
Pierside-Sonar Testing	Pierside testing to ensure systems are fully functional in a controlled pierside environment prior to at-sea test activities.	Pierside-Sonar Testing
Submarine Sonar Testing/Maintenance	Pierside, moored, and underway testing of submarine systems occurs periodically following major maintenance periods and for routine maintenance.	Project Operations (POPS)
Torpedo (Explosive) Testing	Air, surface, or submarine crews employ explosive and non-explosive torpedoes against artificial targets.	Torpedo (Explosive) Testing
Torpedo (Non-explosive) Testing	Air, surface, or submarine crews employ non-explosive torpedoes against targets, submarines, or surface vessels.	Torpedo (Non-explosive) Testing
Mine Warfare		
Mine Countermeasure and Neutralization Testing	Air, surface, and subsurface vessels neutralize threat mines and mine-like objects.	*[Not previously analyzed]*
Mine Detection and Classification Testing	Air, surface, and subsurface vessels and systems detect and classify mines and mine-like objects. Vessels also assess their potential susceptibility to mines and mine-like objects.	Side Scan/Multibeam Sonar

Activity Name	*Activity Description*	*2015 NWTT Final EIS/OEIS Activity Name*
Surface Warfare		
Kinetic Energy Weapon Testing	A kinetic energy weapon uses stored energy released in a burst to accelerate a projectile.	*[Not previously analyzed]*
Unmanned Systems		
Unmanned Aerial System Testing	Unmanned aerial systems are remotely piloted or self-piloted (i.e., preprogrammed flight pattern) aircraft that include fixed-wing, rotary-wing, and other vertical takeoff vehicles. They can carry cameras, sensors, communications equipment, or other payloads.	Unmanned Aircraft System
Unmanned Surface Vehicle System Testing	Unmanned surface vehicles are primarily autonomous systems designed to augment current and future platforms to help deter maritime threats. They employ a variety of sensors designed to extend the reach of manned ships.	Unmanned Surface Vehicle Testing
Unmanned Underwater Vehicle Testing	Testing involves the production or upgrade of unmanned underwater vehicles. This may include testing of mission capabilities (e.g., mine detection), evaluating the basic functions of individual platforms, or conducting complex events with multiple vehicles.	Unmanned Underwater Vehicle Testing
		Unmanned Vehicle Development and Payload Testing
		Performance Testing at Sea
		Proof of Concept Testing
		Development Training and Testing
Vessel Evaluation		
Propulsion Testing	Ship is run at high speeds in various formations and at various depths.	*[Not previously analyzed]*
Undersea Warfare Testing	Ships demonstrate capability of countermeasure systems and underwater surveillance, weapons engagement, and communications systems. This tests ships' ability to detect, track, and engage undersea targets.	*[Not previously analyzed]*
Vessel Signature Evaluation	Surface ship, submarine, and auxiliary system signature assessments. This may include electronic, radar, acoustic, infrared and magnetic signatures.	Electromagnetic Measurement
		Surface Vessel Acoustic Measurement Testing
		Underwater Vessel Acoustic Measurement Testing

Activity Name	*Activity Description*	*2015 NWTT Final EIS/OEIS Activity Name*
Other Testing		
Acoustic and Oceanographic Research	Research using active transmissions from sources deployed from ships, aircraft, and unmanned underwater vehicles. Research sources can be used as proxies for current and future Navy systems.	*[Not previously analyzed]*
Acoustic Component Testing	Various surface vessels, moored equipment, and materials are tested to evaluate performance in the marine environment.	Pierside Acoustic Testing
		Component System Testing
Cold Water Support	Fleet training for divers in a cold water environment, and other diver training related to Navy divers supporting range/test site operations and maintenance.	Cold Water Training
Hydrodynamic and Maneuverability Testing	Submarines maneuver in the submerged operating environment.	Underwater Vessel Hydrodynamic Performance Measurement
Non-Acoustic Component Testing	These tests involve non-acoustic sensors and communication systems. Non-acoustic sensors may also gather other forms of environmental data.	Non-Acoustic Tests
Post-Refit Sea Trial	Following periodic maintenance periods or repairs, sea trials are conducted to evaluate submarine propulsion, sonar systems, and other mechanical tests.	Post-Refit Sea Trial
Radar and Other System Testing	Testing may include use of military or commercial radar, communication systems (or simulators), or high-energy laser weapons. Testing may occur aboard a ship, helicopter, manned or unmanned underwater vehicle against drones, small boats, or other targets.	*[High-energy laser testing not previously analyzed]*
Semi-Stationary Equipment Testing	Semi-stationary equipment (e.g., hydrophones) is deployed to determine functionality.	Measurement System Repair and Replacement
		Target Strength Trial
		Acoustic Test Facility
Simulant Testing	The capability of surface ship defense systems to detect and protect against chemical and biological attacks are tested.	*[Not previously analyzed]*

3.3.2.2 Naval Air Systems Command Testing Activities

Naval Air Systems Command testing events generally fall into the primary mission areas used by the fleets. Naval Air Systems Command activities include, but are not limited to, the testing of new aircraft platforms, weapons, and systems before those platforms, weapons and systems are delivered to the fleet. The majority of testing and development activities (Table 18) conducted by Naval Air Systems Command are similar to fleet training activities, and many platforms (e.g., Maritime Patrol Aircraft) and systems (e.g., sonobuoys) currently being tested are already being used by the fleet or will ultimately be integrated into fleet training activities. However, some testing and development activities may be conducted in different locations and in a different manner than the fleet and therefore, though the potential environmental effects may be the same, the analysis for those activities may differ.

Table 23. Representative Naval Air Systems Command testing activities.

Activity Name	Activity Description
Anti-Submarine Warfare (ASW)	
Tracking Test – Maritime Patrol Aircraft	The test evaluates the sensors and systems used by maritime patrol aircraft to detect and track submarines and to ensure that aircraft systems used to deploy the tracking systems perform to specifications and meet operational requirements.
Tracking Test – Maritime Patrol Aircraft (SUS)	This test evaluates the sensors and systems used by maritime patrol aircraft to communicate with submarines using any of the family of signal underwater sound (SUS) sonobuoy systems.
Other Testing	
ISR/EW MQ-4C Triton UAS Testing	ISR/EW Triton Testing will evaluate the sensors and communication systems on board the MQ 4C Triton unmanned aerial system.

3.3.2.3 Testing Activity Levels

Table 22 (Naval Sea Systems Command) and Table 23 (Naval Air Systems Command) provide summaries of NWTT testing activities. These tables include the event location, source bins used, number of events per year, and maximum levels of sonar (hours) and ordnance used (by bin type), if any. They also compare ongoing maximum annual NWTT Phase II activity levels with the Navy's proposed activity levels for NWTT Phase III (note: green shading indicates decrease from previous levels; orange shading indicates increase from previous levels; no shading indicates no proposed change from on-going activities).

Table 24. Naval Sea Systems Command proposed testing activities for NWTT Phase III (Navy 2020e).

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Anti-Submarine Warfare								
Anti-Submarine Warfare Testing	Acoustic	Offshore Area	13	16 NEPM torpedoes	44	8 NEPM torpedoes	ASW1 (80 hours) ASW2 (40 sonobuoys) ASW3 (80 hours) ASW5 (40 hours) MF1K (48 hours) MF4 (40 hours) MF5 (80 sonobuoys) MF10 (96 hours) MF11 (48 hours) MF12 (80 hours) TORP1 (8 torpedoes)	Daytime testing only.
At-Sea Sonar Testing	Acoustic	Offshore Area	Not previously analyzed as a testing activity	None	4	None	ASW3 (24 hours) HF1 (6 hours) HF5 (24 hours) M3 (96 hours) MF3 (8 hours)	Mostly daytime testing, one test event in this activity may occur 50% at night. Active sonar use is intermittent throughout the duration of the event.

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
		Inland Waters (DBRC)		Not previously analyzed	4 - 6	16–24 NEPM torpedoes	ASW3 (144 hours) HF5 (144 hours) TORP1 (24 torpedoes)	Daytime testing only. Active sonar use is intermittent throughout the duration of the event.
Counter-measure Testing	Acoustic	Offshore Area (QRS)	14	123 NEPM torpedoes	14	12 NEPM torpedoes	ASW3 (24 hours) ASW4 (360 sonobuoys) HF5 (0 hours) HF8 (24 hours) MF1 (4 hours) TORP2 (12 torpedoes)	Up to 50% of testing could occur at night in the Offshore Area and daytime testing only in the Inland Waters. Not all events will include the use of sonar and other transducers; Use of expendable materials is minimized in Inland Waters, and most components of countermeasures are recovered (some components are consumed in use and dissipate in the environment). Obscuring devices deployed in the water may have a self-inflating balloon and tether that helps them to operate at the ideal depth. The balloon allows test units to be recovered in calm conditions, but has a slow leak enabling the empty container to sink to the floor. The tether is a very thin wire or monofilament type material and is an entanglement hazard. No marine vessel stopping testing will occur at Southeast Alaska Acoustic Measurement Facility (SEAFAC). All materials used at SEAFAC would be recovered.
		Inland Waters (DBRC, Keyport Range Site)	74	21 NEPM torpedoes	29	None	ASW3 (24 hours) ASW4 (720 sonobuoys) HF5 (0 hours) TORP1 (0 torpedoes)	
		Western Behm Canal, AK	4	None	1	None	ASW4 (1 sonobuoy)	

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Pierside Sonar Testing	Acoustic	Inland Waters (NS Everett, NBK Bangor, NBK Bremerton)	67	None	88–99	None	ASW3 (1 hour) HF1 (0 hours) HF3 (1 hour) MF1 (6 hours) MF2 (32 hours) MF3 (24 hours) MF9 (69 hours) MF10 (7 hours) MF12 (20 hours)	Dependent on testing event in this activity, range of 0 to 50% of testing may occur at night.
Sub-marine Sonar Testing/ Maintenance	Acoustic	Western Behm Canal, AK	3	None	1–2	None	HF6 (319 hours) MF9 (168 hours)	
Torpedo (Explosive) Testing	Acoustic and Explosive	Offshore Area	3	6 explosive torpedoes 6 NEPM torpedoes	4	8 explosive torpedoes 16 NEPM torpedoes	E8 (4) E11 (4) ASW3 (1 hour) HF1 (2 hours) HF6 (18 hours) MF1 (1 hour) MF3 (1 hour) MF4 (1 hour) MF5 (22 sonobuoys) MF6 (16 sonobuoys) TORP1 (8 torpedoes) TORP2 (8 torpedoes)	Daylight hours only. Only one heavyweight torpedo test could occur in 1 day; two heavyweight torpedo tests could occur on consecutive days. Two lightweight torpedo tests could occur in a single day. All non-explosive torpedoes are recovered. Explosive torpedo testing occurs at least 50 nm from shore and does not occur within the boundaries of the Olympic Coast National Marine Sanctuary.

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Non-explosive Torpedo Testing	Acoustic	Offshore Area	23	119 NEPM torpedoes	22	146 NEPM torpedoes	ASW3 (177 hours) ASW4 (248 sonobuoys) HF1 (2 hours) HF5 (72 hours) HF6 (25 hours) MF1 (9 hours) MF3 (3 hours) MF4 (1 hour) MF5 (118 sonobuoys) MF6 (24 sonobuoys) MF9 (112 hours) MF10 (24 hours) TORP1 (78 torpedoes) TORP2 (68 torpedoes)	A few events within this activity may have nighttime testing up to 50% in the Offshore Area and daytime testing only in the Inland Waters. All exercise torpedoes are recovered. Typically, no more than 8 torpedoes are fired per day during daylight hours.
		Inland Waters (DBRC)	41	189 NEPM torpedoes	61	358 NEPM torpedoes	HF6 (28 hours) LF4 (1 hour) TORP1 (128 torpedoes) TORP2 (224 torpedoes) TORP3 (6 torpedoes)	
Mine Warfare								

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Mine Counter-measure and Neutralization Testing	Acoustic & Explosive	Offshore Area	Not previously analyzed		3	5 mine explosives 36 mine neutralizers	E4 (36) E7 (5) HF4 (225 hours)	Daytime testing only. Testing involving the use of explosives would occur in waters 3 NM or greater from shore at the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 NM or greater from shore elsewhere in the NWTT Offshore Area. Of the 3 proposed events, 2 involve the use of sonar and explosives. The number of explosives shown (i.e., 36 E4 and 5 E7) represent the maximum number used in any given year. The Navy's proposed action also includes a cap on the seven year total number of offshore explosives for this activity as follows: 108 E4 and 15 E7 explosives. Moored mines associated with this activity would be 75 feet or greater from the seafloor. No explosives would be detonated on the bottom (Navy 2020a)
	Acoustic	Inland Waters	Not previously analyzed		3	None	HF4 (675 hours)	Daytime testing only. Explosives are not used in the Inland Waters. Manned aircraft are not used in the Inland Waters except at the designated aircraft-use area Navy 3 OPAREA.
Mine Detection and	Acoustic	Offshore Area (QRS)	Not previously analyzed		1	None	BB1 (16 hours) BB2 (16 hours) LF4 (16 hours)	Daytime testing only. Mine-like targets and temporary anchored devices may be deployed for the duration of a single test

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Classification Testing		Inland Waters (DBRC, Keyport Range Site)	54	None	42	None	BB1 (32 hours) BB2 (32 hours) HF4 (960 hours) LF4 (32 hours)	event or may be left in place for up to 12 months to support multiple events; all devices and their anchors are recovered. Bottom anchors are not deployed in known sensitive shallow water benthic habitats such as eelgrass beds.
Surface Warfare								
Kinetic Energy Weapon Testing	None[3]	Offshore Area	Not previously analyzed		4	80 kinetic energy explosive 160 NEPM large-caliber projectiles	None	Up to 25% of testing could occur at night. Assume one target is expended per event. Explosive rounds are designed to detonate above the surface target. Activity takes place at least 50 nm from shore.
Unmanned Systems								
Unmanned Aerial System Testing	Acoustic	Offshore Area (QRS)	20	None	2	None	None	Daytime testing only. Unmanned aerial systems (UASs) work in compliance with the Federal Aviation Administration regulations UASs can vary in size up to approximately 10 feet in length, with gross vehicle weights of a couple hundred pounds.
		Inland Waters (DBRC, Keyport Range Site, R-6701)	20	None	20	None	None	
Unmanned Surface Vehicle System Testing	None	Offshore Area (QRS)	20	None	4	None	None	Daytime testing only.
		Inland Waters (DBRC, Keyport Range Site)	20	None	20	None	None	

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Unmanned Underwater Vehicle Testing	None	Offshore Area (QRS)	28	27 NEPM torpedoes	38–39	12–24 NEPM torpedoes	FLS2 (8 hours) HF5 (30 hours) TORP1 (24 torpedoes) VHF1 (60 hours)	Dependent of testing event in this activity, range of 0 to 25% (offshore area) or 50% (inland waters) of testing may occur at night. Mine shapes and other temporary anchored targets may be deployed for the duration of a single test event or may be left in place for up to 12 months to support multiple events; all devices and their anchors are recovered. Bottom anchors are not deployed in known sensitive shallow water benthic habitats such as eelgrass beds. Multiple vehicles may operate simultaneously in one or multiple areas.
		Inland Waters (DBRC, Keyport Range Site, Carr Inlet Operations Area)	253	107 NEPM torpedoes	371–379	48–72 NEPM torpedoes	FLS2 (16 hours) HF5 (130 hours) HF9 (20 hours) M3 (256 hours) SAS2 (1,312 hours) TORP1 (72 torpedoes) VHF1 (260 hours)	
Vessel Evaluation								
Propulsion Testing	None	Offshore Area	Not previously analyzed		8–10	None	None	Up to 50% of testing could occur at night. Surface ships will not be conducting test constantly for the entire duration. Surface ships may not be traveling in straight line. Surface ships will operate at least 10 nm from shore, across the full spectrum of capable speeds.

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Undersea Warfare Testing	Acoustic	Offshore Area	Not previously analyzed		1–12	20–78 NEPM torpedoes	ASW3 (540 hours) ASW4 (60 sonobuoys) HF4 (8 hours) MF1 (149 hours) MF4 (9 hours) MF5 (373 sonobuoys) MF6 (172 sonobuoys) MF9 (288 hours) TORP1 (18 torpedoes) TORP2 (60 torpedoes)	Up to 50% of testing could occur at night. Not all sonobuoys used in this activity would include a decelerator/parachute. Ships will not be conducting test constantly during the duration of the allotted time.
Vessel Signature Evaluation	None	Inland Waters (DBRC)	Not previously analyzed		1	None	None	Dependent on testing event in this activity, range of 0 to 50% of testing may occur at night.
Other Testing								
Acoustic and Oceanographic Research	Acoustic	Offshore Area (QRS)	Not previously analyzed		1	None	LF4 (10 hours) MF9 (10 hours)	Daytime testing only.
		Inland Waters (DBRC, Keyport Range Site)	Not previously analyzed		3	None	LF4 (30 hours) MF9 (30 hours)	

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Acoustic Component Testing	Acoustic	Inland Waters (NS Everett, NBK Bangor, NBK Bremerton)	60	None	45	None	LF5 (0 hours) MF8 (0 hours) SD1 (0 hours)	Subject vessel being tested is moored at the Navy piers in Washington, but may be moving or static if the test is conducted at SEAFAC. ROVs may be used to deploy sensors below the water line at the Washington piers, but are unlikely to be used at SEAFAC.
		Western Behm Canal, AK	4	None	13-18	None	HF3 (18 hours) HF6 (274 hours) LF5 (17 hours) MF9 (144 hours)	
Cold Water Support	Acoustic	Offshore Area (QRS)	20	None	0	None	None	If a submarine is used as part of the event (Carr Inlet, SEAFAC), submarine acoustic systems may be activated.
		Inland Waters (Keyport Range Site, DBRC, Carr Inlet Operations Area)	65	None	4	None	HF6 (707 hours)	
		Western Behm Canal, AK	1	None	1	None	HF6 (23 hours)	

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Hydrodynamic and maneuverability testing	None	Western Behm Canal, AK	3	None	1	None	None	For biological resource analysis, vessel noise and vessel strike are only analyzed for the periods while the submarines are surfaced, typically brief in nature. Mitigation measures related to vessel movement are only considered using the period of surfacing as well. For human resource stressor analysis, physical disturbance and strike and physical interactions are only analyzed for the periods (typically brief in nature) while the submarine is surfaced. Underwater communications are used for range and vessel safety purposes.
Non-Acoustic Component Testing	None	Offshore Area	6	None	7–8	None	None	Dependent on testing event in this activity, range of 0 to 50% of testing may occur at night. Manned aircraft are not used in Dabob Bay Range Complex or Keyport Range Site. Underwater communications are used for range and vessel safety purposes. Dependent on testing event in this activity, range of 0 to 50% of testing may occur at night. Unmanned aerial vehicles used in the inland waters areas would be small (e.g., Phantom quadcopter).
		Inland Waters (DBRC, Keyport Range Site, NBK Bangor, Zelatched Point Pier, Keyport Pier)	74	None	75	None	None	
Post-Refit Sea Trial	Acoustic	Inland Waters (DBRC)	32	None	30	None	HF9 (89 hours) M3 (736 hours) MF10 (759 hours)	

Range activity	In-water acoustic/explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ max. no. per year[2]	Additional Information
			No. of events per year	Max. no. of ordnance per year	No. of events[1] per year	Max. no. of ordnance per year		
Radar and Other System Testing	None	Offshore Area	Not previously analyzed		55	None	None	Up to 25% of testing may occur at night. This test includes new stressor (high-energy laser weapon) and a laser-based optical communication system (tested in Inland Waters and Offshore Area); all other systems previously analyzed. High-energy laser weapons would not be tested in the Inland Waters.
		Inland Waters (DBRC, NBK Bremerton)			8	None	None	
Semi-Stationary Equipment Testing	Acoustic	Inland Waters (DBRC, Keyport Range Site)	176	None	120	None	HF6 (459 hours) HF9 (140 hours) LF4 (88 hours) MF9 (138 hours) VHF2 (135 hours)	Dependent on testing event in this activity range of 0 to 25% of testing may occur at night. Anchored equipment and temporary mooring buoys may be deployed for the duration of a single test event or may be left in place for up to 12 months to support multiple events; all devices and their anchors are recovered. Bottom anchors are not deployed in known sensitive shallow water benthic habitats such as eelgrass beds. Acoustic test facility testing would occur at the Keyport Pier.
		Western Behm Canal, AK	2	None	2-3	None	HF6 (12 hours) HF9 (8 hours)	
Simulant Testing	None	Offshore Area	Not previously analyzed		50	None	None	Daytime testing only. All chemical simulants have low toxicity to humans and the environment. All biological simulants are considered to be Biosafety Level 1 organisms. Simulant testing will occur at least 3 nm from shore.

[1]For activities where the maximum number of events varies between years, a range is provided to indicate the "representative–maximum" number of events. For activities where no variation is anticipated, only the maximum number of events within a single year is provided.

[2]The number of hours and items listed is the maximum per year. Actual numbers could be less. Sonar source bins represent acoustic stressors and explosive source bins represent explosive stressors. Bins are defined in Table 5-2 and Table 5-7 of this BA.
[3] In-air detonations only.
Inland waters: NBK Bremerton, Carr Inlet Operations Area, Crescent Harbor EOD Range, DBRC, Hood Canal EOD Range, NS Everett, Keyport Range Site, Naval Magazine Indian Island and NAVY 3 OPAREA
Note: Orange cell indicates proposed increase, green cell indicates proposed decrease, and no shading indicates proposed no change from on-going activities.
Acronyms: DBRC = Dabob Bay Range Complex; EOD = Explosive Ordnance Disposal; NBK = Naval Base Kitsap; NEPM = Non-Explosive Practice Munitions; NS = Naval Station; QRS = Quinault Range Site; R- = Restricted Area.

Table 25. Naval Air Systems Command proposed testing activities for NWTT Phase III (Navy 2020e).

Range Activity	In-water Acoustic/ Explosive	Location	Ongoing Activities		Proposed Action		Sonar/Explosive bins and hours/ Max. no. per year[1]	Additional Information
			No. of events per year	Ordnance (Max. no. per year)	No. of events per year	Ordnance (Max. no. per year)		
Anti-Submarine Warfare								
Tracking Test – Maritime Patrol Aircraft	Acoustic	Offshore Area	43 6	None 70 IEER sonobuoy	4	None	ASW2 (200 sonobuoys) ASW5 (40 hours) MF5 (80 sonobuoys) MF6 (20 sonobuoys)	Assume one decelerator/parachute per sonobuoy Water depth >200 m. Activity occurs >50 NM from shore.
Tracking Test – Maritime Patrol Aircraft (SUS)	Explosive	Offshore Area	5	72 impulsive SUS buoys (e.g., MK-61, MK-64, MK-82)	4	80 impulsive SUS buoys (e.g., MK-61, MK-64, MK-82)	E1 (8) E3 (72)	Assume one decelerator/parachute per sonobuoy Explosive sonobuoy testing occurs at least 50 nm from shore and does not occur within the boundaries of the Olympic Coast National Marine Sanctuary.
Electronic Warfare								
Flare Test	None	Offshore Area	10	600 flares	0	None	None	
Other Testing								
ISR/EW MQ-4C Triton UAS Testing	None	Offshore Area	0	None	20	None	None	The Triton UAS will fly at altitudes ranging from approximately 5,500 to 50,000 feet above mean sea level within the NWTT Study Area.

(1) The number of hours and items listed is the maximum per year. Actual numbers could be less. Sonar source bins represent acoustic stressors and explosive source bins represent explosive stressors. Source bins are defined in Table 19 (sonar) and Table 20 (explosives) above.
Note: Orange cell indicates proposed increase, green cell indicates proposed decrease, and no shading indicates proposed no change from on-going activities.
Acronyms: EW = Electronic Warfare; IEER = Improved Extended Echo Ranging; ISR = Intelligence, Surveillance, Reconnaissance; SUS = Signal Underwater Sound; UAS = Unmanned Aerial System.

3.4 Standard Operating Procedures

When conducting training and testing activities the Navy implements standard operating procedures to provide for safety and mission success. Navy standard operating procedures are broadcast via numerous naval instructions and manuals to ensure compliance. Standard operating procedures applicable to training and testing have been developed through years of experience, and their primary purpose is to provide for safety (including public health and safety) and mission success. In many cases, there are benefits to environmental resources resulting from standard operating procedures.

3.4.1 Vessel Safety

Ships operated by or for the Navy have personnel assigned to stand watch at all times, day and night, when vessels are moving through the water (underway). Watch personnel undergo training on tasks such as avoiding hazards and ship handling. Training includes on-the-job instruction and a formal qualification program to certify that they have demonstrated all necessary skills. Skills include detection and reporting of floating or partially submerged objects. Watch personnel include officers, enlisted men and women, and civilians operating in similar capacities. Their duties as watchstanders may be performed in conjunction with other job responsibilities, such as navigating the ship or supervising other personnel. While on watch, personnel employ visual search techniques, including the use of binoculars and scanning techniques. After sunset and prior to sunrise, watch personnel employ night visual search techniques, which could include the use of night vision devices.

The primary duty of watch personnel is to ensure safety of the ship, and this includes the requirement to detect and report all objects and disturbances sighted in the water that may be indicative of a threat to the ship and its crew, such as debris, a periscope, a surfaced submarine, or a surface disturbance. Per safety requirements, watch personnel also report any marine mammals sighted that have the potential to be in the direct path of the ship as a standard collision avoidance procedure. The standard operating procedures for vessel safety could reduce adverse effects to marine mammals and sea turtles through a reduction in the potential for vessel strike due to the presence of watch personnel at all times.

3.4.2 Weapons Firing Safety

Most weapons firing activities that involve the use of explosive munitions are conducted during daylight hours. In addition, pilots of Navy aircraft are not authorized to expend ordnance, fire missiles, or drop other airborne devices through extensive cloud cover where visual clearance for non-participating aircraft and vessels in the air and on the sea surface is not possible. The two exceptions to this requirement are: (1) when operating in the open ocean, clearance for non-participating aircraft and vessels in the air and on the sea surface through radar surveillance is acceptable; and (2) when the Officer Conducting the Exercise or civilian equivalent accepts responsibility for the safeguarding of airborne and surface traffic. This standard operating

procedure benefits marine mammals and sea turtles by increasing the effectiveness of visual observations for mitigation during applicable explosive weapons firing activities.

3.4.3 Target Deployment and Retrieval Safety

The deployment and retrieval of targets is dependent upon environmental conditions. Firing exercises involving the deployment and retrieval of targets from small boats are typically conducted in daylight hours in Beaufort Sea State[12] number 4 conditions (i.e., winds 11 to 16 knots, small waves 1 to 4 feet becoming longer, numerous whitecaps) or better to ensure safe operating conditions during target deployment and recovery. These standard operating procedures benefit marine mammals and sea turtles by increasing the effectiveness of visual observations for mitigation, thereby reducing the potential for interactions with the weapons firing activities associated with the use of applicable deployed targets.

During activities that involve recoverable targets (e.g., aerial drones), the Navy recovers the target and any associated decelerators/parachutes to the maximum extent practicable consistent with personnel and equipment safety. Recovery of these items helps minimize the amount of materials that remain on the surface or on the seafloor, which could alert enemy forces to the presence of Navy assets during military missions and combat operations. This standard operating procedure benefits marine mammals, sea turtles, and fish by reducing the potential for physical disturbance and strike, entanglement, or ingestion of applicable targets and any associated decelerators/parachutes.

3.4.4 Underwater Detonation Safety

Underwater detonation training takes place in designated areas that are located away from popular recreational dive sites, primarily for human safety. Recreational dive sites often include artificial reefs and wrecks. This standard operating procedure benefits environmental resources (e.g., artificial reefs and the biological resources such as fish that inhabit, shelter in, or feed among them) by reducing the potential for interaction with underwater detonation activities.

3.4.5 Towed In-Water Device Safety

As a standard collision avoidance procedure, prior to deploying a towed in-water device from a manned platform, the Navy searches the intended path of the device for any floating debris, objects, or animals (e.g., driftwood, concentrations of floating vegetation, marine mammals) that have the potential to obstruct or damage the device. Concentrations of floating vegetation can be indicators of potential marine mammal or sea turtle presence because marine mammals and sea turtles have been known to seek shelter in, feed on, or feed among them. For example, young sea turtles have been known to hide from predators and eat the algae associated with floating concentrations of *Sargassum.* This standard operating procedure benefits marine mammals, sea

[12] http://w1.weather.gov/glossary/index.php?word=beaufort+scale

turtles, and vegetation through a reduction in the potential for physical disturbance and strike by towed in-water device.

3.5 Mitigation Measures[13]

The Navy proposed to implement mitigation measures to avoid or reduce potential impacts from acoustic, explosive, and physical disturbance and strike stressors from training and testing activities on ESA-listed species in the action area. NMFS considers these measures as reasonably certain to be implemented and thus components of the Navy's proposed action. These mitigation measures fall into two categories: procedural mitigation and mitigation areas. Procedural mitigation is mitigation that the Navy will implement whenever and wherever an applicable training or testing activity takes place within the NWTT Study Area. Mitigation areas are geographic locations in the NWTT Study Area where the Navy will implement additional measures during all or a part of the year. Additional detail on both proposed procedural mitigation and mitigation areas is provided in the sections below.

The following sections summarize the mitigation measures that the Navy proposes to implement in association with the training and testing activities analyzed in this document. A complete discussion of the mitigation measures, as well as measures considered by the Navy but not proposed, and the evaluation process used by the Navy to develop, assess, and select mitigation measures, can be found the Navy's Final Supplemental Environmental Impact Statement (SEIS) for this action (Navy 2019b). For each of the mitigation measures described below, the Navy operational community provided input on the practicability of each measure and whether additional mitigation could be implemented to further reduce potential impacts to ESA-listed species.

3.5.1 Procedural Mitigation

Procedural mitigation is mitigation that the Navy will implement whenever and wherever training or testing activities involving applicable acoustic, explosive, and physical disturbance and strike stressors take place within the NWTT Study Area. The Navy customized procedural mitigation for the activity categories and stressors applicable to the proposed action. Procedural mitigation generally involves: (1) the use of one or more trained Lookouts to observe for specific biological resources within a mitigation zone; (2) requirements for Lookouts to immediately communicate sightings of specific biological resources to the appropriate watch station for information dissemination; and (3) requirements for the watch station to implement mitigation (e.g., halt an activity) until certain recommencement conditions have been met.

[13] We consider these mitigation measures "conservation measures": actions that will be taken by the Navy and serve to minimize project effects on the species under review. As such we evaluate the effects of these measures as integral parts of the proposed action to be implemented by the Navy.

Lookouts are personnel who perform similar duties as the standard watch personnel described previously, such as observing for objects that could present a potential danger to the observation platform (e.g., debris in the water, incoming vessels, and incoming aircraft). Lookouts have an additional duty of helping meet the Navy's mitigation requirements by visually observing for marine mammals and sea turtles. However, for some activities, Lookouts may also be required to observe for additional biological resources, such as birds, fish, jellyfish aggregations, or floating vegetation. Some biological resources can be indicators of potential marine mammal or sea turtle presence because animals have been known to seek shelter in, feed on, or feed in them. For example, young sea turtles have been known to hide from predators and eat the algae associated with floating vegetation. The Navy proposes to observe for these additional biological resources during certain activities to protect ESA-listed species or to offer an additional protection for marine mammals and sea turtles.

Mitigation zones are areas at the surface of the water within which applicable training or testing activities will be ceased, powered down, or modified to protect specific ESA-listed species from an auditory injury (PTS), non-auditory injury (from impulsive sources), or direct strike (e.g., vessel strike) to the maximum extent practicable. Mitigation zones are measured as the radius from a stressor. Implementation of procedural mitigation is most effective when mitigation zones are appropriately sized to be realistically observed during typical training and testing activity conditions. The Navy customized its mitigation zone sizes and mitigation requirements for each applicable training and testing activity category or stressor. The Navy developed each mitigation zone to be the largest area that (1) Lookouts can reasonably be expected to observe during typical activity conditions (i.e., most environmentally protective), and (2) the Navy can commit to implementing mitigation without impacting safety, sustainability, or the ability to meet mission requirements.

Depending on the activity, a Lookout may be positioned on a ship (i.e., surface ships and surfaced submarines), on a small boat (e.g., a rigid-hull inflatable boat), in an aircraft, on a pier, or on the shore. Certain platforms, such as aircraft and small boats, have manning or space restrictions; therefore, the Lookout on these platforms is typically an existing member of the aircraft or boat crew (e.g., pilot) who is responsible for other essential tasks (e.g., navigation). On platforms that do not have manning and space restrictions (such as large ships), the Officer of the Deck, a member of the bridge watch team, or other personnel may be designated as the Lookout. The Navy is unable to position Lookouts on unmanned vehicles and unmanned aerial systems, or have Lookouts observe during activities that use systems deployed from or towed by unmanned platforms.

The Navy's passive acoustic devices (e.g., remote acoustic sensors, expendable sonobuoys, passive acoustic sensors on submarines) can complement visual observations when passive acoustic assets are already participating in an activity. When in use, the passive acoustic assets can detect vocalizing marine mammals within the frequency bands already being monitored by

Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot be used to determine an animal's location or confirm its presence in a mitigation zone. Marine mammal detections made with the use of passive acoustic devices will be communicated to Lookouts to alert them of possible marine mammal presence in the vicinity. Lookouts will use any information on possible presence of animals from passive acoustic monitoring to assist in their visual observations of the mitigation zone.

The Navy takes several courses of action in response to a sighting of an applicable biological resource (e.g., ESA-listed species, floating vegetation) in a mitigation zone. First, a Lookout will communicate the sighting to the appropriate watch station. Next, the watch station will implement the prescribed mitigation (e.g., powering down sonar, halting an explosion, maneuvering a vessel). If floating vegetation is observed prior to the initial start of an activity, the activity will either be relocated to an area where floating vegetation is not observed, or the initial start of the activity will be halted until the mitigation zone is clear of floating vegetation (the Navy does not propose to halt activities if vegetation floats into the mitigation zone after activities commence as the Navy determined such an action not to be practical for operational and safety reasons). For sightings of marine mammals and sea turtles during an activity, the activity will be suspended or otherwise altered based on the applicable mitigation measures until one of the five recommencement conditions listed below has been met. The recommencement conditions are designed to allow a sighted animal to leave the mitigation zone before an activity or the use of a stressor resumes.

1) The animal is observed exiting the mitigation zone;
2) The animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the stressor source;
3) The mitigation zone has been clear of any additional sightings for a specific wait period;
4) For mobile activities, the stressor source has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting; or
5) For activities using hull-mounted sonar, the ship concludes that dolphins are deliberately closing in on the ship to ride the ship's bow wave, and are therefore out of the main transmission axis of the sonar (and there are no other marine mammal or sea turtle sightings within the mitigation zone).

In some instances, such as if an animal dives underwater after a sighting, it may not be possible for a Lookout to visually verify if that animal has left the mitigation zone. To account for this, one of the recommencement conditions is an established post-sighting wait period. Wait periods are designed to allow animals time to resurface and be available to be sighted again before an activity or the use of a stressor resumes. The Navy proposes a 30-minute wait period to activities conducted from vessels and activities that involve aircraft that are not typically fuel constrained (e.g., maritime patrol aircraft). Thirty minutes is the maximum amount of time that those

activities can be halted without preventing the activity from meeting its intended objective (Navy 2018b). A 30 minute period covers the average dive times of most marine mammals, and a portion of the dive times of sea turtles and deep-diving marine mammals (i.e., sperm whales, dwarf and pygmy sperm whales [*Kogia* species], and beaked whales). The Navy proposes a shorter wait period of 10 minutes for activities that involve aircraft with fuel constraints (e.g., rotary-wing aircraft [i.e., helicopters], fighter aircraft), since 10 minutes is the maximum amount of time that those activities can be halted without compromising safety due to aircraft fuel restrictions (Navy 2018b). A 10 minute period covers a portion of the marine mammal and sea turtle dive times, but not the average dive times of all species.

The first procedural mitigation (Environmental Awareness and Education) is designed to aid Lookouts and other personnel with their observation and environmental compliance responsibilities, as well as training and testing activity reporting requirements. The remainder of the procedural mitigation measures are organized by stressor type and activity category. For sonar and explosive sources, proposed mitigation is dependent on the sonar source and the NEW of the detonation.

3.5.1.1 Environmental Awareness and Education

The Navy provides environmental awareness and education training to aid in visual observation, environmental compliance, and reporting responsibilities. This training helps Navy personnel gain a better understanding of their personal environmental compliance roles and responsibilities and helps to ensure Navy-wide compliance with environmental requirements. The Navy will provide environmental awareness and education training modules to the appropriate personnel as outlined in Table 26.

Table 26. Environmental awareness and education procedural mitigation

Procedural Mitigation Description
Stressor or Activity • All training and testing activities, as applicable
Resource Protection Focus • Marine mammals • Sea turtles
Mitigation Requirements • Appropriate personnel (including civilian personnel) involved in mitigation and training or testing activity reporting under the Proposed Action will complete one or more modules of the U.S. Navy Afloat Environmental Compliance Training Series, as identified in their career path training plan. Modules include: ○ **Introduction to the U.S. Navy Afloat Environmental Compliance Training Series**. The introductory module provides information on environmental laws (e.g., ESA, MMPA) and the corresponding responsibilities that are relevant to Navy training and testing activities. The material explains why environmental compliance is important in supporting the Navy's commitment to environmental stewardship. ○ **Marine Species Awareness Training**. All bridge watch personnel, Commanding Officers, Executive Officers, maritime patrol aircraft aircrews, anti-submarine warfare and mine warfare rotary-wing

Procedural Mitigation Description
aircrews, Lookouts, and equivalent civilian personnel must successfully complete the Marine Species Awareness Training prior to standing watch or serving as a Lookout. The Marine Species Awareness Training provides information on sighting cues, visual observation tools and techniques, and sighting notification procedures. Navy biologists developed Marine Species Awareness Training to improve the effectiveness of visual observations for biological resources, focusing on marine mammals and sea turtles, and including floating vegetation, jellyfish aggregations, and flocks of seabirds. ○ **U.S. Navy Protective Measures Assessment Protocol**. This module provides the necessary instruction for accessing mitigation requirements during the event planning phase using the Protective Measures Assessment Protocol software tool. ○ **U.S. Navy Sonar Positional Reporting System and Marine Mammal Incident Reporting**. This module provides instruction on the procedures and activity reporting requirements for the Sonar Positional Reporting System and marine mammal incident reporting.

3.5.1.2 Active Sonar

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from active sonar, as outlined in Table 27. For low-frequency active sonar at 200 dB re 1 µPa rms and hull-mounted mid-frequency active sonar, bin MF1 has the longest predicted ranges to PTS. For low-frequency active sonar below 200 dB re 1 µPa rms, mid-frequency active sonar sources that are not hull-mounted, and high-frequency active sonar, bin HF4 has the longest predicted ranges to PTS. For the highest source levels in bin MF1 and HF4, the mitigation zones extend beyond the respective average ranges to PTS for marine mammals. The mitigation zones for active sonar will help avoid or reduce the potential for exposure to PTS for marine mammals. The active sonar mitigation zones also extend into a portion of the average ranges to TTS for marine mammals; therefore, mitigation will help avoid or reduce the potential for some exposure to TTS. Active sonar sources that fall within lower source bins or are used at lower source levels have shorter impact ranges than those discussed above; therefore, the mitigation zones will extend further beyond or into the average ranges to PTS and TTS for these sources.

Due to sea turtle hearing capabilities, the mitigation only applies to sea turtles during the use of sources below 2 kilohertz (kHz). The range to auditory effects for most active sonar sources in sea turtle hearing range (e.g., LF4) is zero meters. Impact ranges are longer (i.e., up to tens of meters) for active sonars with higher source levels. The mitigation zones for active sonar extend beyond the ranges to PTS and TTS for sea turtles; therefore, mitigation will help avoid or reduce the potential for exposure to these effects for sea turtles.

Table 27. Procedural mitigation for active sonar.

Procedural Mitigation Description
Stressor or Activity • Low-frequency active sonar, mid-frequency active sonar, high-frequency active sonar ○ For vessel-based active sonar activities, mitigation applies only to sources that are positively controlled and deployed from manned surface vessels (e.g., sonar sources towed from manned surface platforms). ○ For aircraft-based active sonar activities, mitigation applies only to sources that are positively controlled and deployed from manned aircraft that do not operate at high altitudes (e.g., rotary-wing aircraft). Mitigation does not apply to active sonar sources deployed from unmanned aerial systems or aircraft operating at high altitudes (e.g., maritime patrol aircraft).
Resource Protection Focus • Marine mammals • Sea turtles (only for sources <2 kilohertz [kHz])
Number of Lookouts and Observation Platform • Hull-mounted sources: ○ 1 Lookout: Platforms with space or manning restrictions while underway (at the forward part of a small boat or ship) and platforms using active sonar while moored or at anchor (including pierside) ○ 2 Lookouts: Platforms without space or manning restrictions while underway (at the forward part of the ship) • Sources that are not hull-mounted: ○ 1 Lookout on the ship or aircraft conducting the activity
Mitigation Requirements • Mitigation zones: ○ 1,000 yard (914 meter) power down, 500 yard (457 meter) power down, and 200 yard (183 meter) or 100 yard (91 meter) shut down for low-frequency active sonar at 200 decibels (dB) and hull-mounted mid-frequency active sonar ○ 200 yard (183 meter) or 100 yard (91 meter) shut down for low-frequency active sonar <200 dB, mid-frequency active sonar sources that are not hull-mounted, and high-frequency active sonar • Prior to the initial start of the activity (e.g., when maneuvering on station): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of active sonar transmission. • During the activity: ○ Low-frequency active sonar at 200 decibels (dB) and hull-mounted mid-frequency active sonar: Observe the mitigation zone for marine mammals and sea turtles (for sources <2 kHz); power down active sonar transmission by 6 dB if a marine mammal or sea turtle is observed within 1,000 yards (914 meters) of the sonar source; power down an additional 4 dB (10 dB total) if a marine mammal or sea turtle is observed within 500 yards (457 meters); cease transmission if a cetacean or sea turtle is observed within 200 yards (183 meters); cease transmission if a pinniped in the NWTT Offshore Area or Western Behm Canal is observed within 200 yards (183 meter); cease transmission if a pinniped in NWTT Inland Waters is observed within 100 yards (91 meters) (except if hauled out on, or in the water near, man-made structures and vessels). ○ Low-frequency active sonar <200 dB, mid-frequency active sonar sources that are not hull-mounted, and high-frequency active sonar: Observe the mitigation zone for marine mammals and sea turtles (for sources <2 kHz); cease transmission if a cetacean, sea turtle, or pinniped in the NWTT Offshore Area or Western Behm Canal is observed within 200 yards (183 meters) of the sonar source; cease transmission if a pinniped in NWTT Inland Waters is observed within 100 yards (91 meters) (except if hauled out on, or in the water near, man-made structures and vessels). • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: ○

Procedural Mitigation Description
Mitigation Requirements (Continued) ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing or powering up active sonar transmission) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the sonar source; (3) the mitigation zone has been clear from any additional sightings for 10 min. for aircraft-deployed sonar sources or 30 min. for vessel-deployed sonar sources; (4) for mobile activities, the active sonar source has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting; or (5) for activities using hull-mounted sonar, the Lookout concludes that dolphins are deliberately closing in on the ship to ride the ship's bow wave, and are therefore out of the main transmission axis of the sonar (and there are no other marine mammal sightings within the mitigation zone).

3.5.1.3 Weapons Firing Noise

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from weapons firing noise, as outlined in Table 28. The mitigation zone extends beyond the distance to which marine mammals and sea turtles would likely experience PTS or TTS from weapons firing noise; therefore, mitigation will help avoid or reduce the potential for exposure to these impacts.

Table 28. Procedural mitigation for weapons firing noise.

Procedural Mitigation Description
Stressor or Activity • Weapons firing noise associated with large-caliber gunnery activities
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned on the ship conducting the firing ○ Depending on the activity, the Lookout could be the same one described in Table 31 for Explosive Medium-Caliber and Large-Caliber Projectiles or Table 38 for Small-, Medium-, and Large-Caliber Non-Explosive Practice Munitions.
Mitigation Requirements • Mitigation zone: ○ 30° on either side of the firing line out to 70 yards (64 meters) from the muzzle of the weapon being fired • Prior to the initial start of the activity: ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of weapons firing. • During the activity: ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease weapons firing. • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing weapons firing)

Procedural Mitigation Description
until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the firing ship; (3) the mitigation zone has been clear from any additional sightings for 30 min.; or (4) for mobile activities, the firing ship has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting.

3.5.1.4 Explosive Sonobuoys

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive sonobuoys, as outlined in Table 29. In the NMFS NWTT 2015 biological opinion, explosive sonobuoys had two mitigation zone sizes based on NEW and the associated average ranges to PTS. When developing mitigation for the Proposed Action, the Navy analyzed the potential for increasing the size of these mitigation zones. The Navy identified an opportunity to increase the mitigation zone size by 250 yards (228 meters) for sonobuoys using up to 2.5-pound (lb.) NEW so that explosive sonobuoys will implement a 600-yard (549-meter) mitigation zone, regardless of NEW, to enhance protections to the maximum extent practicable.

Table 29. Procedural mitigation for explosive sonobuoys.

Procedural Mitigation Description
Stressor or Activity • Explosive sonobuoys
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in an aircraft or on a small boat • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.
Mitigation Requirements • Mitigation zone: o 600 yards (549 meters) around an explosive sonobuoy • Prior to the initial start of the activity (e.g., during deployment of a sonobuoy field, which typically lasts 20–30 min.): o Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. o Conduct passive acoustic monitoring for marine mammals; use information from detections to assist visual observations. o Visually observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of sonobuoy or source/receiver pair detonations. • During the activity: o Observe the mitigation zone for marine mammals and sea turtles; if observed, cease sonobuoy or source/receiver pair detonations.

Procedural Mitigation Description
Mitigation Requirements (Continued) • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: o The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing detonations) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the sonobuoy; or (3) the mitigation zone has been clear from any additional sightings for 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained. • After completion of the activity (e.g., prior to maneuvering off station): o When practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures. o If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

The Navy developed a new mitigation measure requiring the Lookout to observe the mitigation zone after completion of the activity. In accordance with the NMFS NWTT 2015 biological opinion consultation requirements, the Navy currently conducts post-activity observations for some, but not all explosive activities. When developing mitigation during this consultation, the Navy determined that it could expand this requirement to other explosive activities for enhanced consistency and to help determine if any resources were injured during explosive events, when practical. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. There are typically multiple platforms in the vicinity of activities that use explosive sonobuoys (e.g., safety aircraft). When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting an ESA-listed species.

Some activities that use explosive sonobuoys involve detonations of a single sonobuoy or sonobuoy pair, while other activities involve deployment of multiple sonobuoys that may be dispersed in a pattern over a large distance. Lookouts will have a better likelihood of detecting marine mammals and sea turtles when observing the mitigation zone around a single sonobuoy or sonobuoy pair than when observing multiple sonobuoys dispersed over a large distance. When observing large distances, Lookouts will be more likely to detect large visual cues (e.g., whale blows, breaching whales, or large pods of dolphins) than individual marine mammals, cryptic marine mammal species, and sea turtles.

Bin E3 has the longest predicted impact ranges for explosive sonobuoys used in the NWTT Study Area (e.g., MK-61 Signal Underwater Sound sonobuoys). For the largest explosive in bin E3, the mitigation zone extends beyond the ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The mitigation zone extends beyond the

average ranges to PTS for sea turtles, mid-frequency cetaceans, and otariids, and into a portion of the average ranges to PTS for low-frequency cetaceans. The mitigation zone also extends beyond or into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E3. Smaller explosives in bin E3 and explosives in smaller source bins (E1) have shorter predicted impact ranges; therefore, the mitigation zone will extend further beyond or cover a greater portion of the impact ranges for these explosives.

3.5.1.5 Explosive Torpedoes

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive torpedoes, as outlined in

Table 30. The post-activity observations for explosive torpedoes will help the Navy determine if any resources were injured during the activity. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. Typically, when aircraft are firing explosive torpedoes, there are additional observation aircraft, support vessels (e.g., range craft for torpedo retrieval), or other safety aircraft in the vicinity. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources. Explosive torpedo activities involve detonations at a target located down range of the firing platform. Due to the distance between the mitigation zone and the observation platform, Lookouts will have a better likelihood of detecting large visual cues (e.g., whale blows, breaching whales, or large pods of dolphins) than individual marine mammals, cryptic marine mammal species, and sea turtles.

Bin E11 has the longest predicted impact ranges for explosive torpedoes used in the NWTT Study Area. For the largest explosive in bin E11, the mitigation zone extends beyond the ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The mitigation zone extends beyond the average ranges to PTS for sea turtles, low-frequency cetaceans, and otariids, and into a portion of the average ranges to PTS for low-frequency cetaceans. The mitigation zone also extends into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E11. Explosive torpedoes in smaller source bins (e.g., E8) have shorter predicted impact ranges; therefore, the mitigation zone will extend further beyond or cover a greater portion of the impact ranges for these explosives.

Table 30. Procedural mitigation for explosive torpedoes.

Procedural Mitigation Description
Stressor or Activity • Explosive torpedoes
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in an aircraft • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.
Mitigation Requirements • Mitigation zone: ○ 2,100 yards (1,920 meters) around the intended impact location • Prior to the initial start of the activity (e.g., during deployment of the target): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Conduct passive acoustic monitoring for marine mammals; use information from detections to assist visual observations. ○ Visually observe the mitigation zone for marine mammals, sea turtles, and jellyfish aggregations; if observed, relocate or delay the start of firing. • During the activity: ○ Observe the mitigation zone for marine mammals, sea turtles, and jellyfish aggregations; if observed, cease firing. • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing firing) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended impact location; or (3) the mitigation zone has been clear from any additional sightings for 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained. • After completion of the activity (e.g., prior to maneuvering off station): ○ When practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures. ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

3.5.1.6 Explosive Medium-Caliber and Large-Caliber Projectiles

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive gunnery activities, as outlined in Table 31. In the NMFS NWTT 2015 biological opinion, explosive gunnery activity mitigation zones were based on NEW and the associated average ranges to PTS. When developing mitigation during this consultation, the Navy analyzed the potential for increasing the size of these mitigation zones.

The Navy identified an opportunity to increase the mitigation zone size by 400 yards (366 meters) for surface-to-surface activities to enhance protections to the maximum extent practicable.

The Navy developed a new mitigation measure requiring the Lookout to observe the mitigation zone after completion of the activity. In accordance with the NWTT 2015 biological opinion consultation requirements, the Navy currently conducts post-activity observations for some, but not all explosive activities. When developing mitigation for the Proposed Action, the Navy determined that it could expand this requirement to other explosive activities for enhanced consistency and to help determine if any resources were injured during explosive events, when practical. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. Typically, when aircraft are firing explosive munitions there are additional observation aircraft, multiple aircraft firing munitions, or other safety aircraft in the vicinity. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources.

Large-caliber gunnery activities involve vessels firing projectiles at targets located up to six nautical miles down range. Medium-caliber gunnery activities involve vessels firing projectiles at targets located up to 4,000 yards (3,658 meters) down range, although typically much closer. Due to their relatively lower vantage point, Lookouts (during medium-caliber or large-caliber gunnery exercises) will be more likely to detect large visual cues (e.g., whale blows, breaching whales, or large pods of dolphins) than individual marine mammals, cryptic marine mammal species, and sea turtles when observing around targets located at the furthest firing distances. The Navy will implement larger mitigation zones for large-caliber gunnery activities than for medium-caliber gunnery activities due to the nature of how the activities are conducted. For example, large-caliber gunnery activities are conducted from surface combatants, so Lookouts can observe a larger mitigation zone because they typically have access to high-powered binoculars mounted on the ship deck. This will enable observation of the distant mitigation zone in combination with hand-held binoculars and naked-eye scanning.

Bin E5 (e.g., 5-inch projectiles) has the longest predicted impact ranges for explosive projectiles that apply to the 1,000-yard (914-meter) mitigation zone. Bin E2 (e.g., 40-milimeter projectiles) has the longest predicted impact ranges for explosive projectiles that apply to the 600-yard (549-meter) mitigation zone. The 1,000-yard (914-meter) and 600-yard (549-meter) mitigation zones extend beyond the respective ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The 1,000-yard (914-meter) and 600-yard (549-meter) mitigation zones extend beyond the respective average ranges to PTS for sea turtles, mid-frequency cetaceans, low-frequency cetaceans, and otariids. The mitigation zones also extend beyond or into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non- auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E5 and bin E2. Explosives in smaller source bins (e.g., E1) have shorter

predicted impact ranges; therefore, the mitigation zones will extend further beyond or cover a greater portion of the impact ranges for these explosives.

Table 31. Procedural mitigation for explosive medium-caliber and large-caliber projectiles.

Procedural Mitigation Description
Stressor or Activity • Gunnery activities using explosive medium-caliber and large-caliber projectiles   ○ Mitigation applies to activities using a surface target
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout on the vessel conducting the activity   ○ For activities using explosive large-caliber projectiles, depending on the activity, the Lookout could be the same as the one described in Table 28 for Weapons Firing Noise. • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.

Mitigation Requirements
• Mitigation zones:
  ○ 600 yards (549 meters) around the intended impact location for explosive medium-caliber projectiles
  ○ 1,000 yards (914 meters) around the intended impact location for explosive large-caliber projectiles
• Prior to the initial start of the activity (e.g., when maneuvering on station):
  ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear.
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of firing.
• During the activity:
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease firing.
• Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity:
  ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing firing) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended impact location; (3) the mitigation zone has been clear from any additional sightings for 30 min. for vessel-based firing; or (4) for activities using mobile targets, the intended impact location has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting.
• After completion of the activity (e.g., prior to maneuvering off station):
  ○ When practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures.
  ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

3.5.1.7 Explosive Missiles

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive missiles, as outlined in Table 32. In the NMFS NWTT 2015 biological opinion, explosive missile mitigation zones were based on NEW and the associated average ranges to PTS. When developing the mitigation for this consultation, the Navy analyzed the potential for increasing the mitigation zone sizes. The Navy determined that the current mitigation zone for explosive missiles is the largest area within which it is practical to implement mitigation for this activity; therefore, it will continue implementing this same mitigation zone (i.e., 2,000 yards (1,829 meters)) under the proposed action.

The Navy developed a new mitigation measure requiring the Lookout to observe the mitigation zone after completion of the activity. In accordance with the 2015 NWTT biological opinion consultation requirements, the Navy currently conducts post-activity observations for some, but not all explosive activities. When developing the mitigation for this consultation, the Navy determined that it could expand this requirement to other explosive activities for enhanced consistency and to help determine if any resources were injured during explosive events, when practical. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. Typically, when aircraft are firing explosive munitions there are additional observation aircraft, multiple aircraft firing munitions, or other safety aircraft in the vicinity. For example, during typical explosive missile exercises, two aircraft circle the activity location. One aircraft clears the intended impact location while the other fires, and vice versa. A third aircraft is typically present for safety or proficiency inspections. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources.

Bin E10 (e.g., Harpoon missiles) has the longest predicted impact ranges for explosive missiles used in the NWTT Study Area. The 2,000 yard (1,829 meters) mitigation zone extends beyond the ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The mitigation zone extends beyond the average ranges to PTS for sea turtles, low-frequency cetaceans, mid-frequency cetaceans, and otariids. The mitigation zone also extends beyond or into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E10.

Table 32. Procedural mitigation for explosive missiles.

Procedural Mitigation Description
Stressor or Activity • Aircraft-deployed explosive missiles   ○ Mitigation applies to activities using a surface target
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in an aircraft • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.

Mitigation Requirements
• Mitigation zone:
  ○ 2,000 yards (1,829 meters) around the intended impact location
• Prior to the initial start of the activity (e.g., during a fly-over of the mitigation zone):
  ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear.
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of firing.
• During the activity:
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease firing.
• Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity:
  ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing firing) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended impact location; or (3) the mitigation zone has been clear from any additional sightings for 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained.
• After completion of the activity (e.g., prior to maneuvering off station):
  ○ When practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures.
  ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

3.5.1.8 Explosive Bombs

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive bombs, as outlined in Table 33. In the 2015 NWTT biological opinion, the explosive bombing mitigation zone was based on the NEW and the associated average ranges to PTS. When developing the mitigation during this consultation, the Navy analyzed the potential for increasing the size of this mitigation zone. The Navy determined that the current mitigation zone for explosive bombs is the largest area within which it is

practical to implement mitigation for this activity; therefore, it will continue implementing this same mitigation zone under the Proposed Action.

The Navy developed a new mitigation measure requiring the Lookout to observe the mitigation zone after completion of this activity. In accordance with the 2015 NWTT biological opinion consultation requirements, the Navy currently conducts post-activity observations for some, but not all explosive activities. When developing mitigation for the Proposed Action, the Navy determined that it could expand this requirement to other explosive activities for enhanced consistency and to help determine if any resources were injured during explosive events, when practical. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. Typically, when aircraft are firing explosive munitions there are additional observation aircraft, multiple aircraft firing munitions, or other safety aircraft in the vicinity. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources.

Bombing exercises involve an aircraft deploying munitions at a surface target located beneath the firing platform. During target approach, aircraft maintain a relatively steady altitude of approximately 1,500 feet Lookouts, by necessity for safety and mission success, primarily focus their attention on the water surface surrounding the intended detonation location (i.e., the mitigation zone). Being positioned in an aircraft gives the Lookout a good vantage point for observing marine mammals and sea turtles throughout the mitigation zone.

Bin E10 (e.g., 500-lb. NEW bombs) has the longest predicted impact ranges for explosive bombs used in the NWTT Study Area. The 2,500-yard (2286-meter) mitigation zone extends beyond the ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The mitigation zone extends beyond the average ranges to PTS for sea turtles, mid-frequency cetaceans, low-frequency cetaceans, and otariids. The mitigation zone also extends beyond or into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest bombs in bin E10. Smaller bombs (e.g., 250-lb. bombs) have shorter predicted impact ranges; therefore, the mitigation zone will extend further beyond or cover a greater portion of the impact ranges for these explosives.

Table 33. Procedural mitigation for explosive bombs.

Procedural Mitigation Description
Stressor or Activity • Explosive bombs
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in the aircraft conducting the activity • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.
Mitigation Requirements • Mitigation zone: ○ 2,500 yards (2286-meters) around the intended target • Prior to the initial start of the activity (e.g., when arriving on station): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of bomb deployment. • During the activity (e.g., during target approach): ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease bomb deployment. • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing bomb deployment) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended target; (3) the mitigation zone has been clear from any additional sightings for 10 min.; or (4) for activities using mobile targets, the intended target has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting. • After completion of the activity (e.g., prior to maneuvering off station): ○ When practical (e.g., when platforms are not constrained by fuel restrictions or mission-essential follow-on commitments), observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures. ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

3.5.1.9 Explosive Mine Countermeasure and Neutralization Activities

The Navy has developed new procedural mitigation to avoid or reduce potential impacts on marine mammals and sea turtles from explosive mine countermeasure and neutralization activities, as outlined in Table 33. The mitigation applies to all explosive mine countermeasure and neutralization activities except those that involve the use of Navy divers, which are discussed further below.

Table 34. Procedural mitigation for explosive mine countermeasure and neutralization activities.

Procedural Mitigation Description
Stressor or Activity • Explosive mine countermeasure and neutralization activities
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned on a vessel or in an aircraft when implementing the smaller mitigation zone • 2 Lookouts (one positioned in an aircraft and one on a small boat) when implementing the larger mitigation zone • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.
Mitigation Requirements • Mitigation zones: ○ 600 yards (549 meters) around the detonation site for activities using ≤ 5 pounds net explosive weight ○ 2,100 yards (1,920 meters) around the detonation site for activities using > 5–60 pounds net explosive weight • Prior to the initial start of the activity (e.g., when maneuvering on station; typically, 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of detonations. • During the activity: ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease detonations. ○ The Navy will use the smallest practicable charge size for each activity. ○ The Navy will conduct activities in daylight hours only in Beaufort Sea state number 3 conditions or less. • Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity: ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing detonations) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to detonation site; or (3) the mitigation zone has been clear from any additional sightings for 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained. • After completion of the activity (typically 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained): ○ Observe the vicinity of where detonations occurred; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures. ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred.

The types of charges used in these activities are positively controlled, which means the detonation is controlled by the personnel conducting the activity and is not authorized until the mitigation zone is clear at the time of detonation. The post-activity observations will help the Navy determine if any resources were injured during the activity. Additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources. The small observation area and proximity to the observation platform will result in a high likelihood that the Lookout will be able to detect marine mammals and sea turtles throughout the mitigation zone (regardless of the type of observation platform used). Conducting explosive activities in the daytime in Beaufort sea state number 3 conditions or better (i.e., good visibility conditions) will increase the likelihood that marine mammals and sea turtles will be detected prior to and during the activity. By using the smallest practicable charge for each activity (e.g., using a smaller net explosive weight if the testing program objective can still be met), the Navy will be able to reduce potential impacts on marine mammals, sea turtles, and ESA-listed fish while maintaining the ability to accomplish the required testing program requirements.

Bin E7 (e.g., 60 pounds mine) has the longest predicted impact ranges for explosives that apply to the 2,100 yard (1,920 meter) mitigation zone. Bin E4 (e.g., five pounds NEW charge) has the longest predicted impact ranges for explosives that apply to the 600 yard (549 meter) mitigation zone. The 2,100 yard (1,920 meter) and 600 yard (549 meter) mitigation zones extend beyond the respective ranges to 50 percent non-auditory injury and 50 percent mortality for sea turtles and marine mammals. The 2,100 yard (1,920 meter) mitigation zone extends beyond the respective average ranges to PTS for sea turtles, low-frequency cetaceans, mid-frequency cetaceans, and otariids. The 600 yard (549 meter) mitigation zone extends beyond the respective average ranges to PTS for sea turtles, mid-frequency cetaceans, and otariids, and into a portion of the average ranges to PTS for low-frequency cetaceans. The mitigation zones also extend into a portion of the average ranges to TTS for sea turtles and marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E7 and bin E4. Smaller explosives within bin E7 and bin E4 have shorter predicted impact ranges; therefore, the mitigation zones will cover a greater portion of the impact ranges for these explosives.

In addition to the procedural mitigation described above, the Navy has proposed to conduct mine countermeasure and neutralization testing using explosives within bin E7 no closer to shore than six nautical miles within the Quinault Range Site (Navy 2019a).

3.5.1.10 Explosive Mine Neutralization Activities Involving Navy Divers

The Navy will implement procedural mitigation to avoid or reduce potential impacts on marine mammals and fish (salmonids) from explosive mine neutralization activities involving Navy divers (Table 35). In the NMFS NWTT 2015 biological opinion, the mitigation zone for explosive mine neutralization activities involving Navy divers was based on NEW and the associated average ranges to PTS. When developing the mitigation during this consultation, the Navy analyzed the potential for increasing the size of the mitigation zones. The Navy identified an opportunity to increase the mitigation zone size by 100 yards (91 meters) to enhance protections to the maximum extent practicable. The post-activity observations are a continuation from the 2015 opinion and will help the Navy determine if any resources were injured during the activity. The Navy is adding a requirement that additional platforms already participating in the activity will support observing the mitigation zone before, during, and after the activity while performing their regular duties. When available, having additional personnel support observations of the mitigation zone will help increase the likelihood of detecting biological resources.

Bin E3 has the longest predicted impact ranges for explosives used for these activities in the NWTT Study Area. The 500 yard (457 meter) mitigation zone extends beyond the respective ranges to 50 percent non-auditory injury and 50 percent mortality for marine mammals. The mitigation zone extends beyond the average ranges to PTS for low-frequency cetaceans, mid-frequency cetaceans, and otariids. The mitigation zone also extends beyond or into a portion of the average ranges to TTS for marine mammals. Therefore, depending on the species, mitigation will help avoid or reduce all or a portion of the potential for exposure to mortality, non-auditory injury, PTS, and higher levels of TTS for the largest explosives in bin E3. Smaller explosives within bin E3 have shorter predicted impact ranges; therefore, the mitigation zones will cover a greater portion of the impact ranges for these explosives.

The charges used during explosive mine neutralization activities involving Navy divers in the Study Area are all positively controlled. This will allow Lookouts time to continuously observe the mitigation zone for biological resources right up to the point of detonation. By using the smallest practicable positive control charge for each activity (e.g., using 1.5 pounds net explosive weight in place of 2.5 pounds net explosive weight if the training objective can still be met), the Navy will be able to minimize potential impacts while maintaining the ability to accomplish the required training objectives. The pre-activity observations will typically entail a line transect survey (with each transect being no more than approximately 50 meters wide) at speeds ranging between approximately 5 to 10 knots. The primary Lookouts for this activity will not include the boat drivers; however, the boat drivers will support the Lookouts while performing their regular duties. The small observation area and proximity to observation platforms will result in a high likelihood that Lookouts will be able to detect marine mammals throughout the mitigation zone. Observing for indicators of marine mammal presence will further help avoid or reduce impacts on these resources within the mitigation zone.

Table 35. Procedural mitigation for explosive mine neutralization activities involving Navy divers.

Procedural Mitigation Description
Stressor or Activity • Explosive mine neutralization activities involving Navy divers
Resource Protection Focus • Marine mammals • Fish
Number of Lookouts and Observation Platform • 2 Lookouts on two small boats with one Lookout each, one of which will be a Navy biologist • All divers placing the charges on mines will support the Lookouts while performing their regular duties and will report applicable sightings to the lead Lookout, the supporting small boat, or the Range Safety Officer. • If additional platforms are participating in the activity, personnel positioned in those assets (e.g., safety observers, evaluators) will support observing the mitigation zone for applicable biological resources while performing their regular duties.
Mitigation Requirements • Mitigation zones: ○ 500 yards (457 meters) for marine mammals around the detonation site during activities using > 0.5–2.5 pounds net explosive weight • Prior to the initial start of the activity (starting 30 min. before the first planned detonation): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals; if observed, relocate or delay the start of detonations. ○ The Navy will ensure the area is clear of marine mammals for 30 min. prior to commencing a detonation. ○ A Navy biologist will serve as the lead Lookout and will make the final determination that the mitigation zone is clear of marine mammals prior to the commencement of a detonation. The Navy biologist will maintain radio communication with the unit conducting the event and the other Lookout. • During the activity: ○ Observe the mitigation zone for marine mammals; if observed, cease detonations. ○ To the maximum extent practicable depending on mission requirements, safety, and environmental conditions, boats will position themselves near the mid-point of the mitigation zone radius (but outside of the detonation plume and human safety zone), will position themselves on opposite sides of the detonation location (when two boats are used), and will travel in a circular pattern around the detonation location with one Lookout observing inward toward the detonation site and the other observing outward toward the perimeter of the mitigation zone. ○ The Navy will use only positively controlled charges (i.e., no time-delay fuses). ○ The Navy will use the smallest practicable charge size for each activity. ○ Activities will be conducted in Beaufort Sea state number 2 conditions or better and will not be conducted in low visibility conditions. • Commencement/recommencement conditions after a marine mammal sighting before or during the activity: ○ The Navy will allow a sighted marine mammal to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing detonations) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the detonation site; or (3) the mitigation zone has been clear from any additional sightings for 30 min.

Procedural Mitigation Description
• After each detonation and the completion of an activity (for 30 min): ○ Observe the vicinity of where detonations occurred and immediately downstream of the detonation location; if any injured or dead ESA-listed species are observed, follow established incident reporting procedures. ○ If additional platforms are supporting this activity (e.g., providing range clearance), these assets will assist in the visual observation of the area where detonations occurred. ○ If any injured or dead fish are observed, notify the appropriate Navy Region Environmental Director, Navy Pacific Fleet Environmental Office, and local base wildlife biologist and include information on the number of adults or juveniles and species, if possible. ○

3.5.1.11 Vessel Movement

The Navy will implement procedural mitigation to avoid or reduce the potential for vessel strikes of marine mammals and sea turtles, as outlined in Table 36. Although the Navy is unable to position Lookouts on unmanned vessels, as a standard operating procedure, some vessels that operate autonomously have embedded sensors that aid in avoidance of large objects. The embedded sensors may help those unmanned vessels avoid vessel strikes of marine mammals.

Table 36. Procedural mitigation for vessel movement.

Procedural Mitigation Description
Stressor or Activity • Vessel movement ○ The mitigation will not be applied if: (1) the vessel's safety is threatened, (2) the vessel is restricted in its ability to maneuver (e.g., during launching and recovery of aircraft or landing craft, during towing activities, when mooring, during Transit Protection Program exercises or other events involving escort vessels), (3) the vessel is submerged or operated autonomously, or (4) when impractical based on mission requirements (e.g., during test body retrieval by range craft).
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout on the vessel that is underway
Mitigation Requirements • Mitigation zones: ○ 500 yards (457 meters) around whales ○ 200 yards (183 meters) (for surface ships) around other marine mammals (except bow-riding dolphins and pinnipeds hauled out man-made navigational structures, port structures, and vessels) ○ 100 yards (91 meters) (for small boats, such as range craft) around other marine mammals (except bow-riding dolphins and pinnipeds hauled out man-made navigational structures, port structures, and vessels) ○ Within the vicinity of sea turtles • During the activity: ○ When underway, observe the mitigation zone for marine mammals and sea turtles; if observed, maneuver to maintain distance. • Additional requirements:

Procedural Mitigation Description
○ If a marine mammal or sea turtle vessel strike occurs, the Navy will follow the established incident reporting procedures.

3.5.1.12 Towed In-water Devices

The Navy will implement procedural mitigation to avoid or reduce the potential for strike of marine mammals and sea turtles from towed in-water devices, as outlined in Table 37. The small mitigation zone size and proximity to the observation platform will result in a high likelihood that Lookouts will be able to detect marine mammals throughout the mitigation zone when manned vessels or manned aircraft are towing in-water devices.

Table 37. Procedural mitigation for towed in-water devices.

Procedural Mitigation Description
Stressor or Activity • Towed in-water devices ○ Mitigation applies to devices towed from a manned surface platform or manned aircraft, or when a manned support craft is already participating in an activity involving in-water devices being towed by unmanned platforms ○ The mitigation will not be applied if the safety of the towing platform or in-water device is threatened
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned on the towing platform or support craft
Mitigation Requirements • Mitigation zones: ○ 250 yards (229 meters) (for in-water devices towed by aircraft or surface ships) around marine mammals (except bow-riding dolphins and pinnipeds hauled out on man-made navigational structures, port structures, and vessels) ○ 100 yards (91 meters) (for in-water devices towed by small boats, such as range craft) around marine mammals (except bow-riding dolphins and pinnipeds hauled out on man-made navigational structures, port structures, and vessels). ○ Within the vicinity of sea turtles • During the activity (i.e., when towing an in-water device) ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, maneuver to maintain distance.

3.5.1.13 Small-, Medium-, and Large-Caliber Non-Explosive Practice Munitions

The Navy will implement procedural mitigation to avoid or reduce the potential for strike of marine mammals and sea turtles from small-, medium-, and large-caliber non-explosive practice munitions, as outlined in Table 38. The mitigation zone is designed to be several times larger than the impact footprint for large-caliber non-explosive practice munitions, which are the largest projectiles used for these activities. Small-caliber and medium-caliber non-explosive practice munitions have smaller impact footprints than large-caliber non-explosive practice

munitions; therefore, the mitigation zone will extend even further beyond the impact footprints for these smaller projectiles.

Large-caliber gunnery activities involve vessels firing projectiles at a target located up to six nautical miles down range. Small- and medium-caliber gunnery activities involve vessels or aircraft firing projectiles at targets located up to 4,000 yards (3,658 meters) down range, although typically much closer. Lookouts will have a better likelihood of detecting marine mammals and sea turtles when observing mitigation zones around targets located close to the firing platform. When observing activities that use a target located far from the firing platform, Lookouts will be more likely to detect large visual cues (e.g., whale blows, breaching whales, or large pods of dolphins) than individual marine mammals, cryptic marine mammal species, and sea turtles.

Table 38. Procedural mitigation for small-, medium-, and large-caliber non-explosive practice munitions.

Procedural Mitigation Description
Stressor or Activity • Gunnery activities using small-, medium-, and large-caliber non-explosive practice munitions   ○ Mitigation applies to activities using a surface target
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned on the platform conducting the activity   ○ Depending on the activity, the Lookout could be the same as the one described in Table 28 for Weapons Firing Noise.

Mitigation Requirements
• Mitigation zone:
  ○ 200 yards (183 meters) around the intended impact location
• Prior to the initial start of the activity (e.g., when maneuvering on station):
  ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear.
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of firing.
• During the activity:
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease firing.
• Commencement/recommencement conditions after a marine mammal or sea turtle sighting before or during the activity:
  ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing firing) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended impact location; (3) the mitigation zone has been clear from any additional sightings for 10 min. for aircraft-based firing or 30 min. for vessel-based firing; or (4) for activities using a mobile target, the intended impact location has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting.

3.5.1.14 Non-Explosive Missiles

The Navy will implement procedural mitigation to avoid or reduce the potential for strike of marine mammals and sea turtles from non-explosive missiles, as outlined in Table 39. The mitigation zone for non-explosive missiles and rockets is designed to be several times larger than the impact footprint for the largest non-explosive missile used for these activities. Smaller non-explosive missiles have smaller impact footprints than the largest non- explosive missile used for these activities; therefore, the mitigation zone will extend even further beyond the impact footprints for these smaller projectiles.

Mitigation applies to activities using non-explosive missiles fired from aircraft at targets that are typically located up to 15 nautical miles down range, and infrequently up to 75 nautical miles down range. There is a chance that animals could enter the mitigation zone after the aircraft conducts its close-range mitigation zone observations and before firing begins (once the aircraft has transited to its firing position). Due to the distance between the mitigation zone and the observation platform, Lookouts will have a better likelihood of detecting marine mammals and sea turtles during the close-range observations and are less likely to detect these resources once positioned at the firing location, particularly individual marine mammals, cryptic marine mammal species, and sea turtles.

Table 39. Procedural mitigation for non-explosive missiles.

Procedural Mitigation Description
Stressor or Activity • Aircraft-deployed non-explosive missiles   ○ Mitigation applies to activities using a surface target
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in an aircraft

Mitigation Requirements
• Mitigation zone:
  ○ 900 yards (823 meters) around the intended impact location
• Prior to the initial start of the activity (e.g., during a fly-over of the mitigation zone):
  ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear.
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of firing.
• During the activity:
  ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease firing.
• Commencement/recommencement conditions after a marine mammal or sea turtle sighting prior to or during the activity:
  ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing firing) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement

Procedural Mitigation Description
relative to the intended impact location; or (3) the mitigation zone has been clear from any additional sightings for 10 min. when the activity involves aircraft that have fuel constraints, or 30 min. when the activity involves aircraft that are not typically fuel constrained.

3.5.1.15 Non-Explosive Bombs and Mine Shapes

The Navy will implement procedural mitigation to avoid or reduce the potential for strike of marine mammals and sea turtles from non-explosive bombs and mine shapes, as outlined in Table 40. The mitigation zone for non-explosive bombs and mine shapes is designed to be several times larger than the impact footprint for the largest non-explosive bomb used for these activities. Smaller non-explosive bombs and mine shapes have smaller impact footprints than the largest non- explosive bomb used for these activities; therefore, the mitigation zone will extend even further beyond the impact footprints for these smaller military expended materials.

Activities involving non-explosive bombing and mine laying involve aircraft deploying munitions or mine shapes from a relatively steady altitude of approximately 1,500 feet at a surface target or in an intended minefield located beneath the aircraft. Due to the mitigation zone size, proximity to the observation platform, and the good vantage point from an aircraft, Lookouts will be able to observe the entire mitigation zone during approach of the target or intended minefield location.

Table 40. Procedural mitigation for non-explosive bombs and mine shapes.

Procedural Mitigation Description
Stressor or Activity • Non-explosive bombs • Non-explosive mine shapes during mine laying activities
Resource Protection Focus • Marine mammals • Sea turtles
Number of Lookouts and Observation Platform • 1 Lookout positioned in an aircraft
Mitigation Requirements • Mitigation zone: ○ 1,000 yards (914 meters) around the intended target • Prior to the initial start of the activity (e.g., when arriving on station): ○ Observe the mitigation zone for floating vegetation; if observed, relocate or delay the start until the mitigation zone is clear. ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, relocate or delay the start of bomb deployment or mine laying. • During the activity (e.g., during approach of the target or intended minefield location): ○ Observe the mitigation zone for marine mammals and sea turtles; if observed, cease bomb deployment or mine laying. • Commencement/recommencement conditions after a marine mammal or sea turtle sighting prior to or during the activity:

Procedural Mitigation Description
Mitigation Requirements (Continued) ○ The Navy will allow a sighted marine mammal or sea turtle to leave the mitigation zone prior to the initial start of the activity (by delaying the start) or during the activity (by not recommencing bomb deployment or mine laying) until one of the following conditions has been met: (1) the animal is observed exiting the mitigation zone; (2) the animal is thought to have exited the mitigation zone based on a determination of its course, speed, and movement relative to the intended target or minefield location; (3) the mitigation zone has been clear from any additional sightings for 10 min.; or (4) for activities using mobile targets, the intended target has transited a distance equal to double that of the mitigation zone size beyond the location of the last sighting.

3.5.2 Mitigation Areas

In addition to procedural mitigation, the Navy will implement mitigation measures within specified areas to avoid potential impacts on ESA-listed species and seafloor resources (which serve valuable ecosystem functions and provide habitat for ESA-listed species and their prey). Mitigation areas are geographic locations in the NWTT Study Area where the Navy will implement additional avoidance and minimization measures during all or a part of the year (mitigation applies year-round unless specified otherwise). Should national security present a requirement to conduct activities that the Navy would otherwise prohibit in a particular mitigation area, naval units will obtain permission from the appropriate designated Command authority prior to commencement of the activity. The Navy will provide NMFS with advance notification and include information about the event in its annual activity reports to NMFS.

The Navy considered several factors when determining the location of proposed geographic mitigation areas. First, they evaluated whether the mitigation area would be effective in reducing impacts to resources of biological or ecological importance. Next, the Navy operational community assessed how and to what degree implementation of mitigation measures would be compatible with planning, scheduling, and conducting proposed training and testing activities. A more thorough discussion on the factors used by the Navy to determine which areas to propose for geographic mitigation is provided in the NWTT FSEIS (Navy 2019b).

3.5.2.1 Mitigation Areas for Seafloor Resources

The Navy proposes to implement mitigation to avoid and minimize impacts to seafloor resources from explosives, physical disturbance, and strike stressors in mitigation areas throughout the NWTT study area (Table 41). Seafloor resource mitigation would help the Navy avoid or reduce impacts from explosives, physical disturbance, and strike stressors on seafloor resources, and consequently to any ESA-protected resources that inhabit, shelter, rest, feed, or occur in the mitigation areas.

The Navy developed mitigation areas as either the anchor swing circle diameter or a 350-yard (320 meter) radius around a seafloor resource, as indicated by the best available georeferenced

data. To facilitate mitigation implementation, the Navy will include maps of the best available georeferenced data for artificial reefs, live hard bottom, and shipwrecks in its Protective Measures Assessment Protocol. Mitigation areas apply to georeferenced resources because the Navy requires accurate resource identification and mapping for the mitigation to be effective and practical to implement. Mitigating within the anchor swing circle will protect seafloor resources during precision anchoring activities when factoring in environmental conditions that could affect anchoring position and swing circle size, such as winds, currents, and water depth.

For other activities applicable to the mitigation, a 350-yard (320-meter) radius around a seafloor resource is a conservatively sized mitigation area that will provide protection well beyond the maximum expected impact footprint (e.g., crater and expelled material radius) of the explosives and non-explosive practice munitions used in the NWTT Study Area. The mitigation zone size extends beyond the military expended material with the largest footprint for all areas where this mitigation measure would be implemented. For example, the military expended material with the largest footprint (which is not used in the NWTT Study Area) is an explosive mine with a 650 pounds NEW, which has an estimated impact footprint of approximately 14,800 square feet and an associated radius of 22.7 yards (21 meters). The largest explosive applicable to this mitigation in the NWTT Study Area has a charge size of 60 pounds NEW. Therefore, the 350-yard (320-meter) mitigation zone is well beyond the maximum expected direct impact footprint for the activities that will implement the seafloor resource mitigation in the NWTT Study Area, and further mitigates some level of indirect impact from explosive disturbances. Other applicable explosive activities and non-explosive practice munitions have a smaller impact footprint; therefore, the mitigation area will result in additional protection during those activities.

Table 41. Mitigation areas for seafloor resources.

Mitigation Area Description
Stressor or Activity • Explosives • Physical disturbance and strikes
Mitigation Requirements • **Seafloor Resource Mitigation Areas (year-round)** o Within the anchor swing circle of live hard bottom, artificial reefs, and shipwrecks, the Navy will not conduct precision anchoring (except in designated areas). o Within a 350-yard (320-meter) radius of live hard bottom, artificial reefs, and shipwrecks, the Navy will not conduct explosive mine countermeasure and neutralization activities or explosive mine neutralization activities involving Navy divers (except in designated locations), and the Navy will not place mine shapes, anchors, or mooring devices on the seafloor (except in designated areas).

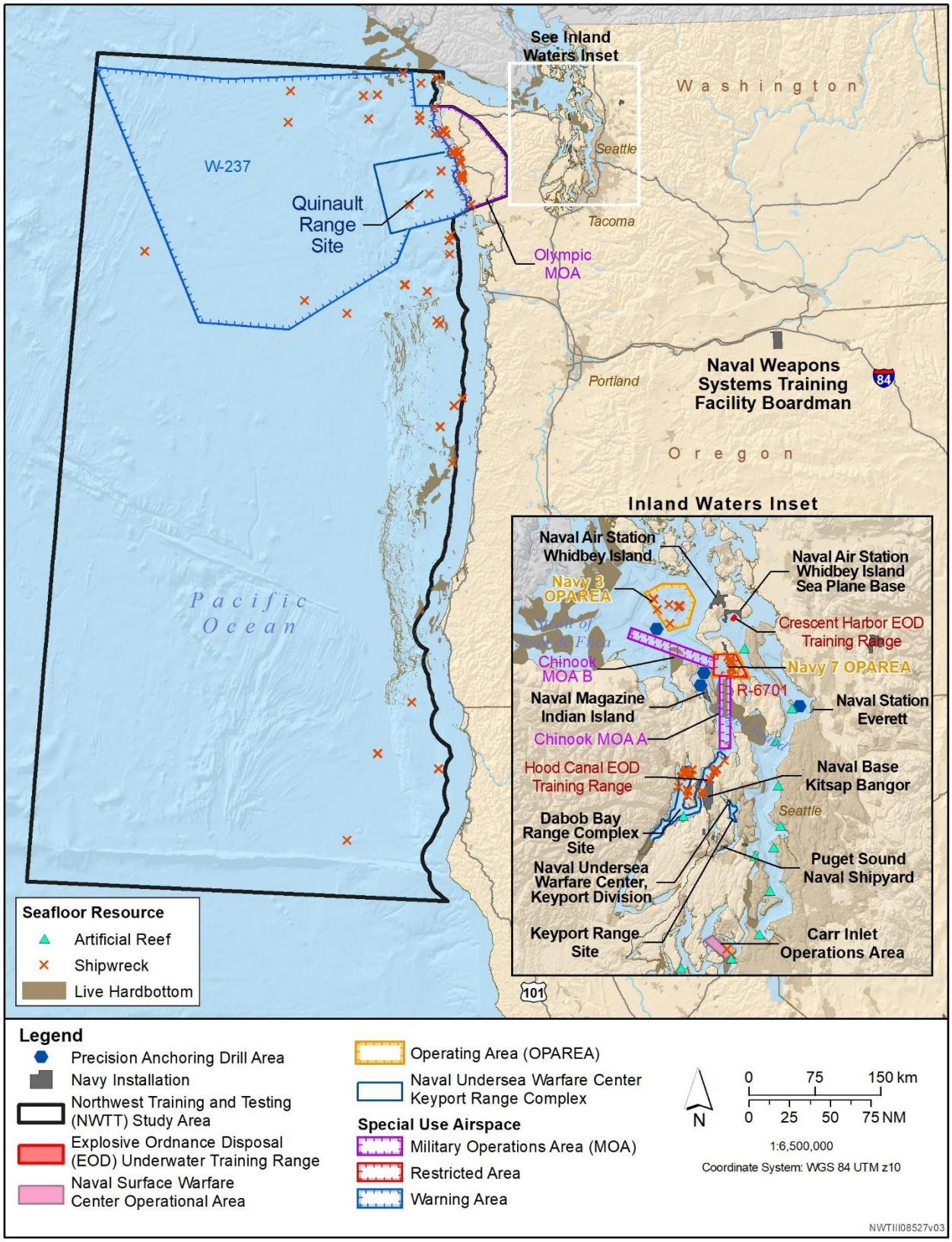


Figure 27. Seafloor resource mitigation areas in the NWTT Study Area.

3.5.2.2 Marine Species Mitigation Areas

Mitigation areas the Navy developed for the purpose of avoiding or reducing potential impacts from the Proposed Action on marine species are shown in Figure 28 and detailed in Table 42. Mitigation benefits for each area are presented in the sections below.

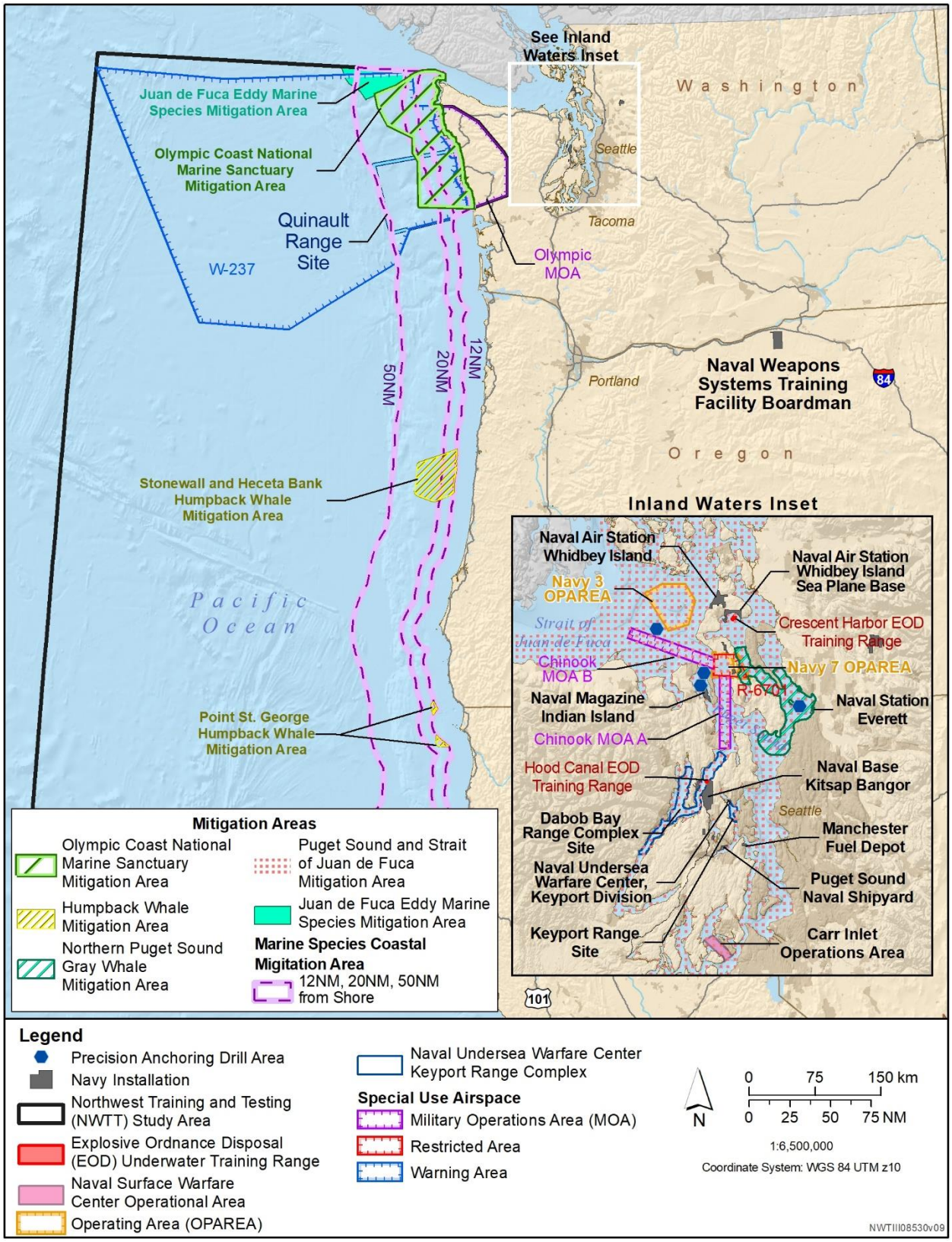


Figure 28. Marine species mitigation areas in the NWTT Study Area.

Table 42. Marine species mitigation areas in the offshore NWTT Study Area.

Mitigation Area Description
Stressor or Activity • Sonar (mitigation does not apply to active sonar sources used for safety of navigation) • Explosives • Physical disturbance and strikes
Resource Protection Focus • Marine mammals (humpback whale, gray whale, Southern Resident killer whale) • Sea turtles (leatherback sea turtle) • Fish (steelhead, Chinook salmon, coho salmon, chum salmon, sockeye salmon, green sturgeon)
Mitigation Requirements[1] • **Marine Species Coastal Mitigation Area (year-round or seasonal if specified)** o Within 50 NM from shore in the Marine Species Coastal Mitigation Area: ▪ The Navy will not conduct explosive training activities. ▪ The Navy will not conduct explosive testing activities (except explosive Mine Countermeasure and Neutralization Testing). ▪ The Navy will not conduct non-explosive missile training activities. ▪ The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft to the possible presence of increased concentrations of Southern Resident killer whales from December 1 to June 30, humpback whales from May 1 through December 31, and gray whales from May 1 to November 30. For safe navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of Southern Resident killer whales, humpback whales, and gray whales that may be vulnerable to vessel strikes or potential impacts from training and testing activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training and testing activities and to aid in the implementation of procedural mitigation. o Within 20 NM from shore in the Marine Species Coastal Mitigation Area: ▪ The Navy will conduct a maximum combined total of 33 hours of surface ship hull-mounted MF1 mid-frequency active sonar during testing annually within 20 NM from shore in the Marine Species Coastal Mitigation Area, the Juan de Fuca Eddy Marine Species Mitigation Area, and the Olympic Coast National Marine Sanctuary Mitigation Area. ▪ To the maximum extent practical, the Navy will conduct explosive Mine Countermeasure and Neutralization Testing from July 1 through September 30 when operating within 20 NM from shore. ▪ From October 1 through June 30, the Navy will conduct a maximum of one explosive Mine Countermeasure and Neutralization Testing event, not to exceed the use of 20 explosives from bin E4 and 3 explosives from bin E7 annually, and not to exceed the use of 60 explosives from bin E4 and 9 explosives from bin E7 over 7 years. ▪ The Navy will not conduct non-explosive large-caliber gunnery training activities. ▪ The Navy will not conduct non-explosive bombing training activities.
o Within 12 NM from shore in the Marine Species Coastal Mitigation Area: ▪ The Navy will not conduct Anti-Submarine Warfare Tracking Exercise – Helicopter, – Maritime Patrol Aircraft, – Ship, or – Submarine training activities (which involve the use of mid-frequency or high-frequency active sonar). ▪ The Navy will not conduct non-explosive Anti-Submarine Warfare Torpedo Exercise – Submarine training activities (which involve the use of mid-frequency or high-frequency active sonar). ▪ The Navy will conduct a maximum of one Unmanned Underwater Vehicle Training event within 12 NM from shore at the Quinault Range Site. Unmanned Underwater Vehicle Training events at the Quinault Range Site will be cancelled or moved to another training location if Southern Resident **Mitigation Requirements[1](Continued)**

Mitigation Area Description
▪ killer whales are detected at the planned training location during the event planning process, or immediately prior to the event, as applicable. ▪ During explosive Mine Countermeasure and Neutralization Testing, the Navy will not use explosives in bin E7 closer than 6 NM from shore in the Quinault Range Site. ▪ The Navy will not conduct non-explosive small- and medium-caliber gunnery training activities. • **Olympic Coast National Marine Sanctuary Mitigation Area (year-round)** o Within the Olympic Coast National Marine Sanctuary Mitigation Area: ▪ The Navy will conduct a maximum of 32 hours of surface ship hull-mounted MF1 mid-frequency active sonar during training annually. ▪ The Navy will conduct a maximum combined total of 33 hours of surface ship hull-mounted MF1 mid-frequency active sonar during testing annually within 20 NM from shore in the Marine Species Coastal Mitigation Area, the Juan de Fuca Eddy Marine Species Mitigation Area, and the Olympic Coast National Marine Sanctuary Mitigation Area. ▪ The Navy will not conduct explosive Mine Countermeasure and Neutralization Testing activities. ▪ The Navy will not conduct non-explosive bombing training activities. • **Juan de Fuca Eddy Marine Species Mitigation Area (year-round)** o Within the Juan de Fuca Eddy Marine Species Mitigation Area: ▪ The Navy will conduct a maximum combined total of 33 hours of surface ship hull-mounted MF1 mid-frequency active sonar during testing annually within 20 NM from shore in the Marine Species Coastal Mitigation Area, the Juan de Fuca Eddy Marine Species Mitigation Area, and the Olympic Coast National Marine Sanctuary Mitigation Area ▪ The Navy will not conduct explosive Mine Countermeasure and Neutralization Testing activities.
• **Stonewall and Heceta Bank Humpback Whale Mitigation Area (May 1–November 30)** o Within the Stonewall and Heceta Bank Humpback Whale Mitigation Area from May 1 to November 30: ▪ The Navy will not use surface ship hull-mounted MF1 mid-frequency active sonar during training or testing. ▪ The Navy will not conduct explosive Mine Countermeasure and Neutralization Testing. • **Point St. George Humpback Whale Mitigation Area (July 1–November 30)** o Within the Point St. George Humpback Whale Mitigation Area from July 1 to November 30: ▪ The Navy will not use surface ship hull-mounted MF1 mid-frequency active sonar during training or testing. ▪ The Navy will not conduct explosive Mine Countermeasure and Neutralization Testing.
[1] *Should national security present a requirement to conduct training or testing prohibited by the mitigation requirements specified in this table, naval units will obtain permission from the appropriate designated Command authority prior to commencement of the activity. The Navy will provide NMFS with advance notification and include relevant information about the event (e.g., sonar hours, explosives use, non-explosive practice munitions use) in its annual activity reports to NMFS.*

Table 43. Marine species mitigation areas in the NWTT Inland Waters.

Mitigation Area Description
Stressor or Activity • Sonar (mitigation does not apply to active sonar sources used for safety of navigation) • Explosives • Physical disturbance and strikes
Resource Protection Focus • Marine mammals (gray whale, Southern Resident killer whale) • Fish (Puget Sound Chinook salmon, Hood Canal summer-run chum salmon, green sturgeon, rockfish)
Mitigation Requirements[1] • **Northern Puget Sound Gray Whale Mitigation Area (March 1–May 31)** o Within the Northern Puget Sound Gray Whale Mitigation Area from March 1 to May 31: ▪ The Navy will not conduct Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises. • **Puget Sound and Strait of Juan de Fuca Mitigation Area (year-round or seasonal if specified)** o Within the Puget Sound and Strait of Juan de Fuca Mitigation Area: ▪ The Navy will not use low-frequency, mid-frequency, or high-frequency active sonar during training or testing within the Puget Sound and Strait of Juan de Fuca Mitigation Area, unless a required element necessitates that the activity be conducted in NWTT Inland Waters during (1) Unmanned Underwater Vehicle Training, (2) Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises, (3) activities conducted by Naval Sea Systems Command at designated locations, and (4) pierside sonar maintenance or testing at designated locations. ▪ The Navy will use the lowest active sonar source levels practical to successfully accomplish each event. ▪ Naval units will obtain permission from the appropriate designated Command authority prior to commencing pierside maintenance or testing with hull-mounted mid-frequency active sonar. ▪ The Navy will conduct a maximum of one Unmanned Underwater Vehicle Training activity annually at the Navy 3 OPAREA, Navy 7 OPAREA, and Manchester Fuel Depot (i.e., a maximum of one event at each location). ▪ The Navy will not use explosives during testing. ▪ The Navy will not use explosives during training except at the Hood Canal EOD Range and Crescent Harbor EOD Range during explosive mine neutralization activities involving the use of Navy divers. ▪ The Navy will not use explosives in bin E4 (>2.5–5 pounds net explosive weight) or above, and will instead use explosives in bin E0 (< 0.1 pounds net explosive weight) or bin E3 (> 0.5–2.5 pounds net explosive weight). ▪ During February, March, and April at the Hood Canal EOD Range, the Navy will not use explosives in bin E3 (> 0.5–2.5 pounds net explosive weight), and will instead use explosives in bin E0 (< 0.1 pounds net explosive weight). ▪ During August, September, and October at the Hood Canal EOD Range, the Navy will avoid using explosives in bin E3 (> 0.5–2.5 pounds net explosive weight) and will instead use explosives in bin E0 (< 0.1 pounds net explosive weight) to the maximum extent practical unless necessitated by mission requirements. ▪ At the Crescent Harbor EOD Range, the Navy will conduct explosive activities at least 1,000 m from the closest point of land. ▪ The Navy will not conduct non-explosive live fire events in the mitigation area (except firing blank weapons), including gunnery exercises, missile exercises, torpedo exercises, bombing exercises, and Kinetic Energy Weapon Testing.
▪ Navy event planners will coordinate with Navy biologists during the event planning process prior to conducting (1) Unmanned Underwater Vehicle Training at the NAVY 3 OPAREA, Manchester Fuel Depot, Crescent Harbor Explosive Ordnance Disposal Range, and NAVY 7 OPAREA (for Southern

Mitigation Area Description
Resident killer whales), (2) Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises (for Southern Resident killer whales and gray whales), (3) explosive mine neutralization activities involving the use of Navy divers (for Southern Resident killer whales), and (4) Small Boat Attack Exercises, which involve firing blank small-caliber weapons (for Southern Resident killer whales and gray whales). Navy biologists will work with NMFS and will initiate communication with the appropriate marine mammal detection networks to determine the likelihood of applicable marine mammal species presence in the planned training location. Navy biologists will notify event planners of the likelihood of species presence. To the maximum extent practical, Navy planners will use this information when planning specific details of the event (e.g., timing, location, duration) to avoid planning activities in locations or seasons where species presence is expected. The Navy will ensure environmental awareness of event participants. Environmental awareness will help alert participating crews to the possible presence of applicable species in the training location. Lookouts will use the information to assist visual observation of applicable mitigation zones and to aid in the implementation of procedural mitigation. Unmanned Underwater Vehicle Training events at the NAVY 3 OPAREA, Manchester Fuel Depot, Crescent Harbor Explosive Ordnance Disposal Range, and NAVY 7 OPAREA will be cancelled or moved to another training location if the presence of Southern Resident killer whales is reported through available monitoring networks during the event planning process, or immediately prior to the event, as applicable. ▪ The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft operating within the Puget Sound and Strait of Juan de Fuca Mitigation Area to the possible presence of concentrations of Southern Resident killer whales from July 1 to November 30 in the Puget Sound and Strait of Juan de Fuca, and concentrations of gray whales from March 1 to May 31 in the Strait of Juan de Fuca and northern Puget Sound. For safe navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of Southern Resident killer whales and gray whales that may be vulnerable to vessel strikes or potential impacts from training and testing activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training and testing activities and to aid in the implementation of procedural mitigation.
[1] Should national security present a requirement to conduct training or testing prohibited by the mitigation requirements specified in this table, naval units will obtain permission from the appropriate designated Command authority prior to commencement of the activity. The Navy will provide NMFS with advance notification and include relevant information about the event (e.g., sonar hours, explosives use, non-explosive practice munitions use) in its annual activity reports to NMFS.

3.5.2.2.1 Marine Species Coastal Mitigation Area

The Marine Species Coastal Mitigation Area is designed to avoid or reduce potential impacts from explosives, non-explosive practice munitions, and active sonar on marine species that inhabit, feed in, or migrate through this area, including killer whales, humpback whales, and gray whales. As shown in Table 42, the Navy will implement mitigation for applicable activities within 50, 20, or 12 nautical miles from shore in the at-sea portion of the NWTT Study Area (See Section 4.1) (including to the Washington shoreline in the northern portion of the offshore area). Mitigation within 12 nautical miles and 20 nautical miles from shore in the Marine Species Coastal Mitigation Area is a continuation from the 2015 NWTT biological opinion. Mitigation within 50 nautical miles from shore in the Marine Species Coastal Mitigation Area is also a

continuation from the 2015 NWTT biological opinion for training activities. Mitigation requirements within 50 nautical miles from shore have been expanded to include explosive testing activities (except explosive mine countermeasure and neutralization testing activities).

The 50 nautical mile from shore portion of the Marine Species Coastal Mitigation Area overlaps important feeding, migration, or critical habitat for humpback whales, gray whales, Southern Resident killer whales, steelhead, Chinook salmon, coho salmon, chum salmon, sockeye salmon, green sturgeon, and leatherback sea turtles. Mitigation within 50 nautical mile from shore will result in an avoidance of potential impacts on marine mammals, ESA-listed fish, and leatherback sea turtles within their important habitat areas from all explosive training activities, all explosive testing activities except explosive mine countermeasure and neutralization testing activities, and non-explosive missile training exercises. The mitigation requirements will also consequently help the Navy avoid potential impacts from active sonar used in conjunction with applicable explosive events that are required to be conducted greater than 50 nautical miles from shore, such as mid-frequency and high-frequency active sonar used during explosive torpedo events (e.g., MF1 and MF4 sonar during Torpedo [Explosive] Testing). As a new mitigation measure for NWTT Phase III, the Navy will issue annual seasonal awareness notification messages to further help avoid potential impacts from vessel strikes and training and testing activities on humpback whales, gray whales, and Southern Resident killer whales in the Marine Species Coastal Mitigation Area. The awareness notification messages will coincide with the seasons in which humpback whales, gray whales, and Southern Resident killer whales are most likely to be observed in concentrations in the mitigation area. Southern Resident killer whales are most likely to be observed in the NWTT Offshore Area in winter and spring (December 1 to June 30), which correlates with prey availability. Gray whales and humpback whales are most likely to be observed in the NWTT Offshore from late spring through fall (May 1 to November 30 and May 1 through December 31, respectively), which correlates to feeding or migration seasons.

The 20 nautical mile from shore portion of the Marine Species Coastal Mitigation Area overlaps important feeding, migration, or critical habitat for gray whales, humpback whales, Southern Resident killer whales, leatherback sea turtles, Chinook salmon, coho salmon, chum salmon, sockeye salmon, steelhead, and green sturgeon. Mitigation requirements within 20 nautical miles from shore will result in an avoidance or reduction of potential impacts from surface ship hull-mounted MF1 mid-frequency active sonar, non-explosive large-caliber gunnery training, and non-explosive bombing training on marine species within these habitats. With regard to explosive mine countermeasure and neutralization testing, proposed mitigation would reduce the maximum potential exposure to explosives in bin E4 and bin E7 by approximately 40 percent during the months when Southern DPS green sturgeon and several ESA-listed salmonid populations are expected to be present in higher densities. Similarly, mitigation to conduct this activity from July 1 through September 30 to the maximum extent practical when operating within 20 nautical miles from shore is designed primarily to avoid or reduce potential impacts on ESA-listed fish species based on their typical occurrence seasonally and at certain water depths.

This mitigation would also likely benefit foraging or migrating humpback whales, migrating gray whales, and foraging or transiting Southern Resident killer whales.

The 12 nautical mile from shore portion of the Marine Species Coastal Mitigation Area overlaps important feeding, migration, or critical habitats for gray whales, humpback whales, Southern Resident killer whales, leatherback sea turtles, Chinook salmon, coho salmon, chum salmon, sockeye salmon, steelhead, and green sturgeon. Mitigation requirements within 12 nautical miles from shore will result in an avoidance or reduction of potential impacts from non-explosive small- and medium-caliber gunnery training, nonexplosive torpedo training (which involves mid-frequency and high-frequency active sonar), and Anti-Submarine Warfare Tracking Exercise – Helicopter, Maritime Patrol Aircraft, Ship, or Submarine training activities (which involve mid-frequency active sonar [including surface ship hull-mounted MF1 mid-frequency active sonar and MF4 dipping sonar] and high-frequency active sonar). Mitigation to conduct a maximum of one Unmanned Underwater Vehicle Training event within 12 nautical miles from shore at the Quinault Range Site, and to cancel or move Unmanned Underwater Vehicle Training events if Southern Resident killer whales are detected within 12 nautical miles from shore at the Quinault Range Site, is expected to help the Navy avoid any potential impacts on Southern Resident killer whales during Unmanned Underwater Vehicle Training events. Mitigation during explosive mine countermeasure and neutralization testing to not use explosives in bin E7 closer than 6 nautical miles from shore in the Quinault Range Site may avoid or minimize overlap of the larger of the explosive bins used in this activity with ESA-listed fish species. The Navy's combined mitigation within the Marine Species Coastal Mitigation Area will result in all live fire training activities being conducted at least 12 nautical miles from shore, with many activities conducted beyond 20 or 50 nautical miles from shore, as described previously.

3.5.2.2.2 Olympic Coast National Marine Sanctuary Mitigation Area

Mitigation within the Olympic Coast National Marine Sanctuary is designed to avoid or reduce potential impacts from mid-frequency active sonar, explosives during mine countermeasure and neutralization testing activities, and non-explosive practice munitions on marine mammals that inhabit, feed in, or migrate through this area, including killer whales, humpback whales, and gray whales. The mitigation will also help the Navy avoid or reduce potential impacts on a wide assemblage of other resources that inhabit, forage in, and migrate through the sanctuary, such as leatherback sea turtles and fish.

Some proposed mitigation measures within the Olympic Coast National Marine Sanctuary Mitigation Area are a continuation from the 2015 NWTT biological opinion. Newly proposed mitigation includes annual restrictions on the use of surface ship hull-mounted MF1 mid-frequency active sonar during testing in three combined mitigation areas: within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area, the new Juan de Fuca Eddy Marine Species Mitigation Area, and within the Olympic Coast National Marine Sanctuary Mitigation Area. The annual restriction for testing previously only applied to the Olympic Coast

National Marine Sanctuary Mitigation Area. Furthermore, the Navy removed an exception in the mitigation language that excluded the Quinault Range Site from the annual sonar restrictions. Now, the Navy's annual restrictions will apply throughout the entire Olympic Coastal National Marine Sanctuary Mitigation Area, including within the portion of the mitigation area that overlaps the Quinault Range Site.

Because the Olympic Coast National Marine Sanctuary Mitigation Area is located entirely within 50 nautical miles from shore in the Marine Species Coastal Mitigation Area, the Navy's combined mitigation will ensure that marine mammals and other sanctuary resources are not exposed to explosives in the sanctuary from any training or testing activity under the Proposed Action. Furthermore, mitigation within 20 nautical miles and 12 nautical miles from shore in the Marine Species Coastal Mitigation Area will help the Navy avoid or reduce potential impacts from active sonar and non-explosive practice munitions on sanctuary resources.

3.5.2.2.3 Stonewall and Heceta Bank Humpback Whale Mitigation Area

The Stonewall and Heceta Bank Humpback Whale Mitigation Area is new for the Proposed Action. Mitigation within the Stonewall and Heceta Bank Humpback Whale Mitigation Area is designed to avoid or reduce potential impacts seasonally from mid-frequency active sonar and explosive mine countermeasure and neutralization testing activities on humpback whales in an important feeding area. The mitigation will also help the Navy avoid or reduce potential impacts on other marine mammals that may inhabit or migrate through this area, including killer whales and gray whales.

Because the Stonewall and Heceta Bank Humpback Whale Mitigation Area is located entirely within 50 nautical miles from shore within the Marine Species Coastal Mitigation Area, the Navy's combined mitigation will ensure that humpback whales and other marine mammals are not exposed to explosives in the mitigation area from any training or testing activity under the Proposed Action during the applicable season. The Stonewall and Heceta Bank Humpback Whale Mitigation Area is partially located within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area; therefore, the Navy's combined mitigation will help the Navy reduce potential impacts from non-explosive practice munitions on humpback whales and other marine mammals within the portion of overlap.

3.5.2.2.4 Point St. George Humpback Whale Mitigation Area

The Point St. George Humpback Whale Mitigation Area is new for the Proposed Action. Mitigation within the Point St. George Humpback Whale Mitigation Area is designed to avoid or reduce potential impacts seasonally from mid-frequency active sonar and explosive mine countermeasure and neutralization testing activities on humpback whales in an important feeding area. The mitigation will also help the Navy avoid or reduce potential impacts on other marine mammals that may inhabit or migrate through this area, including killer whales and gray whales.

Because the Point St. George Humpback Whale Mitigation Area is located entirely within 20 nautical miles from shore within the Marine Species Coastal Mitigation Area, the Navy's combined mitigation will ensure that humpback whales and other marine mammals are not exposed to explosives in the mitigation area from any training or testing activity under the Proposed Action during the applicable season. Additionally, mitigation in the Marine Species Coastal Mitigation Area will help the Navy reduce potential impacts from non-explosive practice munitions on humpback whales and other marine mammals.

3.5.2.2.5 Puget Sound and Strait of Juan de Fuca Mitigation Area

This mitigation area encompasses the full extent of the NWTT Inland Waters portion of the NWTT Study Area. Mitigation within the Puget Sound and Strait of Juan de Fuca Mitigation Area is designed to avoid or reduce potential impacts from active sonar on killer whales and gray whales. The mitigation will also help the Navy avoid or reduce potential impacts on other marine mammals that inhabit, feed in, or migrate through this area. Mitigation at the Hood Canal Explosive Ordnance Disposal (EOD) Range to prohibit explosives in bin E3 in February, March, and April is designed to reduce potential exposures and level of impacts on migrating juvenile Hood Canal summer-run chum salmon. Mitigation to avoid using explosives in bin E3 to the maximum extent practical in August, September, and October is designed to reduce potential exposures and level of impacts on adult migrating Puget Sound Chinook salmon and Hood Canal summer-run chum salmon. Mitigation to conduct explosive activities at least 1,000 meters from the closest point of land will avoid or reduce impacts on fish (e.g., juvenile Chinook salmon) in nearshore habitat areas. Mitigation to prohibit all live fire training and testing activities in the mitigation area, including gunnery exercises, missile exercises, torpedo exercises, bombing exercises, and Kinetic Energy Weapon Testing, will help the Navy avoid any impacts from explosives and non-explosive practice munitions on marine mammals and ESA-listed fish throughout NWTT Inland Waters.

Several mitigation measures within the Puget Sound and Strait of Juan de Fuca Mitigation Area are a continuation from the 2015 NWTT biological opinion. While conducting the NWTT Phase II action, the Navy, in practice, implemented several environmental protection measures that exceeded the mitigation requirements specified in the 2015 biological opinion. During consultations for the proposed action, the Navy determined it would be practical to codify those practices into formal mitigation area measures in NWTT Inland Waters for the Phase III proposed action. The Navy will implement the following mitigation area measures that are a continuation of current practice, but were not previously included in the 2015 biological opinion: 1) Requirements to not use active sonar during training or testing unless a required element necessitates the activity be conducted in NWTT Inland Waters; 2) Requirements to use the lowest active sonar source levels practical to successfully accomplish each event; 3) Requirements to not use explosives during testing; 4) Requirements to not use explosives during training except at the Hood Canal and Crescent Harbor EOD ranges, and for Navy event planners to coordinate with Navy biologists and NMFS, and initiate communication with the appropriate

marine mammal detection networks during the event planning process; and 5) Requirements to not conduct non-explosive live fire events (except firing blank weapons).

The Navy also identified the following opportunities to increase its mitigation measures applicable to the Puget Sound and Strait of Juan de Fuca Mitigation Area based on consultations and its ongoing analysis of the best available science: 1) Requirements to conduct a maximum of one Unmanned Underwater Vehicle Training activity annually at the NAVY 3 OPAREA, NAVY 7 OPAREA, and Manchester Fuel Depot; 2) Requirements for Navy event planners to coordinate with Navy biologists and NMFS during the event planning process prior to conducting Unmanned Underwater Vehicle Training at applicable locations, and to cancel or move events to another training location if the presence of Southern Resident killer whales is reported through available monitoring networks; 3) Requirements to initiate communication with the appropriate marine mammal detection networks prior to conducting Civilian Port Defense – Homeland Security AntiTerrorism/Force Protection Exercises and Small Boat Attack Exercises; and 4) Requirements to issue annual seasonal awareness notification messages to alert ships and aircraft operating within the Puget Sound and Strait of Juan de Fuca Mitigation Area to the possible presence of concentrations of Southern Resident killer whales and gray whales.

3.5.2.2.6 Northern Puget Sound Gray Whale Mitigation Area

The Northern Puget Sound Gray Whale Mitigation Area is new for the Proposed Action. Mitigation within the Northern Puget Sound Gray Whale Mitigation Area is designed to avoid or reduce potential impacts seasonally from active sonar on gray whales in an important feeding area. The mitigation will also help the Navy avoid or reduce potential impacts on other marine mammals that inhabit or migrate through this area seasonally.

3.5.2.2.7 Juan de Fuca Eddy Marine Species Mitigation Area

The Juan de Fuca Eddy Marine Species Mitigation Area is new for the proposed action. Mitigation within the Juan de Fuca Eddy Marine Species Mitigation Area is primarily designed to avoid or reduce potential impacts from surface ship hull-mounted MF1 mid-frequency active sonar and explosives during mine countermeasure and neutralization testing activities on Southern Resident killer whales and humpback whales within important feeding and migration habitats. Waters within the Juan de Fuca Eddy Marine Species Mitigation Area (including areas off Cape Flattery) are important foraging habitat for aggregations of humpback whales and migration habitat for Southern Resident killer whales as they transit between inland waters and the offshore area. The mitigation area is also potentially used by migrating gray whales, as well as other species of marine mammals, including sperm whales. The mitigation area's nutrient-rich waters and seasonal upwelling provide an abundance of marine mammal prey species and favorable foraging conditions for concentrations of marine mammals. The mitigation may also help avoid or reduce potential impacts on some populations of Chinook salmon, coho salmon, chum salmon, and steelhead. Additionally, the mitigation would result in the Navy avoiding any

overlap between explosive mine countermeasure and neutralization testing activities with the ESA-listed Ozette Lake ESU of sockeye salmon.

3.6 MMPA Regulations and Issuance of a Letter of Authorization

Section 7(b)(4)(C) of the ESA provides that if an endangered or threatened marine mammal is involved, the taking must first be authorized by Section 101(a)(5) of the MMPA. On March 11, 2019, NMFS' Permits Division received an application from the Navy requesting regulations and a LOA for the take of marine mammals incidental to Navy training and testing activities to be conducted in the NWTT Study Area over seven years. Ten of the marine mammal species in the LOA are also ESA-listed species (or DPSs). The Navy requested regulations that would establish a process for authorizing take, via a seven-year LOA, of marine mammals for training and testing activities proposed to be conducted from November 2020 through November 2027.

The Permits Division proposes to promulgate regulations pursuant to the MMPA, as amended (16 U.S.C. 1361 et seq.) for the Navy to "take" marine mammals incidental to NWTT activities from November 2020 through November 2027. The regulations propose to authorize the issuance of a LOA that will allow the Navy to "take" marine mammals incidental to their training and testing activities. Issuance of the LOA was dependent on a determination that the total number of marine mammals taken by the activity as a whole would have no more than a negligible impact on the affected species or stock of marine mammals. NMFS has defined negligible impact in 50 C.F.R. §216.103 as "an impact resulting from the specified activity that cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species or stock through effects on annual rates of recruitment or survival."

The Permits Division's proposed regulations are available at the following website: https://www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities. This consultation considers the proposed MMPA regulations for the Navy to "take" marine mammals incidental to NWTT activities, as modified during ESA consultation (NMFS 2020e). The final MMPA regulations, upon publication, will also be available at the website shown above. Note that this biological opinion was completed prior to the publication of the final MMPA regulations in the Federal Register. We anticipate that, upon publication, the MMPA regulations will reflect the mitigation and monitoring measures proposed by the Navy and/or agreed to during ESA consultation (a description of the mitigation measures is in Section 3.5 of this opinion). We will also review mitigation measures imposed by the final regulations to ensure they are consistent with measures to avoid and minimize take prescribed in the ITS. We also anticipate that the levels of take of ESA-listed marine mammals authorized under the final MMPA regulations and LOA will be consistent with those analyzed in this opinion. Upon publication, we will review the MMPA regulations to ensure these conditions are met. If administrative changes are needed following publication of the MMPA regulations, we will update the biological opinion to reflect these changes. If more substantive changes (e.g., those related to the effects analyses, take authorization, and/or

avoidance and minimization measures) are needed, the reinitiation triggers described in Section 14 may apply.

4 ACTION AREA

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02). The action area encompasses the NWTT Study Area, transit corridor, and the area outside of the Study Area where direct and indirect effects of stressors from Navy training and testing activities could be experienced. Figure 29 provides an overview of the action area which includes the inshore and offshore portions of the Northwest Training Range Complex (NWTRC) in Northern California, Oregon, and Washington State, the Naval Undersea Warfare Center (NUWC) Keyport Range Complex in and around Puget Sound, and the Southeast Alaska Acoustic Measurement Facility (SEAFAC) in the Western Behm Canal, Alaska.

The Navy's activities would occur within the boundaries depicted in Figure 29, so that the effects of the action, including effects from sonar and explosives, would not extend beyond these boundaries. Below we describe, in greater detail, the offshore portion of the NWTRC, the inshore portion of the NWTRC (Keyport Range Complex), and SEAFAC.

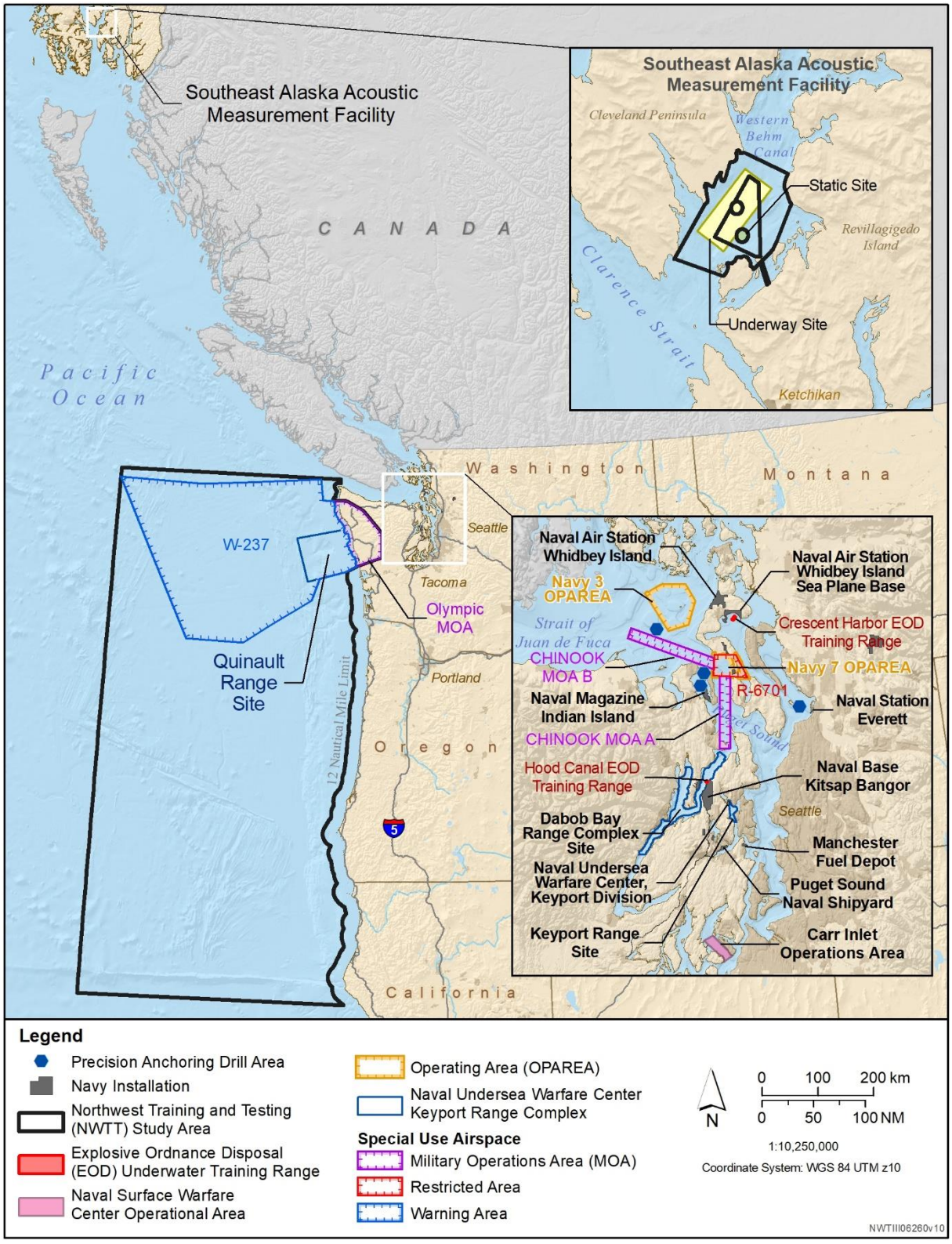


Figure 29. Action area for the Northwest Training and Testing (NWTT) Phase III proposed action.

4.1 Offshore Portion of the Action Area

The offshore component of the action area includes air, surface, and subsurface operating areas extending generally west from the coastline of Washington, Oregon, and Northern California for a distance of approximately 250 nautical miles into international waters (Figure 30). The eastern boundary of the offshore area lays 12 nautical miles off the coastline for most of the action area, including southern Washington, Oregon, and Northern California. The offshore area includes the ocean all the way to three nautical miles off the coastline only along the Washington coast beneath the airspace of W-237 which includes a surf zone component. The surf zone component extends north to south 5 nautical miles along the eastern boundary of W-237A, extends approximately 3 nautical miles to shore along the mean lower low water line, and encompasses 1 mile (1.6 kilometers) of shoreline at Pacific Beach, Washington. Surf-zone activities would be conducted from an area on the shore and seaward. Table 44 provides a summary of bathymetric features within the offshore portion of the action area.

4.1.1 Special Use Airspace

The special use airspace in the offshore area is comprised of W-237, which extends westward off the coast of northern Washington State and is divided into nine sub-areas (A, C-H, and J). The eastern boundary of W-237 lies 3 nautical miles off the coast of Washington. The floor of W-237 extends to the ocean surface, and the ceiling of the airspace varies depending on the sub-area.

The Olympic Military Operations Area overlays both land (the Olympic Peninsula) and sea (extending to 3 nautical miles off the coast of Washington into the Pacific Ocean). The Military Operations Area lower limit is 6,000 feet above mean sea level but not below 1,200 feet above ground level due to the mountainous terrain, and the upper limit is up to but not including 18,000 ft., with a total area coverage of 1,619 square nautical miles (nm^2).

Above the Olympic Military Operations Area is the Olympic Air Traffic Control Assigned Airspace, which has a floor coinciding with the Olympic Military Operations Area ceiling. The Air Traffic Control Assigned Airspace has an upper limit of 35,000 feet. For this consultation, the Olympic Military Operations Area and the Olympic Air Traffic Control Assigned Airspace are components of the offshore area.

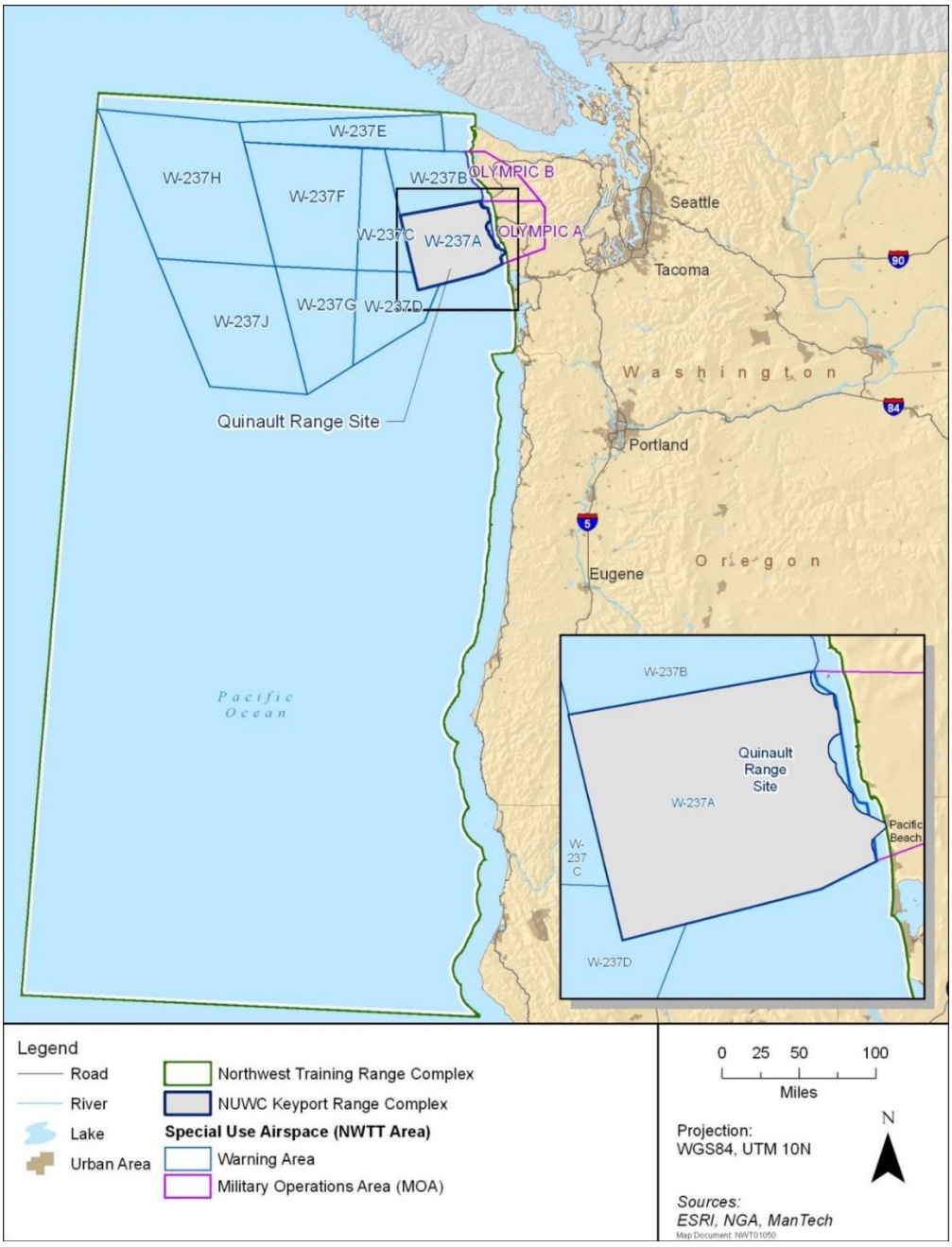


Figure 30. Offshore portion of the NWTT Study Area.

Table 44. Summary of bathymetric features within the offshore portion of the action area.

Range/Component	*Description*	*General Bathymetry[1,2]*
Offshore Area (California Current Large Marine Ecosystem)		
Pacific Northwest Ocean Surface/Subsurface OPAREA	Located from the Strait of Juan de Fuca to approximately 50 nm south of Eureka, California, and from the coast line of Washington, Oregon, and California westward to 130° west longitude.	Varying continental shelf width. Cascadia Abyssal Plain. Steep continental slope. Numerous seamounts, escarpments, canyons, and basins characterize the bathymetry of the OPAREA.
Quinault Range Site*	The Quinault Range Site includes a shore surf zone of 1 mile at Pacific Beach Washington.	The continental shelf is narrow and ranges in width from 8 to 40 mi. The Juan de Fuca and Quinault canyons reside within the shelf, and the continental slope has a steep upper portion and a gently sloping lower portion, grading into the Cascadia Basin.

[1] Navy (2010b)
[2] National Oceanic and Atmospheric Administration (2001). National Oceanic and Atmospheric Administration Nautical Charts were also reviewed to determine depth ranges at specific locations.
* Naval Sea Systems Command Undersea Warfare Center Keyport Range Complex
Notes: ° = degrees, feet = feet, mi. = miles, OPAREA = Operating Area

Source: U.S.Department of the Navy 2007, Naval Sea Systems Command Naval Undersea Warfare Center Keyport Range Complex Final Rule (76 FR 20257)

4.1.2 Sea and Undersea Space

The offshore area includes sea and undersea space approximately 510 nautical miles in length from the northern boundary at the mouth of the Strait of Juan de Fuca to the southern boundary at 40 degrees (°) north (N) latitude, and 250 nautical miles in length from the coastline to the western boundary at 130° west (W) longitude. The southern boundary of 40° N latitude corresponds to the northern boundary of Mendocino County in Northern California. Total surface area of the offshore area is approximately 121,000 nm^2. The offshore area extends to the shoreline only along the northern portion of the Washington coast, and is 12 nautical miles from the coastline from the southern boundary of the Olympic Military Operations Area to Northern California, including the entire coastline of Oregon and Northern California. The offshore area is 3 nautical miles offshore along the northern portion of the Washington coast with a small portion in the Quinault Range called the surf zone. The surf zone component extends north to south 5 nautical miles along the eastern boundary of W-237A, extends approximately 3 nautical miles to shore along the mean lower low water line, and encompasses 1 mile (1.6 kilometers) of shoreline at Pacific Beach, Washington.

Within the boundaries of the offshore area lies the Quinault Range Site (Figure 30), a defined area of sea space where training and testing is conducted. The Quinault Range Site coincides with the boundaries of the southern portion of W-237A and also includes a surf zone component. The surf zone component extends north to south for 5 nautical miles along the eastern boundary of W-237A, approximately 3 nautical miles to shore along the mean lower low water line, and encompasses 1 mi. of shoreline at Pacific Beach, Washington. Surf-zone activities would be conducted from an area on the shore and seaward.

4.2 Inland Waters of the Action Area

The Inland Waters component of the action area includes air, sea, and undersea space inland of the Pacific coastline, from buoy "J" at 48° 29.6' N, 125° W, eastward to include all waters of the Strait of Juan de Fuca and the Puget Sound. None of this area extends into Oregon, California, or Alaska waters or airspace. Within the Inland Waters are specific geographic components in which training and testing occur. The Inland Waters and its component areas are shown in Figure 31 and described in Table 45.

4.2.1 Air Space

Restricted Area 6701 (R-6701, Admiralty Bay) has a lower limit at the ocean surface and an upper limit of 5,000 feet. This airspace covers a total area of 22 nm². Chinook A and B Military Operations Areas are 56 nm² of airspace south and west of Admiralty Bay (Figure 31). The Chinook Military Operations Areas extend from 300 feet to 5,000 feet above the ocean surface. This airspace covers a total area of 56 nm^2.

4.2.2 Sea and Undersea Space

4.2.2.1 Explosive Ordnance Disposal Range

Two active Explosive Ordnance Disposal (EOD) ranges, also used for swimmer training in Mine Countermeasures, are located in the Inland Waters at the following locations, as depicted in Figure 31: Naval Base (NAVBASE) Kitsap Bangor – Hood Canal EOD Range; and Naval Air Station (NAS) Whidbey Island (NASWI) – Crescent Harbor EOD Range.

4.2.2.2 Surface and Subsurface Testing Sites

There are three geographically distinct range sites in the Inland Waters where the Navy conducts surface and subsurface testing and some limited training. The Keyport Range Site is located in Kitsap County and includes portions of Liberty Bay and Port Orchard Reach (also known as Port Orchard Narrows). The Dabob Bay Range Complex Site is located in Hood Canal, in Jefferson, Kitsap, and Mason counties. The Carr Inlet Operating Area (OPAREA) is located in southern Puget Sound.

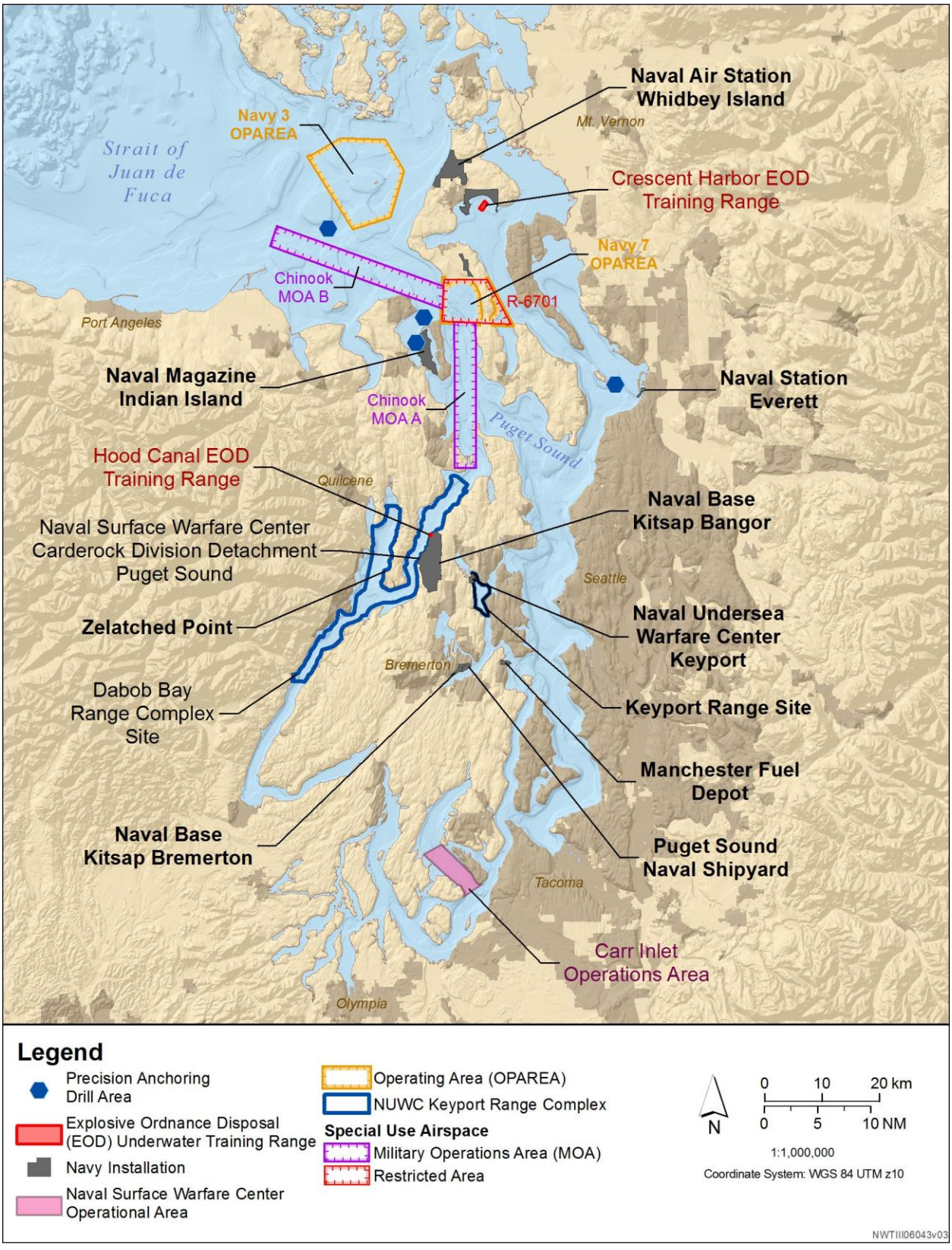


Figure 31. Inland waters of the action area.

Table 45. Summary of bathymetric features within the Inland Waters of the action area.

Range/Component	*Description*	*General Bathymetry[1,2]*
Inland Waters (Puget Sound)		
Keyport Range Site*	Located adjacent to the Naval Undersea Warfare Center Keyport.	Water depth at the Keyport Range Site is less than 100 feet This range provides approximately 3.2 nm^2 of shallow underwater testing area and a shallow lagoon.
Dabob Bay Range Complex Site*	This site is located in Dabob Bay and Hood Canal, as well as the connecting waters between the bay and the canal. The southern boundary extends to the Hamma Hamma River, and the northern boundary is 1 nm south of the Hood Canal Bridge (Highway 104).	Maximum depth in the Dabob Bay is 600 feet The deep water range in Dabob Bay is approximately 14.5 nm^2, and has hard walls with a mud bottom. The Hood Canal contains two deep-water operating areas with an average depth of 200 feet The portion of Hood Canal that connects with Dabob Bay has a water depth of typically greater than 300 feet The total area of the Dabob Bay Range Complex Site is approximately 45.7 nm^2.
Carr Inlet OPAREA	Located in southern Puget Sound, the Carr Inlet OPAREA is an arm of water between Key Peninsula and Gig Harbor Peninsula. The southern end is connected to the southern basin of Puget Sound. Northward, it separates McNeil Island and Fox Island, as well as the peninsulas of Key and Gig Harbors.	Maximum depth at Carr Inlet OPAREA is 545 feet The total area of the Carr Inlet OPAREA is approximately 12 nm^2 in size.

[1] Navy (2010b)

[2] National Oceanic and Atmospheric Administration (2001.) National Oceanic and Atmospheric Administration Nautical Charts were also reviewed to determine depth ranges at specific locations. Some "pierside activities" listed as taking place at these locations actually take place away from the coastal areas and are located inside ranges.

* Naval Sea Systems Command Undersea Warfare Center Keyport Range Complex

Notes: feet = feet, nm^2 = square nautical miles, OPAREA = Operating Area

Source: U.S.Department of the Navy 2007, Naval Sea Systems Command Naval Undersea Warfare Center Keyport Range Complex Final Rule (76 FR 20257)

The Keyport Range Site is located adjacent to NAVBASE Kitsap, providing approximately 3.2 square nautical miles for underwater testing, including in-shore shallow water sites and a shallow lagoon to support integrated undersea warfare systems and vehicle maintenance and engineering activities. Water depth at the Keyport Range Site is less than 100 feet (30 meters). Underwater

tracking of test activities can be accomplished by using temporary or portable range equipment. The Navy has conducted underwater testing at the Keyport Range Site since 1914.

The Dabob Bay Range Complex Site includes Dabob Bay and Hood Canal from one mile south of the Hood Canal Bridge to the Hamma Hamma River, a total area of approximately 45.7 square nautical miles. The Navy has conducted underwater testing at the Dabob Bay Range Complex Site since 1956, beginning with a control center at Whitney Point. The control center was subsequently moved to Zelatched Point.

Dabob Bay is a deep-water area in Jefferson County approximately 14.5 square nautical miles in size, which contains an underwater acoustic tracking range. The acoustic tracking space within the range is approximately 7.3 nautical miles by 1.3 nautical miles with a maximum depth of 600 feet. The Dabob Bay tracking range, the only component of the Dabob Bay Range Complex Site with extensive acoustic monitoring instrumentation installed on the seafloor, provides for object tracking, communications, passive sensing, and target simulation. Many activities conducted within Dabob Bay are supported by land-based facilities at Zelatched Point.

The Carr Inlet OPAREA is a quiet, deep-water inland range approximately 12 square nautical miles in size. It is located in an arm of water between Key Peninsula and Gig Harbor Peninsula. Its southern end is connected to the southern basin of Puget Sound. Northward, Carr Inlet OPAREA separates McNeil Island and Fox Island as well as the Key and Gig Harbor peninsulas. The acoustic tracking space within the range is approximately 6 nautical miles by 2 nautical miles with a maximum depth of 545 feet The Navy previously performed underwater acoustic testing at Carr Inlet from the 1950s through 2009, at which time the underwater acoustic testing activities were relocated to Naval Surface Warfare Center Carderock Division at Naval Base Kitsap Bangor. While no permanently installed structures are present in the Carr Inlet OPAREA, the waterway remains a Naval Restricted Area (33 Code of Federal Regulations [CFR] section 334.1250).

4.2.2.3 Pierside Testing Facilities

Most of the NWTT activities occur in established training and testing ranges; however, the Navy conducts some testing at or near Navy piers. Most pierside testing is sonar maintenance or other type of testing conducted while ships are in port for maintenance or system re-fitting. The Navy piers within the action area are all within Puget Sound and include NAVBASE Kitsap Bremerton in Sinclair Inlet, NAVBASE Kitsap Bangor Waterfront in Hood Canal, and NAVSTA Everett.

4.2.2.4 Navy Surface Operations Areas

In addition to the areas described above, there are two surface and subsurface operations areas used for Navy training and testing within the Inland Waters. Navy 3 OPAREA is a surface and subsurface area off the west coast of northern Whidbey Island. Navy 7 OPAREA is the surface and subsurface area that lies beneath R-6701. This area covers a total area of 56 nm^2.

4.3 Southeast Alaska Acoustic Measurement Facility

The Western Behm Canal is located in Southeast Alaska, near the city of Ketchikan, Alaska. Southeast Alaska Acoustic Measurement Facility (SEAFAC) is located in the Western Behm Canal and covers an area of 48 nm^2 (Figure 32). The Navy has been conducting testing activities at SEAFAC since 1992. The facility replaced the Santa Cruz Acoustic Range Facility in Southern California and is now the location for some acoustic testing previously conducted at the NSWC Carr Inlet Acoustic Range in Washington State.

SEAFAC is comprised of land-based facilities and in-water assets. The land-based facilities located within 5.5 acres on Back Island are not a part of the action area, and Navy activities occurring in this location will be or have been addressed under separate regulatory documentation.

The in-water assets include two sites: the underway site and the static site. Bottom-moored acoustic measurement arrays are located in the middle of the site. These instrumented arrays are established for measuring vessel signatures when a vessel is underway (underway site) and is at rest and moored (static site). Active acoustic sources are used for communications, range calibration, and to provide position information for units operating submerged on the range.

The SEAFAC at-sea areas are:

- Restricted Areas 1 through 5. The five restricted areas are located within Western Behm Canal. The main purposes of the restricted areas are to provide for vessel and public safety, lessen acoustic encroachment from non-participating vessels, and prohibit certain activities that could damage SEAFAC's sensitive in-water acoustic instruments and associated cables. Area 5 encompasses the entire SEAFAC operations area.
- Underway Measurement Site. The underway measurement site is in the center of Western Behm Canal and is 5,000 yards (4,572 meters) wide and 12,000 yards (10,973 meters) long. The acoustic arrays are located at the center of this area (Area 1).
- Static Site. The static site is approximately 2 nautical miles northwest of Back Island. During testing, a vessel is tethered between two surface barges. In most scenarios, the vessel submerges to conduct acoustic measurements. The static site is located at the center of Area 2.
- Area 3 and Area 4. These restricted areas provide protection to underwater cables and the bottom-mounted equipment they encompass.

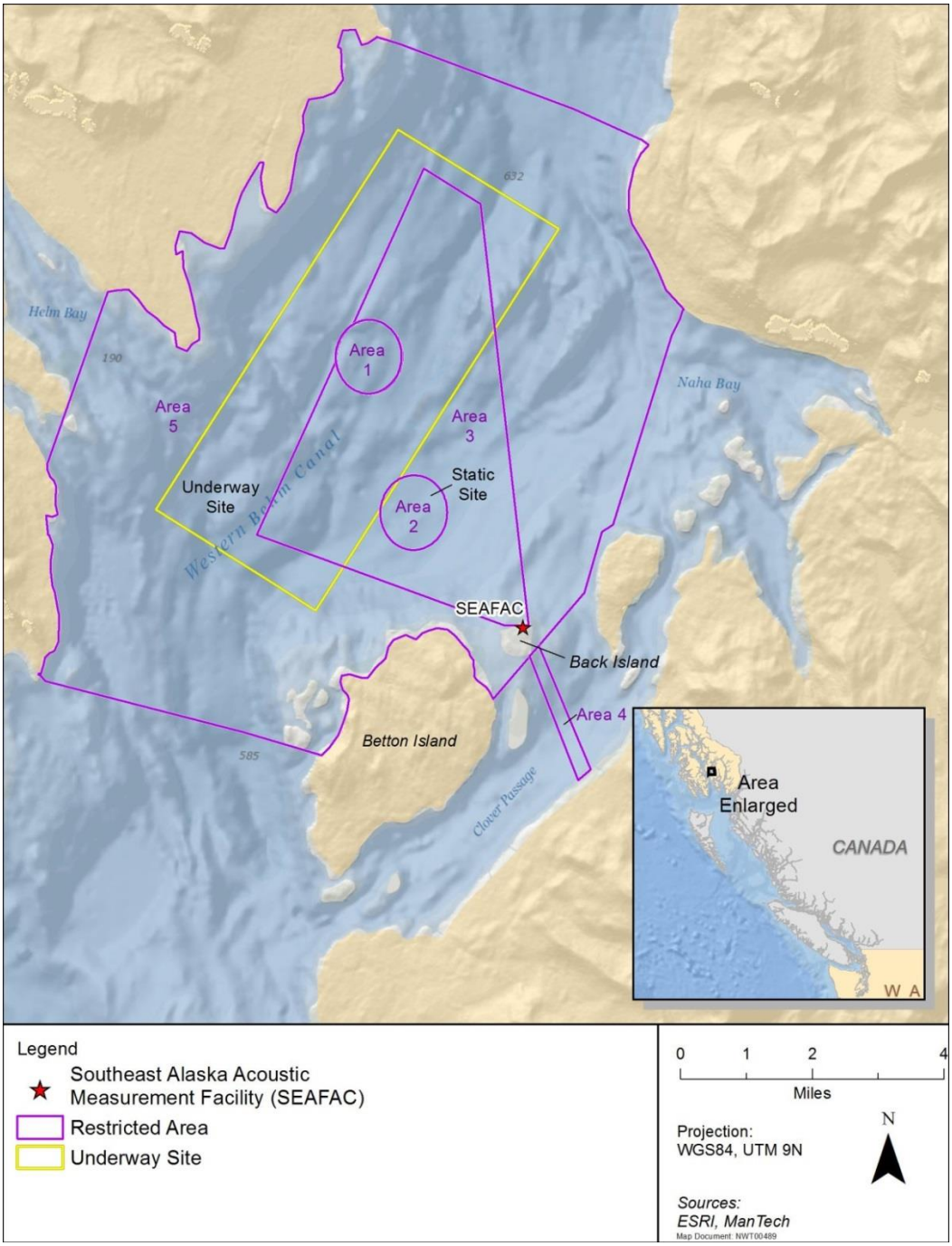


Figure 32. Western Behm Canal, Alaska and the Southeast Alaska Acoustic Measurement Facility.

5 POTENTIAL STRESSORS

The potential stressors we expect to result from the proposed action are acoustic stressors, explosive stressors, energy stressors, physical disturbance and strike stressors, entanglement stressors, and ingestion stressors. In addition to the direct effects of these stressors on ESA-listed species, we consider any indirect effects of stressors through impacts to species' habitat (including water quality or sediments) or prey. Further discussion of each of these stressors is below.

5.1 Acoustic Stressors

Acoustic stressors include acoustic signals emitted into the water for a specific purpose (e.g., by active sonars), as well as incidental sources of broadband sound produced as a byproduct of vessel movement; aircraft transits; and use of weapons or other deployed objects. Explosives also produce broadband sound but are characterized separately from other acoustic sources due to their unique energetic characteristics.

5.1.1 Sonar and other Transducers

Active sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. Some examples are mid-frequency, hull-mounted sonars used to find and track submarines; high-frequency small object detection sonars used to detect mines; high-frequency underwater modems used to transfer data over short ranges; and very high-frequency (greater than 200 kilohertz) Doppler sonars used for navigation, like those used on commercial and private vessels. The characteristics of these sonars and other transducers, such as source level, beam width, directivity, and frequency, depend on the purpose of the source. Higher frequencies can carry more information or provide more information about objects off which they reflect, but attenuate more rapidly. Lower frequencies attenuate less rapidly, so may detect objects over a longer distance, but with less detail.

Propagation of sound produced underwater is highly dependent on environmental characteristics such as bathymetry, bottom type, water depth, temperature, and salinity. The sound received at a particular location will be different than near the source due to the interaction of many factors including propagation loss; how the sound is reflected, refracted, or scattered; the potential for reverberation; and interference due to multi-path propagation. In addition, absorption greatly affects the distance over which higher-frequency sounds propagate. Because of the complexity of analyzing sound propagation in the ocean environment, the Navy relies on acoustic models in its exposure analysis that consider sound source characteristics and varying ocean conditions across the action area. The Navy's acoustic modeling approach is described further in Section 2.2 of this opinion and in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018d).

The Navy employs a variety of sonars and other transducers to obtain and transmit information about the undersea environment. Under the Navy's proposed action, training and testing

activities using sonar and other transducers could occur throughout the action area, although use would generally occur within 200 nautical miles of shore in Navy operating areas, on Navy range complexes, on Navy testing ranges, or around inshore locations (See Section 3 *Description of the Proposed Action* for more specifics on Navy sonar types and hours of use). Below we describe the use of sonar and other transducers for anti-submarine warfare, mine warfare and small object detection, navigation and safety, and communication as potential stressors on ESA-listed resources.

5.1.1.1 Anti-Submarine Warfare

Sonar used during anti-submarine warfare would impart the greatest amount of acoustic energy of any category of sonar and other transducers proposed for use by the Navy. Types of sonars used to detect enemy vessels include hull-mounted, towed, line array, sonobuoy, helicopter dipping, and torpedo sonars. In addition, acoustic targets and decoys (countermeasures) may be deployed to emulate the sound signatures of vessels or repeat received signals. Some anti-submarine warfare tracking exercises and ship unit level training activities would also be conducted using simulators in conjunction with other training exercises.

Most anti-submarine warfare sonars are mid-frequency (1–10 kilohertz) because mid-frequency sound balances sufficient resolution to identify targets with distance over which threats can be identified. However, some sources may use higher or lower frequencies. Duty cycles can vary widely, from rarely used to continuously active. For example, a submarine‘s mission revolves around its stealth; therefore, active sonar is used infrequently because its use would also reveal a submarine's location. Anti-submarine warfare sonars can be wide-angle in a search mode or highly directional in a track mode.

Most anti-submarine warfare activities involving submarines or submarine targets would occur in waters greater than 600 feet deep due to safety concerns about running aground at shallower depths. Sonars used for anti-submarine warfare activities would typically be used beyond 12 nautical miles from shore. Exceptions include use of dipping sonar by helicopters, maintenance of systems while in port, and system checks while transiting to or from port.

5.1.1.2 Mine Warfare, Small Object Detection, and Imaging

Sonars used to locate mines and other small objects, as well those used in imaging (e.g., for hull inspections or imaging of the seafloor), are typically high frequency or very high frequency. Higher frequencies allow for greater resolution and, due to their greater attenuation, are most effective over shorter distances. Mine detection sonar can be deployed (towed or vessel hull-mounted) at variable depths on moving platforms (ships, helicopters, or unmanned vehicles) to sweep a suspected mined area. Hull-mounted anti-submarine sonars can also be used in an object detection mode known as "Kingfisher" mode. Sonars used for imaging are usually used in close proximity to the area of interest, such as pointing downward near the seafloor.

Mine detection sonar use would be concentrated in areas where practice mines are deployed, typically in water depths less than 200 feet and at established training minefields or temporary minefields close to strategic ports and harbors. Kingfisher mode on vessels is most likely to be used when transiting to and from port. Sound sources used for imaging could be used throughout the action area.

5.1.1.3 Navigation and Safety

Similar to commercial and private vessels, Navy vessels employ navigational acoustic devices including speed logs, Doppler sonars for ship positioning, and fathometers. These may be in use at any time for safe vessel operation. These sources are typically highly directional to obtain specific navigational data.

5.1.1.4 Communication

Sound sources used to transmit data (such as underwater modems), provide location (pingers), or send a single brief release signal to bottom-mounted devices (acoustic release) may be used throughout the action area. These sources typically have low duty cycles and are usually only used when it is desirable to send a detectable acoustic message.

5.1.2 Explosives

The near-instantaneous rise from ambient to an extremely high peak pressure is what makes an explosive shock wave potentially damaging to ESA-listed resources. Farther from an explosive, the peak pressures decay and the explosive waves propagate as an impulsive, broadband sound. Several parameters influence the effect of an explosive: the weight of the explosive warhead, the type of explosive material, the boundaries and characteristics of the propagation medium, and, in water, the detonation depth. The NEW, the explosive power of a charge expressed as the equivalent weight of trinitrotoluene, accounts for the first two parameters.

Explosive detonations during training and testing activities associated with high-explosive munitions include, but are not limited to, bombs, missiles, rockets, naval gun shells, torpedoes, mines, demolition charges, and explosive sonobuoys. Explosive detonations associated with torpedoes and sonobuoys would occur in the water column; mines and demolition charges could be detonated in the water column or on the ocean bottom. Explosive detonations associated with bombs, missiles, and naval gun shells could occur in the air (see below) or near the water's surface. To avoid or reduce potential impacts from explosive training activities on marine species, the Navy would not conduct explosive training activities within 50 nautical miles from shore in the Marine Species Coastal Mitigation Area. A small number of explosive mine neutralization training activities would occur in existing mine warfare areas in Inland Waters (i.e., Crescent Harbor and Hood Canal Explosive Ordnance Disposal Ranges).

A new mine countermeasure and neutralization testing activity would occur closer to shore than other activities that involve the use of in-water explosives. This activity would occur greater than 3 nautical miles from shore in the Quinault Range Site (E4 would occur more than 3 nautical

miles from shore while E7 would occur more than 6 nautical miles from shore), or greater than 12 nautical miles from shore elsewhere in the offshore area. Explosives would not be used in the Olympic Coast National Marine Sanctuary. Offshore mine countermeasure and neutralization testing activities involving explosives would occur approximately two times per year and typically in water depths shallower than 1,000 feet For personnel safety, explosive mine countermeasure and neutralization testing activities will be conducted during daylight hours only in Beaufort sea state 3 or less. Moored mines (explosive bin E7) associated with this activity would be located in the water column at least 75 feet or greater from the seafloor and 100 feet or greater below the water surface (Navy 2020a). No explosive mines would be detonated on the ocean bottom. Bin E7 explosives would not be detonated within six nautical miles from land. Mine neutralizers (bin E4) would be used with inert targets either on the seafloor and in the water column, 20 feet or greater below the water surface. The E4 explosives used for this activity would have a directional force aimed at the target. The maximum annual number of explosives proposed by the Navy for this activity is 36 bin E4 and 5 bin E7 explosives. The Navy's proposed action also includes a cap on the seven year total number of offshore explosives for this activity as follows: 108 E4 and 15 E7 explosives.

To minimize the effects of explosive mine countermeasure and neutralization testing activities on ESA-listed resources (see Section 3.5.1.9 for details), the Navy has proposed the following conservation measures: 1) The Navy will, to the maximum extent practicable, conduct mine countermeasure and neutralization testing from July 1 through September 30 when operating within 20 nautical miles from shore *in the Marine Species Coastal Mitigation Area*, and 2) Within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area, the Navy will conduct a maximum of one mine countermeasure and neutralization testing event annually, not to exceed the use of 20 E4 and 3 E7 explosives, from October 1 through June 30. Additionally, within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area, the Navy will not exceed 60 E4 and 9 E7 explosives over seven years, from October 1 through June 30.

Propagation of explosive pressure waves in water is highly dependent on environmental characteristics such as bathymetry, bottom type, water depth, temperature, and salinity, which affect how the pressure waves are reflected, refracted, or scattered; the potential for reverberation; and interference due to multi-path propagation. In addition, absorption greatly affects the distance over which higher frequency components of explosive broadband noise can propagate.

Explosions in air include detonations of projectiles and missiles during air-to-air missile exercises conducted during air warfare. These explosions typically occur far above the water surface. Some typical types of explosive munitions that would be detonated in air during Navy activities are shown in Table 46. In air, the propagation of impulsive noise from an explosion is highly influenced by atmospheric conditions, including temperature and wind.

Table 46. Typical air[1] explosive munitions during Navy activities (Navy 2020e).

Weapon Type[2]	*Net Explosive Weight[3] (lb.)*	*Typical (Minimum) Altitude of Detonation (ft.)*
Surface-to-Air Missile		
RIM-116 Rolling Airframe Missile	39	< 3,000 (500)
RIM-7 Sea Sparrow	36	> 15,000 (500)
Air-to-Air Missile		
AIM-9 Sidewinder	38	> 15,000 (15,000)
AIM-7 Sparrow	36	> 15,000 (15,000)
AIM-120 AMRAAM	17	> 15,000 (15,000)
Air-to-Surface Missile		
AGM-84 Harpoon	488	< 100 (surface)
Projectile - Large Caliber[4]		
5-inch gunnery round	8.8	< 100 (surface)

[1] Includes explosives at altitude and explosives that may be used against a surface target at sea.
[2] Mission Design Series and popular name shown for missiles.
[3] Net Explosive Weight refers to the amount of explosives; the actual weight of a munition may be larger due to other components.
[4] Most medium and large caliber projectiles used during Navy training and testing activities do not contain high explosives.
Notes: AMRAAM = Advanced Medium-Range Air-to-Air Missile

5.1.3 Vessel Noise

Potential impacts of vessel noise on ESA-listed species include masking of other biologically relevant sounds, physiological stress, and changes in behavior. Sounds emitted by large vessels can be characterized as low-frequency, continuous, or tonal, and SPLs at a source will vary according to speed, burden, capacity and length (Kipple and Gabriele 2007; Mckenna et al. 2012; Richardson et al. 1995b). Because of the number of vessels involved in Navy testing and training activities, the vessel speed, and the use of course changes as a tactical measure with the associated sounds, the available evidence leads us to expect marine mammals and sea turtles to treat Navy vessels as stressors. Further, without considering differences in sound fields associated with any active sonar that is used during these activities, the available evidence suggests that unit- and intermediate-level exercises and testing activities would represent different stress regimes because of differences in the number of vessels involved, vessel maneuvers, and vessel speeds.

Naval vessels (including ships and small craft) produce low-frequency, broadband underwater sound, though the exact level of noise produced varies depending on the nature, size, and speed of the ship. Mckenna et al. (2012) determined that container ships produced broadband source

levels around 188 dB re 1 µPa rms and a typical fishing vessel radiates noise at a source level of about 158 dB re 1 µPa rms (Richardson et al. 1995b; Urick 1983a). The average acoustic signature for a Navy vessel is 163 dB re 1 µPa rms, while the average acoustic signature for a commercial vessel is 175 dB re 1 µPa rms (Mintz and Filadelfo 2011b). Typical large vessel ship-radiated noise is dominated by tonals related to blade and shaft sources at frequencies below about 50 Hz and by broadband components related to cavitation and flow noise at higher frequencies (approximately around the one-third octave band centered at 100 Hz) (Mintz and Filadelfo 2011b; Richardson et al. 1995a; Urick 1983c). Vessels ranging from 135 to 337 meters (Nimitz-class aircraft carriers, for example, have lengths of about 332 meters) generate peak source sound levels from 169 to 200 dB re 1 µPa rms between 8 Hz and 430 Hz. Sound produced by vessels will typically increase with speed. During training and testing, speeds of most large naval vessels (greater than 60 feet) generally range from 10 to 15 knots. Navy ships will, on occasion, operate at higher speeds within their specific operational capabilities.

As described in more detail in Section 5.3.1 below (*Vessel Strike*), Navy vessel traffic makes up an extremely small amount of overall vessel traffic (i.e., much less than one percent) in the inland portion of the action area. Navy vessels may represent an even smaller amount of overall vessel traffic noise in the action area because many Navy ships incorporate quieting technology that other vessels (e.g., commercial ships) do not (Mintz and Filadelfo 2011b). For example, surface combatant ships (e.g., guided missile destroyer, guided missile cruiser, and Littoral Combat Ship) and submarines are designed to be very quiet to evade enemy detection. The Navy implements a "Buy Quiet" policy for equipment aboard ships which requires designers and engineers to obtain noise emission data before purchasing to choose the quietest available. The Navy also researches and implements technology improvements that minimize noise. For example, propellers used on Navy ships have been subject to design improvements to reduce excitation. The quietest Navy warships radiate much less broadband noise than a typical fishing vessel, while the loudest Navy ships during travel are almost on par with large oil tankers (Mintz and Filadelfo 2011b).

While commercial traffic (and, therefore, broadband noise generated by it) is relatively steady throughout the year, Navy traffic is episodic in the ocean. Activities involving vessel movements occur intermittently and are variable in duration. Exposure of ESA-listed species to Navy vessel noise would be greatest in the areas of highest vessel traffic in close proximity to ports within Puget Sound including Naval Base Kitsap Bremerton in Sinclair Inlet, Naval Base Kitsap Bangor Waterfront in Hood Canal, and Naval Station Everett.

5.1.4 Aircraft Noise

Many Navy activities involve some level of activity from aircraft that include helicopters, maritime patrols, and fighter jets. Low-flying aircraft produce sounds that marine mammals and sea turtles can potentially hear when they occur at or near the ocean's surface. Helicopters generally tend to produce sounds that can be heard at or below the ocean's surface more than

fixed-wing aircraft of similar size and larger aircraft tend to be louder than smaller aircraft. Underwater sounds from aircraft are strongest just below the surface and directly under the aircraft. Sounds from aircraft would not have physical effects on marine animals but represent acoustic stimuli (primarily low-frequency sounds from engines and rotors) that have been reported to affect the behavior of some marine mammals and sea turtles. There are few studies of the responses of marine animals to air traffic and the few that are available have produced mixed results. Some investigators report responses while others report no responses.

Fixed-wing, tiltrotor, and rotary-wing aircraft are used for a variety of training and testing activities throughout the action area, contributing both airborne and underwater sound to the ocean environment. Aircraft used in training and testing generally have turboprop or jet engines. Motors, propellers, and rotors produce the most noise, with some noise contributed by aerodynamic turbulence. Aircraft sounds have more energy at lower frequencies. Aircraft may transit to or from vessels at sea throughout the action area from established airfields on land. Military activities involving aircraft generally are dispersed over large expanses of open ocean but can be highly concentrated in time and location. Table 47 provides source levels for some typical aircraft used during training and testing in the action area and depicts comparable airborne source levels for the F-35A, EA-18G, and F/A-18C/D aircraft during takeoff. Table 48 shows the number of ongoing events and the number of events proposed that include the use of aircraft.

Sound generated in air is transmitted to water primarily in a narrow area directly below the source. A sound wave propagating from any source must enter the water at an angle of incidence of about 13 degrees or less from the vertical for the wave to continue propagating under the water's surface. At greater angles of incidence, the water surface acts as an effective reflector of the sound wave and allows very little penetration of the wave below the water (Urick 1983a). Water depth and bottom conditions strongly influence how the sound from airborne sources propagates underwater. At lower altitudes, sound levels reaching the water surface would be higher, but the transmission area would be smaller (i.e., sound would radiate out as a cone from the aircraft, with the area of transmission at the water surface being larger at increasing distances). As the sound source gains altitude, sound reaching the water surface diminishes, but the possible transmission area increases.

Table 47. Representative aircraft sound characteristics (Navy 2020e).

Noise Source	*Sound Pressure Level*
In-Water Noise Level	
F/A-18 Subsonic at 1,000 feet (300 meters) Altitude	152 dB re 1 µPa at 2 m below water surface[1]
F/A-18 Subsonic at 10,000 feet (3,000 meters) Altitude	128 dB re 1 µPa at 2 m below water surface[1]
H-60 Helicopter Hovering at 82 feet (25 meters) Altitude	Approximately 125 dB re 1 µPa at 1 m below water surface*
Airborne Noise Level	
F/A-18C/D Under Military Power	143 dBA re 20 µPa at 13 m from source[3]
F/A-18C/D Under Afterburner	146 dBA re 20 µPa at 13 m from source[3]
F35-A Under Military Power	145 dBA re 20 µPa at 13 m from source[3]
F-35-A Under Afterburner	148 dBA re 20 µPa at 13 m from source[3]
H-60 Helicopter Hovering at 82 feet (25 meters) Altitude	113 dBA re 20 µPa at 25 m from source[2]

Sources: [1]Eller and Cavanagh (2000), [2]Bousman and Kufeld (2005), [3]U.S. Naval Research Advisory Committee (2009)

*estimate based on in-air level

Notes: dB re 1 µPa = decibel(s) referenced to 1 micropascal, dBA re 20 µPa = A-weighted decibel(s) referenced to 20 micropascals, m = meter(s), feet = feet

Table 48. Annual number and location of events including aircraft movement (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	Proposed Action	*Ongoing*	Proposed Action
Offshore Area	6,311	7,047	113	260
Inland Waters	100	143	456	61
Western Behm Canal	0	0	4	4
Total	**6,411**	**7,190**	**573**	**325**

5.1.4.1 Fixed-wing aircraft

Noise generated by fixed-wing aircraft is transient in nature and extremely variable in intensity. Most fixed-wing aircraft sorties (a flight mission made by an individual aircraft) would occur above 3,000 feet Air combat maneuver altitudes generally range from 5,000 to 30,000 feet, and typical airspeeds range from very low (less than 200 knots) to high subsonic (less than 600 knots). Sound exposure levels at the sea surface from most air combat maneuver overflights are

expected to be less than 85 A-weighted decibels (based on an F/A-18 aircraft flying at an altitude of 5,000 feet and at a subsonic airspeed (400 knots). Exposure to fixed-wing aircraft noise in water would be brief (seconds) as an aircraft quickly passes overhead.

5.1.4.2 Helicopters

In general, helicopters produce lower-frequency sounds and vibration at a higher intensity than fixed-wing aircraft. Helicopter sounds contain dominant tones from the rotors that are generally below 500 Hz. Helicopters often radiate more sound forward than backward. The underwater noise produced is generally brief when compared with the duration of audibility in the air and is estimated to be 125 dB re 1 µPa at 1 meter below water surface for a UH-60 hovering 82 feet (25 meter) altitude (Kufeld and M. 2005).

Helicopter unit level training typically entails single-aircraft sorties over water that start and end at an air station, although flights may occur from ships at sea. Individual flights typically last about two to four hours. Some events require low-altitude flights over a defined area, such as mine countermeasure activities deploying towed systems. Most helicopter sorties associated with mine countermeasures would occur at altitudes as low as 75-100 feet Likewise, in some anti-submarine warfare events, a dipping sonar is deployed from a line suspended from a helicopter hovering at low altitudes over the water.

5.1.4.3 Sonic Booms

An intense but infrequent type of aircraft noise is the sonic boom, produced when an aircraft exceeds the speed of sound. Supersonic aircraft flights are not intentionally generated below 30,000 feet unless over water and are generally conducted more than 30 nautical miles from inhabited coastal areas or islands. Deviation from these guidelines may occur for tactical missions that require supersonic flight, phases of formal training requiring supersonic speeds, research and test flights that require supersonic speeds, and for flight demonstration purposes when authorized by the Chief of Naval Operations.

Several factors that influence sonic booms include weight, size, and shape of aircraft or vehicle; altitude; flight paths; and atmospheric conditions. A larger and heavier aircraft must displace more air and create more lift to sustain flight, compared with small, light aircraft. Therefore, larger aircraft create sonic booms that are stronger than those of smaller, lighter aircraft. Consequently, the larger and heavier the aircraft, the stronger the shock waves (Navy 2017b). Aircraft maneuvers that result in changes to acceleration, flight path angle, or heading can also affect the strength of a boom. In general, an increase in flight path angle (lifting the aircraft's nose) will diffuse a boom while a decrease (lowering the aircraft's nose) will focus it. In addition, acceleration will focus a boom while deceleration will weaken it. Any change in horizontal direction will focus or intensify a boom by causing two or more wave fronts that originated from the aircraft at different times to coincide exactly (Navy 2017b). Atmospheric conditions such as wind speed and direction, and air temperature and pressure can also influence the sound propagation of a sonic boom.

Of all the factors influencing sonic booms, increasing altitude is the most effective method of reducing the sonic boom intensity that is experienced at the sea or shore level. The width of the boom "carpet" or area exposed to a sonic boom beneath an aircraft is about one mile for each 1,000 feet of altitude. For example, an aircraft flying supersonic, straight, and level at 50,000 feet can produce a sonic boom carpet about 50 mi. wide. The sonic boom, however, would not be uniform, and its intensity at the water surface would decrease with greater aircraft altitude. Maximum intensity is directly beneath the aircraft and decreases as the lateral distance from the flight path increases until shock waves refract away from the ground or water surface and the sonic boom attenuates. The lateral spreading of the sonic boom depends only on altitude, speed, and the atmosphere and is independent of the vehicle's shape, size, and weight. The ratio of the aircraft length to maximum cross-sectional area also influences the intensity of the sonic boom. The longer and more slender the aircraft, the weaker the shock waves. The wider and more blunt the aircraft, the stronger the shock waves can be (Navy 2017b).

In air, the energy from a sonic boom is concentrated in the frequency range from 0.1 to 100 Hz. The underwater sound field due to transmitted sonic boom waveforms is primarily composed of low-frequency components (Sparrow 2002b), and frequencies greater than 20 Hz have been found to be difficult to observe at depths greater than 33 feet (10 meters) (Sohn et al. 2000a). F/A-18 Hornet supersonic flight was modeled to obtain peak SPLs and energy flux density at the water surface and at depth (Laney and Cavanagh 2000). These results are shown in Table 49.

Table 49. Sonic boom underwater sound levels modeled for supersonic flight from a representative aircraft (Navy 2020e).

*Mach Number**	*Aircraft Altitude (km)*	*Peak SPL (dB re 1 µPa)*			*Energy Flux Density (dB re 1 µPa²-s)[1]*		
		At surface	*50 m Depth*	*100 m Depth*	*At surface*	*50 m Depth*	*100 m Depth*
1.2	1	176	138	126	160	131	122
	5	164	132	121	150	126	117
	10	158	130	119	144	124	115
2	1	178	146	134	161	137	128
	5	166	139	128	150	131	122
	10	159	135	124	144	127	119

* Mach number equals aircraft speed divided by the speed of sound.

Notes: SPL = sound pressure level, dB re 1 µPa = decibel(s) referenced to 1 micropascal, dB re 1 μPa^2-s = decibel(s) referenced to 1 micropascal squared seconds, m = meter(s)

[1] Equivalent to SEL for a plane wave.

5.1.5 Weapons Firing, Launch and Impact Noise

The Navy trains and tests using a variety of weapons. Depending on the weapon, noise may be produced at launch or firing, while in flight, or upon impact. Not all weapons utilize explosives, either by design or because they are non-explosive practice munitions. Noise produced by explosives, both in air and water, were discussed in Section 5.1.2.

Small- to medium-caliber rounds up to but not including the 57 millimeter (mm) non-explosive round could be used 12 nautical miles or more from shore. Large-caliber non-explosive rounds could be used 20 nautical miles or more from shore. Medium- and large-caliber explosive rounds could be used 50 nautical miles or more from shore. Examples of some types of weapons noise resulting from the proposed action are shown in Table 50.

Firing a gun produces a muzzle blast in air that propagates away from the gun with strongest directivity in the direction of fire. Because the muzzle blast is generated at the gun, the noise decays with distance from the gun. As the pressure from the muzzle blast from a ship-mounted large caliber gun propagates in air toward the water surface, the pressure can be both reflected from the water surface and transmitted into the water. Most sound enters the water in a narrow cone beneath the sound source (within about 13 to 14 degrees of vertical), with most sound outside of this cone being totally reflected from the water surface. In-water sound levels were measured during the muzzle blast of a 5-inch large caliber naval gun. The highest possible sound level in the water (average peak SPL of 200 dB re 1 µPa, measured 5 feet below the surface) was obtained when the gun was fired at the lowest angle, placing the blast closest to the water surface (Yagla and Stiegler 2003b). The unweighted SEL would be expected to be 15 to 20 dB lower than the peak pressure, making the highest possible SEL in the water about 180 to 185 dB re 1 μPa^2-s directly below the muzzle blast. Configuration of the 5-inch gun on Navy ships also affects how sound from much muzzle blast could enter the water. On cruisers, when swung out to either side the barrel of the gun extends beyond the ship deck and over water. On destroyers, of which there are more of in the Navy's fleet, when swung out to either side the barrel of the gun is still over the ship's deck. Other gunfire arrangements, such as with smaller-caliber weapons or greater angles of fire, would result in less sound entering the water. The sound entering the water would have the strongest directivity directly downward beneath the gun blast, with lower sound pressures at increasing angles of incidence until the angle of incidence is reached where no sound enters the water.

Table 50. Examples of some types of weapons noise from NWTT activities (Navy 2020e).

Noise Source	*Sound Level*
In-Water Noise Level	
Naval Gunfire Muzzle Blast (5-inch)	Approximately 200 dB re 1 µPa peak directly under gun muzzle at 1.5 m below the water surface[1]
Airborne Noise Level	
Naval Gunfire Muzzle Blast (5-inch)	178 dB re 20 µPa peak directly below the gun muzzle above the water surface[1]
Hellfire Missile Launch from Aircraft	149 dB re 20 µPa at 4.5 m[2]
Advanced Gun System Missile (115-millimeter)	133–143 dBA re 20 µPa between 12 and 22 m from the launcher on shore[3]
RIM 116 Surface-to-Air Missile	122–135 dBA re 20 µPa between 2 and 4 m from the launcher on shore[3]
Tactical Tomahawk Cruise Missile	92 dBA re 20 µPa 529 m from the launcher on shore[3]

Notes: dB re 1 µPa = decibel(s) referenced to 1 micropascal, dB re 20 µPa = decibel(s) referenced to 20 micropascals, dBA re 20 µPa = A-weighted decibel(s) referenced to 20 micropascals, m = meter(s)
Sources: [1]Yagla and Stiegler (2003a); [2]U.S. Department of the Army (1999); [3]U.S. Department of the Navy (2013b)

Large-caliber gunfire also sends energy through the ship structure and into the water. This effect was investigated in conjunction with the measurement of five inches gun firing described above. The energy transmitted through the ship to the water for a typical round was about six percent of that from the muzzle blast impinging on the water (U.S. Department of the Navy 2000). Therefore, sound transmitted from the gun through the hull into the water is a minimal component of overall weapons firing noise.

Supersonic projectiles, such as a fired gun shell or kinetic energy weapon, create a bow shock wave along the line of fire. A bow shock wave is an impulsive sound caused by a projectile exceeding the speed of sound. The bow shock wave itself travels at the speed of sound in air. The projectile bow shock wave created in air by a shell in flight at supersonic speeds propagates in a cone (generally about 65 degrees) behind the projectile in the direction of fire (Pater 1981). Like sound from the gun muzzle blast, sound waves from a projectile in flight could only enter the water in a narrow cone beneath the sound source, with in-air sound being totally reflected from the water surface outside of the cone. The region of underwater sound influence from a single traveling shell would be relatively narrow, and the duration of sound influence would be brief at any location. Measurements of a five inch projectile shock wave ranged from 140 to 147 dB re 20 µPa SPL peak taken at the ground surface at 0.59 nautical miles distance from the firing location and 10 degrees off the line of fire for safety (approximately 190 meters from the shell's trajectory) (U.S. Department of the Navy 1981). Hyperkinetic projectiles may travel up to and

exceed approximately six times the speed of sound in air, or about 6,500 feet per second (U.S. Department of the Navy 2014b)(U.S. Department of the Navy 2014b)(U.S. Department of the Navy, 2014b)(U.S. Department of the Navy 2014b)(U.S. Department of the Navy 2014b)(U.S. Department of the Navy 2014b). Sound due to missile and target launches is typically at a maximum at initiation of the booster rocket. It rapidly fades as the missile or target reaches optimal thrust conditions and the missile or target reaches a downrange distance where the booster burns out and the sustainer engine continues. Missiles can be rocket or jet propelled. Examples of launch noise sound levels are shown in Table 50.

Any object dropped in the water would create a noise upon impact, depending on the object's size, mass, and speed. Sounds of this type are produced by the kinetic energy transfer of the object with the target surface and are highly localized to the area of disturbance. A significant portion of an object's kinetic energy would be lost to splash, any deformation of the object, and other forms of non-mechanical energy (Mclennan 1997). The remaining energy could contribute to sound generation. Most objects would be only momentarily detectable, but some large objects traveling at high speeds could generate a broadband impulsive sound upon impact with the water surface. Sound associated with impact events is typically of low frequency (less than 250 Hz) and of short duration.

5.2 Energy Stressors

Energy stressors include in-water electromagnetic devices, in-air electromagnetic devices, and lasers, each of which is described further in the sections below.

5.2.1 Electromagnetic Devices

In-water electromagnetic energy devices include towed or unmanned mine warfare systems that simply mimic the electromagnetic signature of a vessel passing through the water. None of the devices include any type of electromagnetic "pulse." A mine neutralization device could be towed through the water by a surface vessel or remotely operated vehicle, emitting an electromagnetic field and mechanically generated underwater sound to simulate the presence of a ship. The sound and electromagnetic signature cause nearby mines to detonate.

Generally, voltage used to power these systems is around 30 volts. Since saltwater is an excellent conductor, just 35 volts (capped at 55 volts) is required to generate the current needed to power the systems. The static magnetic field generated by the mine neutralization devices is of relatively minute strength. Typically, the maximum magnetic field generated would be approximately 2,300 microteslas[14]. This level of electromagnetic density is very low compared to magnetic fields generated by other everyday items (e.g., the magnetic field generated is between the levels of a refrigerator magnet, which is 15,000 to 20,000 microteslas).

[14] The microtesla is a unit of measurement of magnetic flux density, or "magnetic induction."

The only training exercise included as part of the proposed action with in-water electromagnetic devices would occur once every two years in Inland Waters. In years of occurrence, this activity has four separate events in which the in-water electromagnetic devices would be used. There are no testing activities proposed that involve the use of in-water electromagnetic devices.

In addition to in-water electromagnetic devices, the proposed action includes activities that involve the use of in-air electromagnetic devices. Sources of electromagnetic energy in the air include communications transmitters, kinetic energy weapons, radars, and electronic countermeasure transmitters. However, in-air electromagnetic devices are not considered stressors to the ESA-listed species analyzed in this opinion (i.e., no effect), and are therefore not discussed further.

5.2.2 Lasers

Low-energy lasers are used to illuminate or designate targets, to measure the distance to a target, to guide weapons, to aid in communication, and to detect or classify mines. High-energy laser weapons are a newly proposed stressor. High-energy laser weapons testing involves the use of directed energy as a weapon against small surface vessels and airborne targets. They would be employed from surface ships and are designed to create small but critical failures in potential targets. High-energy laser weapons are expected to be used at short ranges. If there is a miss from a boat target, the laser beam may strike the water in the 200 meters (219 yards) to 6.5 kilometers (3.5 nautical miles) range or more, assuming an engagement range of 200 meters (219 yards) to 5 kilometers (2.7 nautical miles). At these ranges, the low angles to the water will reflect most of the laser energy. The laser will lose a significant amount of energy within only a few centimeters from the surface. The penetration will raise the water temperature based on the beam's incident angle to the surface of the water near the beam. Only the water in the immediate vicinity of the laser beam just under the surface would be affected. The hot water would quickly mix with the cooler surrounding water. As a result, striking the ocean with a high-energy laser beam should not be a hazard to underwater marine life, except at or very near the laser beam just below the ocean surface. The Navy has proposed 54 annual testing events (all in offshore areas) that include high-energy laser weapons as part of NWTT Phase III activities. High-energy laser weapons would not be used during training activities.

There are safeguards on high-energy laser platforms that reduce the probability of the laser striking the water. These safeguards include the following: 1) The high energy laser platform has provisions that prevent misfiring (i.e., firing when not intended) that all but eliminate the possibility of that event; 2) The high-energy laser platforms have built-in constraints that only permit firing when it is locked onto a target. It also automatically interrupts firing if the target track on a target is lost; and 3) Operators are trained to stop firing when the laser aim point moves off of the selected target.

At-sea testing may include use of laser-based optical communication systems. As the laser penetrates the water, at least 1.7 percent of a laser beam is scattered and reflected at the surface.

Once it is underwater, the light will lose power at an exponential rate due to scattering and absorption. A laser used entirely in the water only loses power from absorption and scattering as it travels because it does not have to pass through the boundary between air and water. An in-water laser is still considered to have an extremely low potential to impact marine species due to its relatively low intensity at large distances and the highly aversive effect at close range for animals with vision (Zorn et al. 2000). Based on the parameters of the lasers used in Navy optical communication systems and the behavior and life history of major biological groups, the greatest potential for impact would likely be to the eye of a marine species.

5.3 Physical Disturbance and Strike Stressors

Physical disturbance and strike stressors described in the sections below include vessel strike, in-water devices, military expended materials, sea-floor devices, cavitation from vessels, precision anchoring, and personnel disturbance.

5.3.1 Vessel Strike

Vessels used by the Navy during training and testing activities include ships (e.g., aircraft carriers, surface combatants), support craft, and submarines ranging in size from 15 feet to over 1,000 feet Large Navy ships generally operate at speeds in the range of 10 to 15 knots, and submarines generally operate at speeds in the range of 8 to 13 knots. Small craft (for purposes of this discussion, less than 40 feet [12 m] in length), which are all support craft, have much more variable speeds (dependent on the mission). While these speeds are representative of most events, some vessels need to operate outside of these parameters. There are a few specific events including high speed tests of newly constructed vessels such as aircraft carriers, amphibious assault ships and the joint high speed vessel (which will operate at an average speed of 35 knots) where vessels would operate at higher speeds. Table 51 provides examples of the types of vessels, length, and speeds typically used in Navy testing and training activities.

A new activity for NWTT Phase III that could increase the risk of vessel strike is propulsion testing. During this activity the ship is tested for maneuverability, including full power and endurance runs. Ships will operate in the offshore portion of the NWTT Study Area at least 10 nautical miles from shore. Ships will operate at a full spectrum of capable speeds. The high speed portion of the test typically lasts for four hours, which involves going full speed ahead, coming to a stop and going full speed astern. Endurance runs are conducted for a total of two hours. Typically, only one vessel would be used per event. Between eight to ten events could occur per year, with each event typically lasting up to five days.

Table 51. Representative vessel types, lengths, and speeds (Navy 2020e).

Type	*Example(s)*	*Length*	*Typical Operating Speed*
Aircraft Carrier	Aircraft Carrier (CVN)	>1,000 ft.	10–15 knots
Surface Combatant	Cruisers (CG), Destroyers (DDG), Littoral Combat Ships (LCS)	300–700 ft.	10–15 knots
Amphibious Warfare Ship	Amphibious Assault Ship (LHA, LHD), Amphibious Transport Dock (LPD), Dock Landing Ship (LSD)	300–900 ft.	10–15 knots
Combat Logistics Force Ships	Fast Combat Support Ship (T-AOE), Dry Cargo/Ammunition Ship (T-AKE), Fleet Replenishment Oilers (T-AO)	600–750 ft.	8–12 knots
Support Craft/Other	Amphibious Assault Vehicle (AAV); Combat Rubber Raiding Craft (CRRC); Landing Craft, Mechanized (LCM); Landing Craft, Utility (LCU); Submarine Tenders (AS); Yard Patrol Craft (YP)	15–140 ft.	0–20 knots
Support Craft/Other – Specialized High Speed	High Speed Ferry/Catamaran; Patrol Combatants (PC); Rigid Hull Inflatable Boat (RHIB); Expeditionary Fast Transport (EPF); Landing Craft, Air Cushion (LCAC)	33–320 ft.	0–50+ knots
Submarines	Fleet Ballistic Missile Submarines (SSBN), Attack Submarines (SSN), Guided Missile Submarines (SSGN)	300–600 ft.	8–13 knots

Notes: > indicates greater than

There are a total of 44 Navy vessels (6 destroyers, 14 submarines, 2 aircraft carriers, and 22 security vessels) homeported in the action area that at one time or another will be engaged in training and testing activities. Given deployments and maintenance schedules, this total number of vessels is never all present in the action area at the same time. The number of Navy vessels actively in use at any given time varies and is dependent on local training or testing requirements. Vessel movement would be widely dispersed throughout the action area, but more concentrated in portions of the action area near ports, naval installations, and range complexes. Navy traffic in the action area is heaviest offshore of the naval ports at Everett and Bremerton.

The Center for Naval Analysis conducts studies to determine traffic patterns of Navy and non-Navy vessels. The most recent analysis covered the period 2011–2015, and included U.S. Navy surface ship traffic and non-military vessels such as cargo vessels, bulk carriers, commercial fishing vessels, oil tankers, passenger vessels, tugs, and research vessels (Mintz 2012; Mintz 2016). Navy vessel traffic makes up approximately one-tenth of one percent of overall vessel traffic in the inland portion of the action area. In contrast to the approximately 171,000 commercial vessel transits in the inland portion of the NWTT action area, Navy vessels are projected to undertake approximately 240 transits per year for training and testing activities (Navy 2020e). While commercial traffic (and, therefore, broadband noise generated by it) is relatively steady throughout the year, Navy traffic is episodic in the ocean. Vessels engaged in

training and testing may consist of a single vessel involved in unit-level activity for a few hours or multiple vessels involved in a major training exercise that could last a few weeks within a given area. Activities involving vessel movements occur intermittently and are variable in duration. Activities range from involving one or two vessels to several vessels operating over various time frames and locations. Vessel movements in the action area fall into one of two categories; (1) those activities that occur in the offshore component of the action area and (2) those activities that occur in inshore waters.

Activities that occur in the offshore component of the action area may last from a few hours to a few weeks. Vessels associated with those activities would be widely dispersed in the offshore waters, but more concentrated in portions of the action area in close proximity to ports, naval installations, range complexes, and testing ranges. In contrast, activities that occur in inshore waters can last from a few hours to up to 12 hours of daily movement per vessel per activity. The vessels operating within the inshore waters are generally smaller than those in the offshore waters.

Table 52 shows the number and location of proposed events that include the use of vessels. The manner in which the Navy uses vessels to accomplish its training and testing activities is likely to remain consistent with the range of variability observed over the last decade. In the 10-year period from 2009 to 2018, there was an average of approximately 84.9 vessel days per year in the action area, as compared to an estimated 82.6 vessel days per year for the proposed action.

Table 52. Annual number and location of events including vessel movement (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	***Proposed Action***	***Ongoing***	***Proposed Action***
Offshore Area	1,156	1,144	181	283
Inland Waters	368	327	916	918
Western Behm Canal	0	0	60	63
Total	**1,524**	**1,471**	**1,157**	**1,264**

5.3.2 Military Expended Materials

Military expended materials that may cause physical disturbance or strike include: 1) all sizes of non-explosive practice munitions (Table 53), 2) expended materials other than munitions (Table 54), 3) fragments from high-explosive munitions (Table 55), and 4) expendable targets (Table 56).

Table 53. Annual maximum number and location of expended non-explosive practice munitions (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Bombs				
Offshore Area	110	84	0	0
Missiles				
Offshore Area	15	4	0	0
Sabot – Kinetic Energy Rounds				
Offshore Area	0	0	0	80
Large-Caliber Projectiles				
Offshore Area	2,800	2,800	0	160
Medium-Caliber Projectiles				
Offshore Area	42,164	26,410	0	0
Small-Caliber Projectiles				
Offshore Area	121,200	121,000	0	0
Small-Caliber Projectile Casings				
Inland Waters	3,036	3,036	0	0
Sonobuoys (includes Buoys, Bathythermograph Buoys, and Signal Underwater Sound buoys)				
Offshore Area	8,928	9,338	1,000	4,233
Inland Waters	0	0	6	0
Marine Markers				
Offshore Area	334	230	190	0
Inland Waters	0	40	0	0
Anti-Torpedo Torpedo				
Offshore Area	0	0	123	58
Inland Waters	0	0	81	176

Table 54. Annual maximum number and location of other expended or recovered items (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Acoustic Countermeasures (Recovered)				
Western Behm Canal	0	0	20	5
Acoustic Countermeasures (Expended)				
Offshore Area	0	0	663	751
Inland Waters	0	0	1,837	720
Western Behm Canal	0	0	4	1
Anchors (Expended)				
Inland Waters	0	0	884	720
Anchors (Recovered)				
Offshore Area	0	0	180	445
Inland Waters	0	40	2,462	2,527
Western Behm Canal	0	0	20	20
Canisters – Miscellaneous (Expended)				
Offshore Area	170	170	0	0
Western Behm Canal	0	0	0	4
Heavyweight Torpedoes (Recovered)				
Offshore Area	0	2	220	148
Inland Waters	0	0	189	230
Lightweight Torpedoes (Recovered)				
Offshore Area	0	16	41	78
Inland Waters	0	0	62	48
Illumination Flares (Expended)				
Offshore Area	24	4	0	0

Table 55. Annual maximum number and location of explosive munitions that may result in fragments (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Torpedoes				
Offshore Area	0	0	6	8
Neutralizers				
Offshore Area	0	0	0	36
Explosive Mines				
Offshore Area	0	0	0	5
Sonobuoys and Buoys				
Offshore Area	0	0	142	80
Bombs				
Offshore Area	10	2	0	0
Missiles				
Offshore Area	27	14	0	0
Large-Caliber Projectiles				
Offshore Area	390	112	0	80
Medium-Caliber Projectiles (includes Grenades)				
Offshore Area	6,368	250	0	0
Explosive Ordnance Disposal Underwater Detonations				
Inland Waters	42	42	0	0

Table 56. Annual maximum number and location of expended and recovered[1] targets (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Sub-surface Targets (Mobile)				
Offshore Area	393	469	23	185
Inland Waters	0	0	768	1,127
Sub-surface Targets (Stationary)				
Offshore Area	0	0	7	3,335
Inland Waters	0	0	5,422	7,317
Surface Targets (Mobile)				
Offshore Area	0	0	0	162
Surface Targets (Stationary)				
Offshore Area	372	374	22	253
Inland Waters	0	0	407	542
Air Targets				
Offshore Area	188	133	0	162
Mine Shapes (Non-Explosive) - Recovered				
Offshore Area	112	0	36	181
Inland Waters	42	112	12,982	3,776
Western Behm Canal	0	0	20	20
Mine Shapes (Non-Explosive) - Expended				
Offshore Area	0	0	0	280
Inland Waters	0	0	0	336

[1] Unless specified as "expended," the Navy makes best effort to recover all targets.
[2] In some cases the 2015 numbers have been adjusted to conform to current definitions of targets.

5.3.3 In-Water Devices

In-water devices include unmanned vehicles, such as remotely operated vehicles, unmanned surface vehicles and unmanned undersea vehicles and towed devices. These devices are self-propelled and unmanned or towed through the water from a variety of platforms, including helicopters, unmanned underwater vehicles, and surface ships (Table 57). In-water devices are generally smaller than most Navy vessels, ranging from several inches to about 50 feet, and can operate anywhere from the water surface to the benthic zone. Table 58 shows the number and location of proposed events that include the use of in-water devices.

Table 57. Representative types, sizes, and speeds of Navy in-water devices (Navy 2020e).

Type	*Example(s)*	*Length*	*Typical Operating Speed*
Towed Device	Minehunting Sonar Systems; Improved Surface Tow Target; Towed Sonar System; MK-103, MK-104 and MK-105 Minesweeping Systems; Organic Airborne and Surface Influence Sweep	< 33 ft.	10–40 knots
Unmanned Surface Vehicle	MK-33 Seaborne Power Target Drone Boat, QST-35A Seaborne Powered Target, Ship Deployable Seaborne Target, Small Waterplane Area Twin Hull, Unmanned Influence Sweep System	< 50 ft.	Variable, up to 50+ knots
Large Unmanned Surface Vehicle	Research and Development Surface Vessels, Patrol Boats	< 200 ft.	Typical 1–15 knots, sprint 25–50 knots
Unmanned Underwater Vehicle	Acoustic Mine Targeting System, Airborne Mine Neutralization System, AN/AQS Systems, Archerfish Common Neutralizer, Crawlers, CURV 21, Deep Drone 8000, Deep Submergence Rescue Vehicle, Gliders, Expendable Mobile Anti-Submarine Warfare Training Targets, Magnum Remotely Operated Vehicle, Manned Portables, MK 30 Anti-Submarine Warfare Targets, Remote Multi-Mission Vehicle, Remote Minehunting System, Large Displacement Unmanned Underwater Vehicle	< 60 ft.	1–15 knots
Torpedoes	Light-weight and Heavy-weight Torpedoes	< 33 ft.	20–30 knots

Table 58. Annual number and location of events including in-water devices (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Offshore Area	495	541	156	215
Inland Waters	1 (Note 1)	59	576	664
Western Behm Canal	0	0	8	19
Total	**496**	**600**	**740**	**898**

Note 1: This ongoing event occurs once every two years.

5.3.4 Seafloor Devices

Seafloor devices represent items used during training or testing activities that are deployed onto the seafloor and recovered. These items include moored mine shapes, recoverable anchors, bottom-placed instruments, temporary bottom cable arrays, energy harvesting devices, and robotic vehicles referred to as "crawlers." Bottom-placed instruments usually include an anchor which may be expended while recovering the instrument. Seafloor devices are either stationary

or move very slowly along the bottom. Table 59 shows the number of ongoing events and the number of events proposed by the Navy that include the use of seafloor devices.

Table 59. Annual number and location of events including seafloor devices (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Anchors				
Offshore Area	0	0	91	70
Inland Waters	10	40	433	512
Western Behm Canal	0	0	1	1
Bottom-Placed Instruments				
Inland Waters	0	0	74	75
Mine Shapes				
Offshore Area	0	0	62	54
Inland Waters	13	13	446	454
Western Behm Canal	0	0	5	2
All Seafloor Devices (Anchors, Bottom-Placed Instruments, and Mine Shapes combined)[1]				
Offshore Area	0	0	111	92
Inland Waters	23	53	581	616
Western Behm Canal	0	0	5	2

[1]Because some activities include the use of more than one type of seafloor device, the number of events including anchors, bottom-placed instruments, or mine shapes may be less than the sum of each of those categories.

5.4 Entanglement Stressors

The Navy proposes to utilize a variety of materials that could pose an entanglement risk to ESA-listed species including fiber optic cables, guidance wires, sonobuoys wires, decelerators and parachutes, and biodegradable polymers. Depending on the type of material, entanglement could occur at the sea surface, in the water column, or on the seafloor. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with certain types of military expended materials could result in negative sub-lethal effects and mortality. For one of these materials to result in entanglement it must be long enough to wrap around the appendages of marine animals. Another critical factor is rigidity; the item must be flexible enough to wrap around appendages or bodies.

5.4.1 Wires and Cables

During some proposed training and testing activities, the Navy may temporarily install and remove or expend different types of wires and cables. Temporary installations could include arrays or mooring lines attached to the seafloor or to surface buoys or vessels.

Table 60 shows the number of events using fiber optic cables, guidance wires, and sonobuoy wires analyzed in the 2015 NWTT biological opinion (ongoing) and the number included in the proposed action.

Table 60. Annual number and location of expended wires and cables (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Fiber Optic Cables				
Offshore Area	0	0	20	36
Inland Waters	0	0	122	197
Guidance Wires				
Offshore Area	0	2	92	152
Inland Waters	0	0	155	230
Sonobuoy Wires (includes Bathythermograph Buoys)				
Offshore Area	8,928	9,338	1,000	4,049
Inland Waters	0	0	6	48

5.4.1.1 Fiber Optic Cables

Although a portion may be recovered, some fiber optic cables used during Navy training and testing activities would be expended. The length of the expended tactical fiber would vary depending on the activity. Tactical fiber has a silica core and acylate coating, and looks and feels like thin monofilament fishing line. Tensile strength and cable diameter may vary depending on the type of tactical fiber used, however, tactical fibers are generally 242 µm (0.24 mm) in diameter, have a 12 pounds tensile strength, and a 3.4 mm bend radius (Corning Incorporated, 2005 and Raytheon Company, 2015 as cited in Navy 2017b). Tactical fiber is relatively brittle; it readily breaks if knotted, kinked, or abraded against a sharp object. Deployed tactical fiber will break if looped beyond its bend radius (3.4 mm), or exceeds its tensile strength (12 lb.). Tactical fibers are often designed with controlled buoyancy to minimize the fiber's effect on vehicle movement. The tactical fiber would be suspended within the water column during the activity, and then be expended and sink to the seafloor (effective sink rate of 1.45 centimeters per second (Navy 2017b)) where it would be susceptible to abrasion and burial by sedimentation.

5.4.1.2 Guidance Wires

Guidance wires are used during heavy-weight torpedo firings to help the firing platform control and steer the torpedo. They trail behind the torpedo as it moves through the water. The guidance wire is then released from both the firing platform and the torpedo, and sinks to the ocean floor. The torpedo guidance wire is a single-strand, thin gauge, coated copper alloy. The tensile breaking strength of the wire is a maximum of 40.4 pounds (Swope and McDonald 2013), contrasting with the rope or lines associated with commercial fishing towed gear (trawls), stationary gear (traps), or entanglement gear (gillnets) that use ropes with substantially higher (up to 500 to 2,000 lb.) breaking strength as their "weak links." However, the guidance wire has a somewhat higher breaking strength than the monofilament used in the body of most commercial gillnets (typically 31 pounds or less). The resistance to looping and coiling suggest that torpedo guidance wire does not have a high entanglement potential compared to other entanglement hazards (Swope and McDonald 2013). Torpedo guidance wire sinks at a rate of 0.24 meters per second (Swope and McDonald 2013).

5.4.1.3 Sonobuoy and Bathythermograph Wires

Sonobuoys consist of a surface antenna and float unit and a subsurface hydrophone assembly unit. The two units are attached through a thin-gauge, dual-conductor, and hard-draw copper strand wire, which is then wrapped by hollow rubber tubing or a bungee in a spiral configuration. The tensile breaking strength of the wire and rubber tubing is no more than 40 lbs. The length of the wire is housed in a plastic canister dispenser, which remains attached upon deployment. The length of wire that extends out is no more than 1,500 feet and is dependent on the water depth and type of sonobuoy. Attached to the wire is a kite-drogue and damper disk stabilizing system made of non-woven nylon fabric. The nylon fabric is very thin and can be broken by hand. The wire runs through the stabilizing system and leads to the hydrophone components. The hydrophone components may be covered by thin plastic netting depending on the type of sonobuoy, but pose no entanglement risk. Each sonobuoy has a saltwater-activated polyurethane float that inflates when the sonobuoy is submerged and keeps the sonobuoy components floating vertically in the water column below it. Sonobuoys remain suspended in the water column for no more than 30 hours, after which they sink to the seafloor.

Bathythermographs are similar to sonobuoys in that they consist of an antenna, a float unit, and a subsurface unit (to measure temperature of the water column in the case of the bathythermograph) that is connected to the float unit by a wire. The bathythermograph wire is similar to the sonobuoy wire described above.

5.4.2 Decelerators and Parachutes

Decelerators and parachutes used during training and testing activities are classified into four different categories based on size: small, medium, large, and extra-large (Table 61). Aircraft-launched sonobuoys and lightweight torpedoes use nylon decelerators/parachutes ranging in size from 18 to 48 inches in diameter (small). The majority of the decelerators/parachutes in the small

size category are smaller (18 inches) cruciform shape decelerators/parachutes associated with sonobuoys. Illumination flares use medium-sized decelerators/parachutes, up to approximately 19 feet in diameter. Both small- and medium-sized decelerators/parachutes are made of cloth and nylon, many with weights on their short attachment lines to speed their sinking. At water impact, the decelerator/parachute assembly is expended and sinks away from the unit. The decelerator/parachute assembly may remain at the surface for 5 to 15 seconds before the decelerator/parachute and its housing sink to the seafloor, where it becomes flattened (Group 2005). Once settled on the bottom, the canopy may temporarily billow if bottom currents are present.

Table 61. Size categories for decelerators/parachutes expended during training and testing activities (Navy 2020e).

Size Category	*Diameter (ft.)*	*Associated Activity*
Small	1.5–6	Air-launched sonobuoys, lightweight torpedoes, and drones (drag decelerator/parachute)
Medium	19	Illumination flares
Large	30–50	Drones (main decelerator/parachute)

Aerial targets (drones) use large (between 30 and 50 feet in diameter) and extra-large (80 feet in diameter) decelerators/parachutes. Large and extra-large decelerators/parachutes are also made of cloth and nylon, with suspension lines of varying lengths (large: 40 to 70 feet in length [with up to 28 lines per decelerator/parachute]; and extra-large: 82 feet in length [with up to 64 lines per decelerator/parachute]). Some aerial targets also use a small drag parachute (6 feet in diameter) to slow their forward momentum prior to deploying the larger primary decelerator/parachute. Unlike the small- and medium-sized decelerators/parachutes, drone decelerators/parachutes do not have weights attached and may remain at the surface or suspended in the water column for some time prior to eventual settlement on the seafloor. Table 62 show the number and location of decelerator/parachutes expended during proposed training and testing activities.

Table 62. Annual number and location of expended decelerators/parachutes (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Small Decelerators/Parachutes				
Offshore Area	8,928	9,354	1,068	1,711
Inland Waters	0	0	113	176
Medium Decelerators/Parachutes				

Offshore Area	24	4	0	0
Large Parachutes				
Offshore Area	145	98	0	0

5.4.3 Biodegradable Polymer

The proposed use of biodegradable polymers introduces a new stressor that was not analyzed in the 2015 NWTT biological opinion. The biodegradable polymers that the Navy proposes to uses are designed to temporarily interact with the propeller(s) of a target craft, rendering it ineffective. Some of the polymer constituents would dissolve within two hours of immersion. The Navy has proposed to use biodegradable polymers during four events annually, all in Inland Waters (Dabob Range and Keyport). One round of biodegradable material is used for each test run. Each round of materials consists of about six pounds of biodegradable fibers and polymers. Typically, 15-20 test runs will be performed during each testing day totaling 120 lbs. The total water surface area impacted by the biodegradable materials is approximately ten square feet.

The majority of the expended material is collected and removed from the water after each test run. Based on the constituents of the biodegradable polymer the Navy proposes to use, it is anticipated that any material remaining in the water column will break down into small pieces within a few days to weeks. These pieces will break down further and dissolve into the water column within weeks to a few months. Degradation and dispersal timelines are influenced by water temperature, currents, and other oceanographic features. The longer the polymer remains in the water, the weaker it becomes, making it more brittle and likely to break. At the end of dispersion, the remaining materials are generally separated fibers with lengths on the order of 54 micrometers.

5.5 Ingestion Stressors

Some of the expended materials resulting from NWTT activities are small enough to be ingested by marine mammals, sea turtles, and fish. These include: non-explosive practice munitions (small- and medium-caliber), fragments from high-explosives, fragments from targets, chaff, flare casings (including plastic end caps and pistons), and some decelerators and parachutes. Solid metal materials, such as small-caliber projectiles or fragments from high-explosive munitions, sink rapidly to the seafloor. Lighter plastic items may be caught in currents and gyres or entangled in floating kelp and could remain in the water column for hours to weeks or indefinitely before sinking (e.g., plastic end caps [from chaff cartridges] or plastic pistons [from flare cartridges]).

5.5.1 Non-Explosive Practice Munitions

Only small- or medium-caliber projectiles and flechettes (small metal darts) from some non-explosive rockets would be small enough for marine animals to ingest, depending on the animal. Small- and medium-caliber projectiles include all sizes up to and including those that are 2.25 inches in diameter. Flechettes from some non-explosive rockets are approximately 2 inches in

length. Each non-explosive flechette rocket contains approximately 1,180 individual flechettes that are released. These solid metal materials would quickly move through the water column and settle to the seafloor.

5.5.2 Fragments from High Explosive Munitions

Many different types of high-explosive munitions can result in fragments that are expended at sea during training and testing activities. Types of high-explosive munitions that can result in fragments include torpedoes, neutralizers, grenades, projectiles, missiles, rockets, buoys, sonobuoys, countermeasures, mines, and bombs. Fragments would result from fractures in the munitions casing and would vary in size depending on the NEW and munition type. These solid metal materials would quickly sink through the water column and settle to the seafloor.

5.5.3 Target Related Materials

At-sea targets are usually remotely-operated airborne, surface, or subsurface traveling units, many of which are designed to be recovered for reuse. However, if they are used during activities that use high-explosives then they may result in fragments and ultimate loss of the target. Expendable targets that may result in fragments would include air-launched decoys, surface targets (e.g., marine markers, cardboard boxes, and 10 feet diameter red balloons), and mine shapes. Most target fragments would sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time.

5.5.4 Chaff

Chaff consists of reflective, aluminum-coated glass fibers used to obscure ships and aircraft from radar-guided systems. Chaff, which is stored in canisters, is either dispensed from aircraft or fired into the air from the decks of surface ships when an attack is imminent. The glass fibers create a radar cloud that mask the position of the ship or aircraft. Chaff is composed of an aluminum alloy coating on glass fibers of silicon dioxide (Navy 2017b). Chaff is released or dispensed from cartridges that contain millions of fibers. When deployed, a diffuse cloud of fibers is formed that is undetectable to the human eye. Chaff is a very light material, similar to fine human hair. It can remain suspended in air anywhere from 10 minutes to 10 hours and can travel considerable distances from its release point, depending on prevailing atmospheric conditions (Arfsten et al. 2002; Navy 2017b). Doppler radar has tracked chaff plumes containing approximately 900 grams of chaff drifting 200 miles from the point of release, with the plume covering more than 400 miles (Arfsten et al. 2002).

The chaff concentrations that marine animals could be exposed to following the discharge of multiple cartridges (e.g., following a single day of training) is difficult to accurately estimate because it depends on several variable factors. First, specific release points are not recorded and tend to be random, and chaff dispersion in air depends on prevailing atmospheric conditions. After falling from the air, chaff fibers would be expected to float on the sea surface for some

period, depending on wave and wind action. The fibers would be dispersed farther by sea currents as they float and slowly sink toward the bottom.

Table 63 shows the number and location of chaff cartridges used during training and testing activities, most to be used in special use airspace, not near shore.

5.5.5 Flares

Flares are pyrotechnic devices used to defend against heat-seeking missiles, where the missile seeks out the heat signature from the flare rather than the aircraft's engines. Similar to chaff, flares are also dispensed from aircraft. The flare device consists of a cylindrical cartridge approximately 1.4 inches in diameter and 5.8 inches in length. Flares are designed to burn completely. The only material that would enter the water would be a small, round, plastic compression pad or piston (0.45 to 4.1 grams depending on flare type). The flare pads and pistons float in sea water. Table 63 shows the number and location of flares used during training and testing activities, most to be used in special use airspace, not near shore.

Table 63. Annual number and location of expended chaff and flares and associated materials (Navy 2020e).

Activity Area	Training		Testing	
	Ongoing	*Proposed Action*	*Ongoing*	*Proposed Action*
Chaff				
Offshore Area	5,000	5,000	0	0
Flares				
Offshore Area	500	700	600	0
Compression Pad or Plastic Piston				
Offshore Area	500	700	600	0
Endcap – Chaff and Flare				
Offshore Area	5,500	5,700	600	0
Flare O-Ring				
Offshore Area	504	704	600	0

5.6 Secondary Stressors

The proposed action may result in secondary stressors that affect ESA-listed marine mammals, sea turtles, and fish indirectly through impacts to species habitat (including water quality or sediments) or prey. Potential secondary stressors include (1) explosives and byproducts, (2) metals, (3) chemicals from flares and propellants, and (4) chemical and biological simulants.

The use of Navy explosives could impact other species in the marine food web, including prey species that the ESA-listed species considered in this opinion feed upon. Underwater explosions may reduce available prey items for ESA-listed species by either directly killing prey or through

behavioral responses such as diving, scattering or avoidance of an area. Explosions can also leave explosive byproducts in the water which could have impacts on water quality.

Metals are introduced into seawater and sediments as a result of training and testing activities involving ship hulks, targets, munitions, and other military expended materials (Navy 2019c). Some metals bioaccumulate and physiological impacts begin to occur only after several trophic transfers concentrate the toxic metals. Several Navy training and testing activities introduce chemicals into the marine environment that are potentially harmful in higher concentrations. Chemicals introduced are principally from flares and propellants for missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment.

Chemical and biological simulants could also potentially affect ESA-listed species through secondary impacts to their habitat and prey. The Navy has proposed up to 50 simulant testing events as part of NWTT Phase III. This is a new activity for the NWTT action area and would be conducted in the offshore area at least 3 nautical miles from shore. Testing involves the deployment of compounds (i.e., simulants) as substitutes for chemical and biological warfare agents. Examples of chemical simulants that would be used include glacial acetic acid and triethyl phosphate. Biological simulants used by the Navy include spore-forming bacteria, non-spore-forming bacteria, ovalbumin, bacteriophage MS2, and *Aspergillus niger*. The simulants are generally dispersed by hand at the detector or by aircraft as a fine mist or aerosol.

6 SPECIES AND DESIGNATED CRITICAL HABITAT THAT MAY BE AFFECTED

This section identifies the ESA-listed species and designated critical habitat that potentially occur within the action area that may be affected by the proposed action along with their regulatory status (Table 64). Section 6.1 identifies those species that may be affected but are not likely to be adversely affected by the proposed action because the effects of the proposed action, evaluated by each stressor, were deemed insignificant, discountable, or fully beneficial. In this section we also discuss designated (or proposed) critical habitats within which proposed NWTT activities would not occur, nor would the activities be expected to have any impacts on the critical habitat or the essential physical and biological features[15] (PBFs) of this habitat (indicated with an * in Table 64). The species that may be affected but are not likely to be adversely affected, and the critical habitat that would not be affected by the proposed action are not carried forward in our effects analysis.

In Section 6.2, we provide a summary of the biology, ecology, and population status of those species that are likely to be adversely affected by one or more stressors created by the proposed action and detail information on their life histories in the action area, if known. We also describe designated (or proposed) critical habitat that may be affected by one or more stressors created by the proposed action, and discuss the condition and current function of such habitats, including the PBFs that contribute to the conservation value of the critical habitat. The species that are likely to be adversely affected and the critical habitat that may be affected by the proposed action are carried forward in our effects analysis (Section 8).

Table 64. ESA-listed species and designated (or proposed) critical habitat that may be affected by the proposed action.

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Blue Whale (*Balaenoptera musculus)*	E – 35 FR 18319	-- --	07/1998 10/2018
Fin Whale (*Balaenoptera physalus)*	E – 35 FR 18319	-- --	75 FR 47538 07/2010
Gray Whale (*Eschrichtius robustus*) – Western North Pacific DPS	E – 35 FR 18319	-- --	-- --
Humpback Whale (*Megaptera novaeangliae*) – Central America DPS	E – 81 FR 62259	84 FR 54354 (proposed)	11/1991

[15] Critical habitat designations use the terms "physical and biological feature" or "primary constituent elements" depending on when the designation was made. In this consultation, PBF represents both phrases.

Species	ESA Status	Critical Habitat	Recovery Plan
Humpback Whale (*Megaptera novaeangliae*) – Mexico DPS	T – 81 FR 62259	84 FR 54354 (proposed)	11/1991
Killer Whale (*Orcinus orca*) – Southern Resident killer whales DPS	E – 70 FR 69903 Amendment 80 FR 7380	71 FR 69054 84 FR 49214 (proposed)	73 FR 4176 01/2008
North Pacific Right Whale (*Eubalaena japonica*)	E – 73 FR 12024	73 FR 19000*	78 FR 34347 06/2013
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	-- --	12/2011
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	-- --	75 FR 81584 12/2010
Marine Mammals – Pinnipeds			
Guadalupe Fur Seal (*Arctocephalus townsendi*)	T – 50 FR 51252	-- --	-- --
Steller Sea Lion (*Eumetopias jubatus*) – Western DPS	E – 55 FR 49204	58 FR 45269*	73 FR 11872 2008
Marine Reptiles			
Leatherback Turtle (*Dermochelys coriacea*)	E – 35 FR 8491	44 FR 17710 and 77 FR 4170	10/1991 – U.S. Caribbean, Atlantic, and Gulf of Mexico 63 FR 28359 05/1998 – U.S. Pacific
Fishes			
Bocaccio (*Sebastes paucispinis*) – Puget Sound/Georgia Basin DPS	E – 75 FR 22276 and 82 FR 7711	79 FR 68041	81 FR 54556 (Draft) 10/2017
Chinook salmon (*Oncorhynchus tshawytscha)* – California Coastal ESU	T – 70 FR 37160	70 FR 52488*	81 FR 70666
Chinook salmon (*Oncorhynchus tshawytscha)* – Central Valley Spring-Run ESU	T – 70 FR 37160	70 FR 52488*	79 FR 42504
Chinook salmon (*Oncorhynchus tshawytscha)* – Lower Columbia River ESU	T – 70 FR 37160	70 FR 52629*	78 FR 41911

Species	ESA Status	Critical Habitat	Recovery Plan
Chinook salmon (*Oncorhynchus tshawytscha)* – Puget Sound ESU	T – 70 FR 37160	70 FR 52629	72 FR 2493
Chinook salmon (*Oncorhynchus tshawytscha)* – Sacramento River Winter-Run ESU	E – 70 FR 37160	58 FR 33212*	79 FR 42504
Chinook salmon (*Oncorhynchus tshawytscha)* – Snake River Fall-Run ESU	T – 70 FR 37160	58 FR 68543*	80 FR 67386 (Draft)
Chinook salmon (*Oncorhynchus tshawytscha)* – Snake River Spring/Summer Run ESU	T – 70 FR 37160	64 FR 57399*	81 FR 74770 (Draft) 11-2017-Final
Chinook salmon (*Oncorhynchus tshawytscha)* – Upper Columbia River Spring-Run ESU	E – 70 FR 37160	70 FR 52629*	72 FR 57303
Chinook salmon (*Oncorhynchus tshawytscha)* – Upper Willamette River ESU	T – 70 FR 37160	70 FR 52629*	76 FR 52317
Chum salmon (*Oncorhynchus keta*) – Columbia River ESU	T – 70 FR 37160	70 FR 52629*	78 FR 41911
Chum salmon (*Oncorhynchus keta*) – Hood Canal Summer-Run ESU	T – 70 FR 37160	70 FR 52629	72 FR 29121
Coho salmon (*Oncorhynchus kisutch*) – Central California Coast ESU	E – 70 FR 37160	64 FR 24049*	77 FR 54565
Coho salmon (*Oncorhynchus kisutch*) – Lower Columbia River ESU	T – 70 FR 37160	81 FR 9251*	78 FR 41911
Coho salmon (*Oncorhynchus kisutch*) – Oregon Coast ESU	T – 73 FR 7816	73 FR 7816*	81 FR 90780
Coho salmon (*Oncorhynchus kisutch*) – Southern Oregon and Northern California Coasts ESU	T – 70 FR 37160	64 FR 24049*	79 FR 58750
Eulachon (*Thaleichthys pacificus*) –Southern DPS	T – 75 FR 13012	76 FR 65323*	9/2017
Green Sturgeon (*Acipenser medirostris*) – Southern DPS	T – 71 FR 17757	74 FR 52300	2010 (Outline) 8/2018- Final
Sockeye salmon (*Oncorhynchus nerka*) – Ozette Lake ESU	T – 70 FR 37160	70 FR 52630*	74 FR 25706
Sockeye salmon (*Oncorhynchus nerka*) – Snake River ESU	E – 70 FR 37160	58 FR 68543*	80 FR 32365

Species	ESA Status	Critical Habitat	Recovery Plan
Steelhead (*Oncorhynchus mykiss*) – California Central Valley DPS	T – 71 FR 834	70 FR 52487*	79 FR 42504
Steelhead (*Oncorhynchus mykiss*) – Central California Coast DPS	T – 71 FR 834	70 FR 52487*	81 FR 70666
Steelhead (*Oncorhynchus mykiss*) – Lower Columbia River DPS	T – 71 FR 834	70 FR 52629*	78 FR 41911
Steelhead (*Oncorhynchus mykiss*) – Middle Columbia River DPS	T – 71 FR 834	70 FR 52629*	74 FR 50165
Steelhead (*Oncorhynchus mykiss*) – Northern California DPS	T – 71 FR 834	70 FR 52487*	81 FR 70666
Steelhead (*Oncorhynchus mykiss*) – Puget Sound DPS	T – 72 FR 26722	81 FR 9251*	84 FR 71379
Steelhead (*Oncorhynchus mykiss*) – Snake River Basin DPS	T – 71 FR 834	70 FR 52629*	81 FR 74770 (Draft) 11-2017-Final
Steelhead (*Oncorhynchus mykiss*) – South-Central California Coast DPS	T – 71 FR 834	70 FR 52487*	78 FR 77430
Steelhead (*Oncorhynchus mykiss*) – Southern California DPS	E – 71 FR 834	70 FR 52487*	77 FR 1669
Steelhead (*Oncorhynchus mykiss*) – Upper Columbia River DPS	T – 71 FR 834	70 FR 52629*	72 FR 57303
Steelhead (*Oncorhynchus mykiss*) – Upper Willamette River DPS	T – 71 FR 834	70 FR 52629*	76 FR 52317
Yelloweye Rockfish (*Sebastes rubberimus*) – Puget Sound/Georgia Basin DPS	T – 75 FR 22276 and 82 FR 7711	79 FR 68041	81 FR 54556 (Draft) 10/2017

* indicates that critical habitat for this species does not overlap with the action area and the proposed action would likely have no effect on this critical habitat.

6.1 Species Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed species that are not likely to be adversely affected by the proposed action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species. If we conclude that an ESA-listed species is not likely to be exposed to the proposed activities, we must also conclude that the species is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. An ESA-listed species or designated critical habitat that is exposed to a potential stressor but is likely to be unaffected by the exposure is also not likely to be adversely affected by the proposed action.

An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat. Beneficial effects are usually discussed when the project has a clear link to the ESA-listed species or its specific habitat needs, and consultation is required because the species may be affected.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect. That means the ESA-listed species may be expected to be affected, but the intensity of the impacts would not reach a scale where take would occur (e.g., harm, harassment).

Discountable effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did impact a listed species), but it is extremely unlikely to occur.[16]

We applied these criteria to the ESA-listed species in Table 64 above. We summarize our results below for ESA-listed species and critical habitat that are not likely to be adversely affected by any stressor created by the proposed action.

6.1.1 North Pacific Right Whale

The North Pacific right whale remains one of the most endangered whale species in the world. Their abundance likely numbers fewer than 1,000 individuals. Several lines of evidence indicate a total population size of less than 100. Based on photo-identification from 1998 to 2013 (Wade et al. 2011a) estimated 31 individuals, with a minimum population estimate of 25.7 individuals. Genetic data have identified 23 individuals based on samples collected between 1997 and 2011 (Leduc et al. 2012). There is currently no information on the population trend of North Pacific right whales.

As a result of past commercial whaling, the remnant population of North Pacific right whales has been left vulnerable to genetic drift and inbreeding due to low genetic variability. This low diversity potentially affects individuals by depressing fitness, lowering resistance to disease and parasites, and diminishing the whales' ability to adapt to environmental changes. At the

[16] When the terms "discountable" or "discountable effects" appear in this opinion, they refer to potential effects that are found to support a "not likely to adversely affect" conclusion because they are extremely unlikely to occur. The use of these terms should not be interpreted as having any meaning inconsistent with our regulatory definition of "effects of the action."

population level, low genetic diversity can lead to slower growth rates, lower resilience, and poorer long-term fitness (Lacy 1997). Marine mammals with an effective population size of a few dozen individuals likely can resist most of the deleterious consequences of inbreeding (Lande 1991). It has also been suggested that if the number of reproductive animals is fewer than fifty, the potential for impacts associated with inbreeding increases substantially. Rosenbaum et al. (2000) found that historic genetic diversity of North Pacific right whales was relatively high compared to North Atlantic right whales (*E. glacialis*), but samples from extant individuals showed very low genetic diversity, with only two matrilineal haplotypes among the five samples in their dataset.

The North Pacific right whale inhabits the Pacific Ocean, particularly between 20 and 60 degrees latitude. Prior to exploitation by commercial whalers, concentrations of right whales in the North Pacific where found in the Gulf of Alaska, Aleutian Islands, south central Bering Sea, Sea of Okhotsk, and Sea of Japan. There has been little recent sighting data of right whales occurring in the central North Pacific and Bering Sea. However, since 1996, North Pacific right whales have been consistently observed in Bristol Bay and the southeastern Bering Sea during summer months. Presently, sightings are extremely rare, occurring primarily in the Okhotsk Sea and the eastern Bering Sea (Brownell Jr. et al. 2001; Shelden et al. 2005; Wade et al. 2006; Zerbini et al. 2010). There are far fewer sightings of North Pacific right whales in the Gulf of Alaska than the Bering Sea (Brownell Jr. et al. 2001; Wade et al. 2011a; Zerbini et al. 2010). In addition to sighting data (Matsuoka et al. 2013; Wade et al. 2011a; Wade et al. 2011b), passive acoustic data have indicated the presence of North Pacific right whales in the Gulf of Alaska (Mellinger et al. 2004b; Sirovic et al. 2015). No right whales were detected from more than 5,324 hours of passive acoustic data obtained from Navy-funded monitoring devices in the north-central Gulf of Alaska (Baumann-Pickering et al. 2012; Debich et al. 2013), but calls were detected in 2013 during two days (21 June and 3 August) from a device located at Quinn Seamount (Sirovic et al. 2015).

Occurrence in the Offshore Portion of the NWTT Action Area

Various sightings of North Pacific right whales in the general vicinity of the action area have occurred on an irregular basis. Two right whales were sighted in 1983 on Swiftsure Bank at the entrance to the Strait of Juan de Fuca (Osborne et al. 1988). In May 1992 there was a sighting of a single of North Pacific right whale over Quinault Canyon (Green et al. 1992; Rowlett et al. 1994). There were no sightings of North Pacific right whales during six ship surveys conducted in summer and fall off California, Oregon, and Washington from 1991 through 2008 (Barlow 2010). Recently, two sightings and an acoustic detection of a North Pacific right whale have occurred in the vicinity of or in the action area. On June 20, 2013, a newspaper reported that a single right whale was sighted in waters off Haida Gwaii, British Colombia (located approximately 200 nautical miles (370 km) north of the action area; Hume (2013)). Right whales had not been detected by two bottom deployed passive acoustic monitoring devices in the offshore waters of Washington State from 2011 through 2012, but on June 29, 2013 (9 days after

the sighting to the north), a Navy-funded passive acoustic monitoring device at Quinault Canyon (in the action area) detected two right whale calls within a two-hour period (Sirovic et al. 2015). Approximately 4 months later (October 2013) another (different) right whale was sighted in a group of humpback whales off the entrance to the Strait of Juan de Fuca and moving south (DoN 2015; Pynn 2013). In 2017, a lone right whale was briefly observed close to shore off La Jolla Cove in Southern California (Price 2017). Because of the low population numbers (likely less than 1,000) in the North Pacific and the few individuals that have been observed (Brownell Jr. et al. 2001; Wade et al. 2006), even given these recent sightings/detections, this species is considered extremely rare in the action area. Based on this information, there is a very low probability of encountering this species anywhere in the coastal and offshore waters in the action area and their co-occurrence with potential stressors caused by the proposed action is extremely unlikely. Therefore the potential for any stressor to cause an effect to this species is also extremely unlikely (i.e., discountable).

Occurrence in Inland Waters of the NWTT Action Area

As noted above, the rarity of coastal records suggests right whales would not be present in more inland areas. The occurrence of a North Pacific right whale within the Inland Waters is considered extralimital. Therefore the potential for any stressor to cause an effect to this species is extremely unlikely (i.e., discountable).

Occurrence in the Western Behm Canal, Alaska

North Pacific right whales were not observed during the Alaska Fisheries Science Center's National Marine Mammal Laboratory 1991 through 2007 surveys of the Inland Waters of southeast Alaska (Dahlheim et al. 2009). Given their small population size (likely less than 1,000) and lack of sightings in southeast Alaska as noted above, North Pacific right whales are considered extralimital within the Behm Canal portion of the NWTT action area.

Since it is very unlikely that North Pacific right whales would be encountered in all portions of the NWTT action area, the potential for any stressor to cause an effect to this species is extremely unlikely (i.e., discountable).

Critical Habitat

In April 2008 (73 FR 19000), NMFS clarified that two areas previously designated as critical habitat for right whales in the North Pacific (71 FR 38277) also applied to the listed North Pacific right whale. The areas encompass about 36,750 square miles of marine habitat, which include feeding areas within the Gulf of Alaska and the Bering Sea that support the species.

The Navy's training and testing activities would not occur in the designated critical habitat nor would the activities be expected to have any impacts to the critical habitat or the essential physical and biological features (formerly referred to as primary constituent elements).

Conclusion

The extremely low population numbers of North Pacific right whale in the North Pacific Ocean over the past five decades and the rarity of reports from these waters suggests that North Pacific right whales are not reasonably likely to be exposed to the proposed action, its component activities, and the associated stressors considered in this consultation. This is because, in all portions of the NWTT action area, it is very unlikely that North Pacific right whales would be encountered and the potential for any stressor to cause an effect to this species is extremely unlikely. Therefore, we determine that the North Pacific right whale is not likely to be adversely affected by the proposed action.

Based on our determination that the proposed action is not likely to adversely affect this species, we further conclude that the proposed action is also not likely to jeopardize the continued existence of the North Pacific right whale. As a result of this finding, North Pacific right whale will not be discussed in the remainder of this opinion.

Due to the lack of designated critical habitat for North Pacific right whale in the action area, the proposed action would have no effect on the designated critical habitat for North Pacific right whales, hence we conclude that the proposed action would not destroy or adversely modify North Pacific right whale designated critical habitat. As a result of this finding, North Pacific right whale designated critical habitat will not be discussed further in this opinion.

6.1.2 Western North Pacific Gray Whale

Gray whales are mysticetes, or baleen whales. Gray whales are the only species in the family Eschrichtiidae. These large whales can grow to about 50 feet (15 meters) long, and weigh approximately 80,000 pounds (35,000 kg). Females are slightly larger than males. They have a mottled gray body, with small eyes located just above the corners of the mouth. Their "pectoral fins" (flippers) are broad, paddle-shaped, and pointed at the tips. Lacking a dorsal fin, they instead have a "dorsal hump" located about two-thirds of the way back on the body, and a series of 8 to 14 small bumps, known as "knuckles," between the dorsal hump and the tail flukes. The tail flukes are more than 15 feet (3 meters) wide, have S-shaped trailing edges, and a deep median notch.

Gray whales are frequently observed traveling alone or in small, unstable groups, although large aggregations may be seen on feeding and breeding grounds. Similar to other baleen whales, long-term bonds between individuals are rare. Gray whales are bottom feeders, and suck sediment and the "benthic" amphipods that are their prey from the sea floor. To do this, they roll on their sides and swim slowly along, filtering their food through coarse baleen plates, of which they have 130 to 180 on each side of the upper jaw. In doing so, they often leave long trails of mud behind them, and "feeding pits" in the sea floor.

Gray whales occur in two genetically distinct populations on the eastern and western sides of the North Pacific Ocean (Brownell Jr. et al. 2009; Burdin et al. 2011; Kanda et al. 2010; Lang et al.

2004; Lang et al. 2005; Lang et al. 2010; Leduc et al. 2002; Swartz et al. 2006; Weller et al. 2007; Weller et al. 2004; Weller et al. 2006).

Western North Pacific gray whales migrate annually along Asia during autumn, although migration routes are poorly known. Migration from summer foraging areas off the northeastern coasts of Sakhalin Island and south-eastern Kamchatka along the Japanese coasts to the South China Sea is suspected (Commission 2004; IWC 2003; Omura 1988; Tsidulko et al. 2005; Weller et al. 2008; Weller et al. 2012b).

Eastern and western North Pacific gray whales were once considered geographically separated along either side of the ocean basin, but recent photoidentification, genetic, and satellite tracking data refute this. Two western North Pacific gray whales have been satellite tracked from Russian foraging areas east along the Aleutian Islands, through the Gulf of Alaska, and south to the Washington State and Oregon coasts in one case (Mate et al. 2011) and to the southern tip of Baja California and back to Sakhalin Island in another (IWC 2012). Comparisons of eastern and western North Pacific gray whale catalogs have thus far identified 23 western gray whales occurring on the eastern side of the basin during winter and spring (Weller et al. 2013). Burdin et al. (2011) found an additional individual. During one field season off Vancouver Island, western gray whales were found to constitute 6 of 74 (8.1 percent) of photoidentifications (Weller et al. 2012a). In addition, two genetic matches of western gray whales off Santa Barbara, California have been made (Lang et al. 2011). Individuals have also been observed migrating as far as Central Baja Mexico (Weller et al. 2012b). Group sizes vary, but are roughly 2 (range 1 to 14) for non-calf groups and slightly larger for groups containing calves (Weller et al. 2007; Weller et al. 2004; Weller et al. 2006; Weller et al. 1999; Yakovlev and Tyurneva 2004).

Distribution

Western North Pacific gray whales exhibit extensive plasticity in their occurrence, shifting use areas within and between years, as well as over longer time frames, such as in response to oceanic climate cycles (e.g., El Nino-Southern Oscillation, Pacific Decadal Oscillation, and Arctic Oscillation) (Gardner and Chavez-Rosales 2000; Meier et al. 2007; Tyurneva et al. 2009; Vladimirov et al. 2006a; Vladimirov et al. 2006b; Vladimirov et al. 2005; Vladimirov et al. 2008; Vladimirov et al. 2009; Vladimirov et al. 2010; Weller et al. 2012b; Yablokov and Bogoslovskaya 1984; Yakovlev and Tyurneva 2005). Species distribution extends south along Japan, the Koreas, and China from the Kamchatka Peninsula (IWC 2003; Kato and Kasuya. 2002; Omura 1988; Reeves et al. 2008; Weller et al. 2003). Other possible range states include Vietnam, the Philippines, and Taiwan, although only historical whaling records support occurrence in these areas (Henderson 1990; Ilyashenko 2009). Range has likely contracted from the Koreas and other southern portions of the range versus pre-whaling periods. Prey availability and, to a lesser extent, sea ice extent, are probably strong influences on the habitats used by western North Pacific gray whales (Clarke and Moore 2002; Moore 2000).

Occurrence in the Offshore Portion of the NWTT action area

During visual surveys off the Washington coast from August 2004 through September 2008, there were a total of 55 gray whale sightings of 116 individuals (Oleson et al. 2009). Clear seasonal differences in gray whale distribution were noted based on three distinct time periods: (1) winter (December through January), corresponding to the timing of their southbound migration; (2) spring (February through April), corresponding to the timing of their northbound migration; and (3) summer/fall (May through October), a time when any gray whales present are primarily members of the Pacific Coast Feeding Group. Oleson et al. (2009) found significant differences in the sighting distributions between these three time periods, based on an analysis of distance from shore, distance from the shelf break, and water depth. During the winter southbound migration, gray whales were sighted mainly offshore, with an average distance of 18 mi. (29 km) from the coast. This compared to the spring northbound migration when the average distance was 6.2 mi. (10 km) from shore. During summer and fall, gray whale sightings were clustered in two areas, in and around the entrance to Grays Harbor, Washington and in an offshore area approximately 12 to 15 mi. (20 to 25 km) from shore (Oleson et al. 2009). These offshore sightings were unusual given that in the Pacific Northwest the Pacific Coast Feeding Group is typically close to shore (Calambokidis et al. 2002).

In 42 small boat surveys from Grays Harbor out to the 1,000-meter isobath off Quinault conducted in the summer over a five-year period between 2004 and 2009, there were eight sightings of gray whales (Oleson and Hildebrand 2012). Aerial surveys conducted in waters off Washington, Oregon, and Northern California in the spring, summer, and fall of 2011 and 2012 found gray whales present during all surveys periods (Adams et al. 2014b). Four of the five seasonal gray whale feeding areas located along the West Coast of the United States are near but not within the Offshore portion of the action area (Aquatic Mammals 2015; Calambokidis et al. 2015). The fifth feeding area—the Northwest Washington feeding area—partially overlaps with the offshore area, as shown in Figure 3.4-4 in the 2015 NWTT Final EIS/OEIS. This area is identified as important for feeding gray whales from May through November (approximately seven months) (Calambokidis et al. 2015). Fewer gray whale detections via passive acoustic monitoring have been obtained from mid-shelf and deep water sites along the coast of Washington and Oregon than from inshore sites along these coasts, where this species has been shown to be present nearly year-round (Emmons et al. 2019).

The occurrence of Eastern North Pacific gray whales and members of the Pacific Coast Feeding Group in the action area is considered to be seasonal and to most likely occur in the offshore portion of the action area during migrations between summer feeding grounds and winter breeding grounds (Carretta et al. 2013). Given their small population size and limited number of sightings off the U.S. west coast, the occurrence of Western North Pacific gray whales in the offshore portion of the action area is rare. Therefore, the potential for any stressor to cause an effect to this species is extremely unlikely (i.e., discountable) and this species is not considered further in this opinion.

Occurrence in Inland Waters of the NWTT action area

The identified gray whale "potential presence" migration area extends into and includes all U.S. waters from the entrance of the Strait of Juan de Fuca landward (Calambokidis et al. 2015). This portion of the potential presence migration area therefore overlaps all the Inland Waters portion of the action area. This potential presence area is identified as seasonally important from January through July, and October through December; approximately 10 months of the year. In addition, a biologically important feeding area also has been identified in northern Puget Sound located south and east of Whidbey Island and east of Camano Island to Everett (Calambokidis et al. 2015). This feeding area is used in the spring for two to three months, typically beginning in March and generally ending by June (Calambokidis et al. 2015). A geographic mitigation area has been established for gray whales feeding in the Inland Waters area.

As gray whales migrate between feeding and breeding grounds, a few enter the Strait of Juan de Fuca to feed in Inland Waters (Cascadia Research 2017c; Cogan 2015). Based on data collected from 1984 to 2011 during the feeding season, the observation rate increased to a peak in October in the Strait of Juan de Fuca (Scordino et al. 2017). Gray whales have been detected in Washington inland waters in all months of the year, with peak abundance from March through June (Calambokidis et al. 2017c; Calambokidis et al. 2010). Typically fewer than 20 gray whales are documented annually in the inland waters of Washington and British Columbia, based on a review of Orca Network (Calambokidis et al. 2015; Cogan 2015; Washington Department of Fish and Wildlife 2013). Western North Pacific gray whales in the Inland Waters portion of the action area are considered to have a rare presence. Therefore, the potential for any stressor to cause an effect to this species is extremely unlikely (i.e., discountable) and this species is not considered further in this opinion.

Occurrence in the Western Behm Canal, Alaska – NWTT action area

There are no identified gray whale feeding or migration areas near the Western Behm Canal; the closest being approximately 60 nautical miles to the southwest and out along the Pacific Coast of Southeast Alaska near Dixon Entrance (Ferguson et al. 2015). Gray whales were not observed during 1991 through 2007 surveys of the inland waters of southeast Alaska (Dahlheim et al. 2009), and they are considered extralimital in this region of the NWTT action area. Therefore, the potential for any stressor to cause an effect to this species is extremely unlikely and this species is not considered further in this opinion.

Critical Habitat

NMFS has not designated critical habitat for western North Pacific gray whale.

Conclusion

Acoustic modeling predicts that gray whales could be exposed to sonar and other active acoustic sources associated with training and testing activities in the offshore area. However, the

quantitative analysis of the presence of Western North Pacific gray whales in the action area is so low that it is extremely unlikely that any would be encountered. This is because, in all portions of the NWTT action area, it is very unlikely that Western North Pacific gray whales would be encountered, the potential for any stressor to cause an effect to this species is extremely unlikely and therefore discountable. Therefore, we determine that the Western North Pacific gray whale is not likely to be adversely affected by the proposed action.

Based on our determination that the proposed action is not likely to adversely affect this species, we also conclude that the proposed action is also not likely to jeopardize the continued existence of the western North Pacific gray whale. No critical habitat has been designated. As a result of these findings, North Pacific gray whale will not be discussed further in this opinion.

6.1.3 Steller Sea Lion Western Distinct Population Segment (DPS)

As of 2017, the best estimate of abundance of the Western Steller sea lion DPS in Alaska was 11,952 pups and 42,315 non-pups (total N_{min} = 54,267) (Muto et al. 2018a). This represents a large decline from counts in the 1950s (N = 140,000) and 1970s (N = 110,000).

Steller sea lion Western DPS site counts decreased forty percent from 1991 to 2000, an average annual decline of 5.4 percent; however, counts increased three percent between 2004 and 2008, the first recorded population increase since the 1970s (NMFS 2008f). However, there are regional differences in population growth rate, with positive trends in the eastern portion of the range, and negative trends west of Samalga Pass (approximately 170 degrees West) (Muto 2016). These trends indicate that overall, the Western DPS may be stable or exhibiting a slight negative trend as a whole.

Based on the results of genetic studies, the Steller sea lion population was reclassified into two distinct population segments: western and eastern. The data which came out of these studies indicated that the two populations had been separate since the last ice age (Bickham et al. 1998). Further examination of the Steller sea lions from the Gulf of Alaska (i.e., the Western DPS) revealed a high level of haplotypic diversity, indicating that genetic diversity had been retained despite the decline in abundance (Bickham et al. 1998).

Steller sea lions are distributed mainly around the coasts to the outer continental shelf along the North Pacific Ocean rim from northern Hokkaiddo, Japan through the Kuril Islands and Okhotsk Sea, Aleutian Islands and central Bering Sea, southern coast of Alaska and south to California. The Western DPS includes Steller sea lions that reside in the central and western Gulf of Alaska, Aleutian Islands, as well as those that inhabit the coastal waters and breed in Asia (e.g., Japan and Russia). The Western DPS also includes Steller sea lions that have been born at and west of Cape Suckling, Alaska (144 degrees West) but members of the Eastern (ESA-delisted) and Western DPS are known to cross this boundary (Muto et al. 2018a).

The species was listed as threatened in 1990 because of significant declines in population sizes (55 FR 49204). At the time, the major threat to the species was thought to be reduction in prey

availability. To protect and recovery the species, NMFS established the following measures: prohibition of shooting at or near sea lions; prohibition of vessel approach to within three nautical miles of specific rookeries, within 0.5 miles on land, and within sight of other listed rookeries; and restriction of incidental fisheries take to 675 sea lions annually in Alaskan waters. In 1997, the Western DPS was reclassified as endangered because it had continued to decline since its initial listing in 1990 (62 FR 24345). Despite the added protection (and an annual incidental fisheries take of twenty-six individuals), the DPS is likely still in decline (though the decline has slowed or stopped in some portions of the range). The reasons for the continued decline are unknown but may be associated with nutritional stress as a result of environmental change and competition with commercial fisheries. The DPS appears to have little resilience to future perturbations.

Occurrence in the Offshore Portion of the NWTT action area

Steller sea lions in the offshore portion of the action area are predominantly from the ESA-delisted Eastern DPS. While a relatively small number of sea lions from the Western DPS are known to migrate into Eastern DPS territory (east of 144 degrees West longitude), the majority are temporary visitors, returning to their natal rookeries for the breeding season. When in the Eastern DPS territory, sea lions from the Western DPS occur almost exclusively in the northern portion of southeast Alaska, several hundred kilometers north of the offshore area (Jemison et al. 2013). Sea lions from the Western DPS are "extremely unlikely" to be present south of Sumner Strait near Wrangell, Alaska (NMFS 2013d).

There are two unpublished reports of branded juvenile males from the Western DPS occurring in Washington state waters. In 2005, a one-year-old male was sighted at Tatoosh Island (at the entrance to the Strait of Juan de Fuca), and in July and August of 2013 a three-year-old male was sighted in the same general location and at Carrol Island (off southern Washington) (DeLong, 2018 cited in Navy 2019 (Jemison et al. 2013; Navy 2019a). Given that these were opportunistic sightings of two branded individuals separated by a span of 12 years, it is reasonable to assume that unbranded Steller sea lions from the Western DPS have also occurred in Washington State waters and that individuals, likely juvenile males, would occasionally be present in state waters in the future. Juvenile male sea lions wandering outside the core range of the population or stock is not uncommon (Fritz et al. 2016; Jemison et al. 2013; Raum-Suryan et al. 2004). Given that the presence of animals from the Western DPS is extremely unlikely in the action area, and that current research indicates exchange between the DPSs is centered in southeast Alaska, well north of the Offshore portion of the action area, we assume that sea lions from the Western DPS occurring in this portion of the action area would be few in number and should be considered rare. It is unlikely that individuals from the Western DPS that do venture into the action area would be present contemporaneously in time and space with Navy training and testing activities given they should be few in number. The occurrence of a Western DPS Steller sea lion within the offshore waters is considered rare. Therefore the potential for any stressor to cause an effect

to this species is extremely unlikely, and thus discountable, and this species is not considered further in this opinion.

Occurrence in Inland Waters of the NWTT action area

Steller sea lions from the ESA-delisted Eastern DPS, but not the endangered Western DPS, would occur mainly along the Washington coast from the Columbia River to Cape Flattery (Jeffries et al. 2000; Madson et al. 2017; Wiles 2015; Wright et al. 2017a). Smaller numbers of Steller sea lions use the Strait of Juan de Fuca, San Juan Islands, and Puget Sound south to the mouth of the Nisqually River in Thurston and Pierce counties (Wiles 2015). A total of 22 haulouts used by Steller sea lions from the Eastern DPS (and other pinnipeds) are located in Washington inland waters, and an additional 6 haulouts are located on the Canadian side of the Strait of Juan de Fuca and southern Georgia Strait (Jeffries 2014; Wiles 2015). As noted above, a few Steller sea lions from the Western DPS may occasionally be present in the action area (DeLong, 2018 cited in Navy 2019 (Jemison et al. 2013; Navy 2019f), and, based on the observed behavior of Steller sea lions from the Eastern DPS, it is conceivable that Western DPS individuals could also enter the Inland Waters area and co-occur at haulouts frequented by Steller sea lions from the Eastern DPS.

Steller sea lions are seasonally present in Puget Sound. While nearly all, if not all, Steller sea lions occurring in Puget Sound are expected to be from the Eastern DPS, the seasonal occurrence of any Western DPS sea lions is assumed to follow a similar seasonal cycle. An estimate of several dozen to a few hundred Steller sea lions (mostly males) are present in Puget Sound at any time with peak abundance in fall and winter (Smultea et al. 2017). No sea lions were sighted from May through July during aerial surveys of Puget Sound from 2014 through 2016 (Smultea et al. 2017). However, aerial surveys conducted in 2013 and 2014 recorded peak abundance of over 600 Steller sea lions on Tatoosh Island at the mouth of the Strait of Juan de Fuca in late July (Jeffries 2014). Jeffries (2014) identified five winter haulout sites in Puget Sound used by Steller sea lions, ranging north to south from immediately south of Port Townsend (near Admiralty Inlet) to southern Puget Sound near Olympia. The highest total count at these Puget Sound haulouts was 50 Steller sea lions recorded in November (Jeffries 2014). Although Steller sea lions may occur throughout Puget Sound, they have generally been observed in greater numbers in Admiralty Inlet (Smultea et al. 2017).

Race Rocks is a well-established winter haulout site on the Canadian side of the Strait of Juan de Fuca used by hundreds of Steller sea lions as they enter inland waters to feed on herring (Edgell and Demarchi 2012). Peak abundance at Race Rocks based on sightings from 1997 to 2009 occurs in October. Based solely on abundance and the presumption of co-occurrence, a small number of Steller sea lions from the Western DPS may be present at Race Rocks in fall and winter. During the summer breeding season, very few, if any, Steller sea lions from the Western DPS would be expected in the Inland Waters portion of the action area (Jeffries 2014; Smultea et

al. 2017). The occurrence of a Western DPS Steller sea lion within the Inland Waters is considered extralimital. Therefore the potential for any stressor to cause an effect to this species is extremely unlikely, and thus discountable.

Occurrence in the Western Behm Canal, Alaska

Steller sea lions from the Eastern DPS occurring in U.S. waters are most prevalent in southeast Alaska, where over 65 percent of the U.S. population resides (Navy 2019a). The majority of rookeries and haulouts in southeast Alaska are located over 250 kilometers north of the Behm Canal area (Jemison et al. 2013), and there are no haulouts in or near Behm Canal. Steller sea lions from the Western DPS are "extremely unlikely" to be present south of Sumner Strait (DeLong 2018 cited in (Fritz et al. 2016; Jemison et al. 2013; Navy 2019a; NMFS 2013d), which is approximately 70 nautical miles north of waters adjacent Behm Canal. In southeast Alaska, the majority of the documented movement of Steller sea lions from the Western DPS occurs in northern Southeast Alaska at two rookeries (i.e., Graves Rock and White Sisters) characterized as a mixing zone region for sea lions from both DPSs (Jemison et al. 2013). Hastings et al. (2020) estimated that the southeastern movement probability of juvenile Western DPS Steller sea lions from this mixing zone to the area around the Western Behm Canal was 1.1 percent, while that of juveniles moving out of Lynn Canal, Frederick Sound, and Biali Rocks into the area around the Western Behm Canal was zero percent. Hastings et al. (2020) also estimated that animals older than 4 years of age had a zero percent chance of moving into this area from the mixing zone. Only one or two individual animals from the Western DPS have been documented at haulouts along Alaska's Pacific coast as far south as Forrester Island (Jemison et al. 2013), which is approximately 100 nautical miles by sea from the entrance to Western Behm Canal. The occurrence of a Western DPS Steller sea lion within the Western Behm Canal is considered rare. Therefore the potential for any stressor to cause an effect to this species is extremely unlikely, and thus discountable.

Critical Habitat

In 1997, NMFS designated critical habitat for the Steller sea lion (58 FR 45269). The critical habitat includes specific rookeries, haulouts, and associated areas, as well as three foraging areas that are considered to be essential for the health, continued survival, and recovery of the species.

In Alaska, areas include major Steller sea lion rookeries, haulouts and associated terrestrial, air, and aquatic zones. Critical habitat includes a terrestrial zone extending 3,000 feet (0.9 kilometers) landward from each major rookery and haulout; it also includes air zones extending 3,000 feet (0.9 kilometers) above these terrestrial zones and aquatic zones. Aquatic zones extend 3,000 feet (0.9 kilometers) seaward from the major rookeries and haulouts east of 144 degrees West. In addition, NMFS designated special aquatic foraging areas as critical habitat for the Steller sea lion. These areas include the Shelikof Strait (in the Gulf of Alaska), Bogoslof Island, and Seguam Pass (the latter two are in the Aleutians). These sites are located near Steller sea lion

abundance centers and include important foraging areas, large concentrations of prey, and host large commercial fisheries that often interact with the species.

The Navy's training and testing activities would not occur in the designated critical habitat nor would the activities be expected to have any impacts to the critical habitat including its essential PBFs. Therefore, Navy NWTT activities in the action area would have no effect on designated critical habitat for the Western DPS of Steller sea lion.

Conclusion

As discussed above, the occurrence of a Western DPS Steller sea lion within Western Behm Canal and the offshore waters portion of the action area is considered rare, and this species is considered extralimital in the inland portion. Therefore, potential effects from the proposed action are discountable, and we determine that the Western DPS Steller sea lion is not likely to be adversely affected by the proposed action.

Based on our determination that the proposed action is not likely to adversely affect this species, we further conclude that the proposed action is also not likely to jeopardize the continued existence of Western DPS Steller sea lion. As a result of this finding, Western DPS Steller sea lion will not be discussed in the remainder of this opinion.

Due to the lack of designated critical habitat for Western DPS Steller sea lion in the action area, the proposed action would have no effect on the designated critical habitat for Western DPS Steller sea lion, hence we conclude that the proposed action would not destroy or adversely modify Western DPS Steller sea lion designated critical habitat. As a result of this finding, Western DPS Steller sea lion designated critical habitat will not be discussed further in this opinion.

6.1.4 Ozette Lake Sockeye Salmon

The sockeye salmon is an anadromous species, although some sockeye salmon spend their entire lives (about five years) in freshwater. Adult sockeye salmon are about 1.5 to 2.5 feet long and eight pounds. Sockeye salmons are bluish black with silver sides when they are in the ocean, and they turn bright red with a green head when they are spawning. On March 25, 1999, NMFS listed the Ozette Lake sockeye salmon ESU as threatened (64 FR 14528) and reaffirmed the ESU's status as threatened on June 28, 2005(*Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs*).

Most sockeye salmon exhibit a lake-type life history (i.e., they spawn and rear in or near lakes), though some exhibit a river-type life history. Spawning generally occurs in late summer and fall, but timing can vary greatly among populations. In lakes, sockeye salmon commonly spawn along "beaches" where underground seepage provides fresh oxygenated water. Females spawn in three to five redds over a couple of days. Incubation period is a function of water temperature

and generally lasts 100-200 days (Burgner 1991). Sockeye salmon spawn once, generally in late summer and fall, and then die (semelparity).

Sockeye salmon fry primarily rear in lakes; river-emerged and stream-emerged fry migrate into lakes to rear. In the early fry stage from spring to early summer, juveniles forage exclusively in the warmer littoral (i.e., shoreline) zone where they depend mostly on fly larvae and pupae, copepods, and water fleas. Sub-yearling sockeye salmon move from the littoral habitat to a pelagic (i.e., open water) existence where they feed on larger zooplankton; however, flies may still make up a substantial portion of their diet. From one to three years after emergence, juvenile sockeye salmon generally rear in lakes, though some river-spawned sockeye salmon may migrate to sea in their first year. Juvenile sockeye salmon feeding behaviors change as they transition through life stages after emergence to the time of smoltification. Distribution in lakes and prey preference is a dynamic process that changes daily and yearly depending on many factors including water temperature, prey abundance, presence of predators and competitors, and size of the juvenile. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye salmon populations (lower than 52 degree North latitude) and as late as early July in northern populations (62 degrees North latitude) (Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending one to four years at sea. The diet of adult salmon consists of amphipods, copepods, squid and other fish.

The historical abundance of the Ozette Lake ESU of sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). Escapement estimates (run size minus broodstock take) from 1996-2006 range from a low of 1,404 in 1997 to a high of 6,461 in 2004, with a median of approximately 3,800 sockeye salmon per year (geometric mean: 3,353) (Rawson et al. 2009). Current abundance estimates for Ozette Lake ESU sockeye salmon are presented in Table 65 below.

Table 65. Abundance Estimates for the Ozette Lake ESU of Sockeye salmon (NMFS 2019c).

Production	Life Stage	Abundance
Natural and Hatchery (Clipped and Intact Adipose)	Adult	5,036
Natural	Juvenile	1,037,787
Listed Hatchery Adipose Clipped	Juvenile	45,750
Listed Hatchery Intact Adipose	Juvenile	259,250

Productivity has fluctuated up and down over the last few decades, but overall appears to have remained stable (NWFSC 2015b). Given the degree of uncertainty in the abundance estimates, any interpretation of trends of small magnitude or over short time periods is speculative. (NWFSC 2015b).

For the Ozette Lake sockeye salmon ESU, the proportion of beach spawners is likely low; therefore, hatchery-originated fish are not likely to greatly affect the genetics of the naturally-spawned population. However, Ozette Lake sockeye salmon have a relatively low genetic diversity compared to other sockeye salmon populations examined in Washington State (NWFSC 2015b). Genetic differences do occur among age cohorts. However, because different age groups do not reproduce together, the population may be more vulnerable to significant reductions in population structure due to catastrophic events or unfavorable conditions affecting a single year class.

The Ozette Lake sockeye salmon ESU is composed of one historical population with multiple spawning aggregations and two populations from the Umbrella Creek and Big River sockeye salmon hatchery programs (NWFSC 2015b). Historically, at least four lake beaches were used for spawning; today only two beach spawning locations: Allen's and Olsen's Beach. Additionally, spawning occurs in the two tributaries of the hatchery programs (NWFSC 2015b). The Umbrella creek population is a large component of the total population (averaging over 50 percent for the last decade of data).

NMFS listed the Ozette Lake sockeye salmon ESU because of habitat loss and degradation from the combined effects of logging, road building, predation, invasive plant species, and overharvest. Ozette Lake sockeye salmon have not been commercially harvested since 1982 and only minimally harvested by the Makah Tribe since 1982 (0 to 84 fish per year); there is no known marine fishing of this ESU. Overall abundance is substantially below historical levels, and whether the decrease in abundance is a result of fewer spawning aggregations, lower abundances in each aggregation, or a combination of both factors is unknown. Regardless, this ESU's viability has not improved, and the ESU would likely have a low resilience to additional perturbations. However, recovery potential for the Ozette Lake sockeye salmon ESU is good, particularly because of protections afforded it based on the lake's location within a Olympic National Park (NWFSC 2015b).

It is thought that sockeye follow a similar migration pattern as chum once they enter the ocean, moving north and west along the coast, and have moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). Previously, French et al. (1976a) summarized the general migration pattern of sockeye salmon originating in the various tributaries of the northeastern Pacific Ocean from the Alaska Peninsula to the Columbia River. Tag recovery data indicated a general mixing of these stocks during their residence in the northeastern Pacific Ocean. These fish primarily occur east of 160 degrees West and north of 48 degrees North (north of the NWTT action area). Pearcy and Fisher (1990) observed the highest catch per unit effort of juvenile sockeye inshore of 37 km, though some were caught over 55 km offshore. They noted that, similar to juvenile chum salmon, juvenile sockeye salmon were less abundant than either coho or salmon off the Oregon and Washington coast.

As discussed in the *Effects of the Action* (Section 8 below), the only stressor we determined is likely to adversely affect ESA-listed salmonids is explosive detonation. Since the Navy will not conduct explosive activities in the Olympic Coast National Marine Sanctuary or the Juan de Fuca Eddy Marine Species Mitigation Area and considering the oceanic migration path described above, Lake Ozette sockeye would be expected to be north of the action area before they would be far enough offshore to co-occur with Navy explosive activities. Therefore, the likelihood of affecting Ozette Lake sockeye salmon is extremely unlikely and thus discountable.

We conclude that Navy training and testing activities in the NWTT action area are not likely to adversely affect the Ozette Lake sockeye salmon, and this ESU, including effects to the ESU, is not considered further in this opinion. Based on our determination that the proposed action is not likely to adversely affect this ESU, we also conclude that the proposed action is also not likely to jeopardize the continued existence of the Ozette Lake sockeye salmon ESU.

Critical habitat for the Ozette Lake ESU of sockeye salmon encompasses areas along the Ozette River, Ozette Lake, Umbrella Creek, Big River, and Crooked Creek (NMFS 2005b). All areas of critical habitat occur outside of the proposed action area. Due to this, the proposed action would have no effect on the designated critical habitat for Ozette Lake sockeye salmon ESU.

Based on our determination that the proposed action is not likely to adversely affect this species, we also conclude that the proposed action is also not likely to jeopardize the continued existence of the Ozette Lake sockeye salmon ESU. As a result of these findings, the Ozette Lake ESU of sockeye salmon will not be discussed further in this opinion.

Due to the lack of designated critical habitat for Ozette Lake sockeye salmon ESU in the action area, the proposed action would have no effect on the designated critical habitat for Ozette Lake sockeye salmon ESU, hence we conclude that the proposed action would not destroy or adversely modify Ozette Lake sockeye salmon ESU designated critical habitat. As a result of this finding, Ozette Lake sockeye salmon ESU designated critical habitat will not be discussed further in this opinion.

6.1.5 Steelhead – Southern California DPS

Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead grow to 55 pounds (25 kilogram) in weight and 45 in (120 cm) in length, though average size is much smaller. On August 18, 1997 NMFS listed the Southern California DPS of steelhead as endangered (62 FR 43937) and reaffirmed the DPS's status as endangered on January 5, 2006 (71 FR 5248).

There is limited life history information for Southern California steelhead. In general, migration and life history patterns of Southern California steelhead populations are dependent on rainfall and stream flow (Moore 1980). Steelhead within this DPS can withstand higher temperatures

compared to populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead compared to the more northerly populations (Moore 1980).

Limited information exists on Southern California steelhead runs. Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS historically. In contrast, less than 500 adults are estimated to occupy the same four waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults (Busby et al. 1996).

There are currently no population trend estimates and limited information is available regarding the structural and genetic diversity for the Southern California DPS steelhead. There is little new evidence to indicate that the status of the Southern California DPS of steelhead has changed appreciably in either direction since the last status review (Williams et al. 2011c). The extended drought and the recent genetic data documenting the high level of introgression and extirpation of native *O. mykiss* stocks in the southern portion of the DPS has elevated the threats level to the already endangered populations; the past drought, and the lack of comprehensive monitoring, has also limited the ability to fully assess the status of individual populations and the DPS as whole.

Given the extremely low abundance of ESA-listed Southern California steelhead in general and within the NWTT action area and the limited likelihood of co-occurrence with training and testing stressors, the likelihood of the proposed action adversely affecting Southern California steelhead is extremely unlikely and therefore discountable. We conclude that Navy training and testing activities in the NWTT action area are not likely to adversely affect the Southern California steelhead DPS.

Based on our determination that the proposed action is not likely to adversely affect this species, we conclude that the proposed action is also not likely to jeopardize the continued existence of the Southern California steelhead DPS. As a result of these findings, Southern California steelhead DPS will not be discussed further in this opinion.

Due to the lack of designated critical habitat for Southern California steelhead DPS in the action area, the proposed action would have no effect on the designated critical habitat for Southern California steelhead DPS, hence we conclude that the proposed action would not destroy or adversely modify Southern California steelhead DPS designated critical habitat. As a result of this finding, Southern California steelhead DPS designated critical habitat will not be discussed further in this opinion.

6.2 Status of Species Likely to be Adversely Affected

This section examines the status of each species that are likely to be adversely affected by the proposed action. The status includes the existing level of risk that the ESA-listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing

decisions. The species status section helps to inform the description of the species' current "reproduction, numbers, or distribution," which is part of the jeopardy determination as described in 50 C.F.R. §402.02. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on NMFS' website: (https://www.fisheries.noaa.gov/species-directory/threatened-endangered), among others.

This section also describes the designated (or proposed) critical habitat in the action area that may be affected by the proposed action including the essential physical and biological features (PBFs) that contribute to the conservation value of the critical habitat.

6.2.1 Blue Whale

The blue whale is a widely distributed baleen whale found in all major oceans (Figure 33). Blue whales are the largest animal on earth and distinguishable from other whales by a long-body and comparatively slender shape, a broad, flat "rostrum" when viewed from above, proportionally smaller dorsal fin, and are a mottled gray color that appears light blue when seen through the water. Most experts recognize at least three subspecies of blue whale, *B. m. musculus*, which occurs in the Northern Hemisphere, *B. m. intermedia*, which occurs in the Southern Ocean, and *B. m. brevicauda*, a pygmy species found in the Indian Ocean and South Pacific. The blue whale was originally listed as endangered on December 2, 1970.

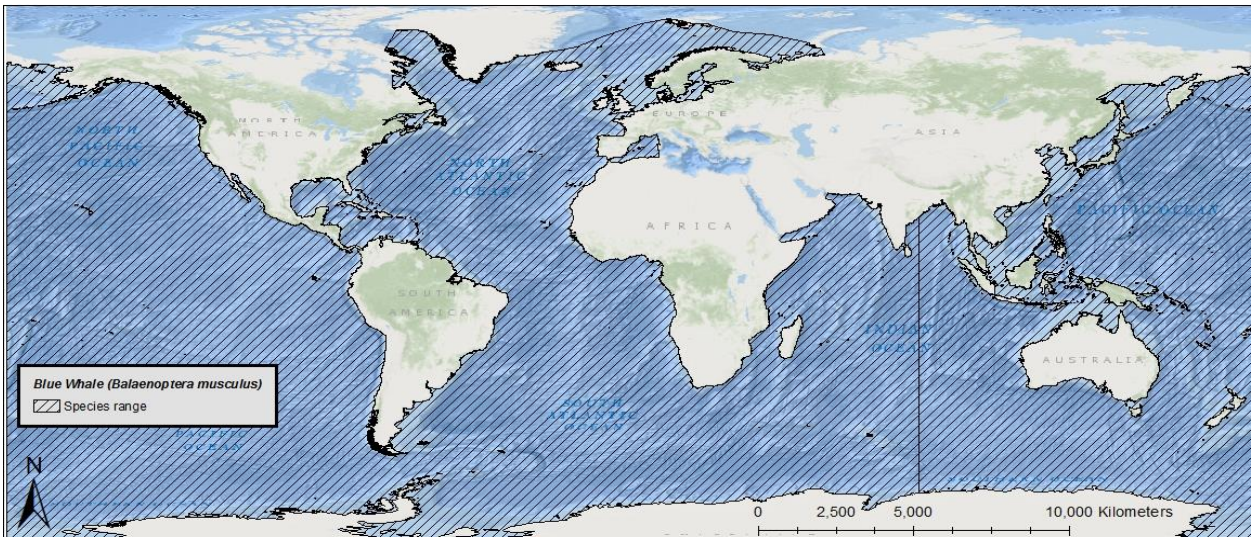


Figure 33. Map identifying the range of the endangered blue whale.

Information available from the recovery plan (NMFS 1998), recent SARs (Carretta et al. 2018a; Hayes et al. 2018; Muto et al. 2018a), and status review (COSEWIC 2002) were used to summarize the life history, population dynamics and status of the species as follows.

Distribution

In general, blue whale distribution is driven largely by food requirements; blue whales are more likely to occur in waters with dense concentrations of their primary food source, krill. While they can be found in coastal waters, they are thought to prefer waters further offshore. In the North Atlantic Ocean, the blue whale range extends from the subtropics to the Greenland Sea. They are most frequently sighted in waters off eastern Canada with a majority of sightings taking place in the Gulf of St. Lawrence. In the North Pacific Ocean, blue whales range from Kamchatka to southern Japan in the west and from the Gulf of Alaska and California to Costa Rica in the east. They primarily occur off the Aleutian Islands and the Bering Sea. In the northern Indian Ocean, there is a "resident" population of blue whales with sightings being reported from the Gulf of Aden, Persian Gulf, Arabian Sea, and across the Bay of Bengal to Burma and the Strait of Malacca. In the Southern Hemisphere, distributions of subspecies (*B. m. intermedia* and *B. m. brevicauda*) seem to be segregated. The subspecies *B. m. intermedia* occurs in relatively high latitudes south of the "Antarctic Convergence" (located between 48°S and 61°S latitude) and close to the ice edge. The subspecies *B. m. brevicauda* is typically distributed north of the Antarctic Convergence.

Blue whales appear to migrate to waters offshore of Washington, Oregon, and northern California to forage. Thus far, blue whales are associated with deeper, pelagic waters in the action area; they have not been reported to occur proximate to the coast or in Puget Sound itself. Širović et al. (2012); (2011) reported detection of blue whale vocalizations in the offshore waters of Washington from late fall through February. Oleson and Hildebrand (2011) also reported visual blue whale sightings in these areas. Although a resident population of blue whales might occur off the coast of Vancouver Island throughout the year (Burtenshaw et al. 2004), most blue whales that occur in the action area for this consultation appear to migrate between summer, foraging areas and winter rearing areas along the Pacific Coast of the United States. That seasonal migration brings them to waters off the NWTRC (with some individuals continuing north to the Gulf of Alaska) during the warm, summer season with a southward migration to waters off California, south to Central America, during the winter season (Calambokidis et al. 2009a; Gregr et al. 2000; Mate et al. 1998).

Occurrence in the Offshore Portion of the NWTT action area

The U.S. west coast is known to be a feeding area for blue whales during summer and fall (Bailey et al. 2010; Calambokidis et al. 2009a), although primary occurrence for this species is south of 44 degrees North (Forney et al. 2012; Hamilton et al. 2009; Sirovic et al. 2015). Blue whales feed in the area as late as October, although fewer individuals are seen because the majority of the population migrates south. Acoustic data collected by Sound Surveillance System hydrophones reveal that males call during the fall in this area (Stafford et al. 2001). More recently, Navy-funded acoustic monitoring studies have detected blue whales along the Washington coast between August and February, with peak calling from October to December, and no detections between April and July (Širović et al. 2012; Širović et al. 2011). An individual

blue whale was also sighted off Washington in January 2009, in waters approximately 3,281 feet (1,000 meters) deep (Oleson et al. 2012).

In 42 small boat surveys from Grays Harbor out to the 1,000 meter isobath off Quinault, conducted over a five-year period in the summer between 2004 and 2009, there was one sighting of a blue whale (Oleson et al. 2012). In December 2011, six blue whales were sighted off the Washington coast, which was the highest number of blue whales ever sighted off that coast and only the third confirmed sighting in 50 years (Cascadia Research 2012). Model predictions based on tagging data indicated the highest blue whale presence off Washington in June and July with a presence into November (Hazen et al. 2016). Aerial surveys conducted in waters off southern Washington, Oregon, and Northern California covering the approximate nearshore half of the action area in the spring, summer, and fall of 2011 and 2012, encountered a total of 16 blue whales only during the fall and only off Oregon (Adams et al. 2014a). Acoustic monitoring in waters off the coast of Washington suggested a yearly seasonal pattern of blue whale presence from summer through winter (calls were absent from approximately March through July) (Debich et al. 2014; Kerosky et al. 2013; Oleson and Hildebrand 2012; Soule and Wilcock 2013; Trickey et al. 2015b; Wiggins et al. 2017b). This seasonality is consistent with the data from satellite-tagged blue whales being in the NWTT action area from August through November (summer through fall) (U.S. Department of the Navy 2018a).

Between 2014 and 2017, satellite tags were placed on 63 blue whales from the same stock in the waters off the U.S. West Coast, including in the offshore portion of the NWTT action area (Mate et al. 2017; U.S. Department of the Navy 2018a). The NWTT action area was used by only nine of the 63 tagged blue whales with an average of approximately 23 days spent in the NWTT action area; only one of these 63 blue whales ventured as far north as the W-237 Warning Area in waters off Washington (U.S. Department of the Navy 2018a). The Navy has determined the blue whale's occurrence in the offshore area would be seasonal.

Another study (Mate et al. 2015) was completed to determine movement patterns, occurrence, and residence times of blue and fin whales within Navy training and testing areas along the U.S. West Coast. Tagging occurred off the coast of southern California in August and September 2014. The study concluded that there was a great deal of individual variation in the tracks of both blue and fin whales, with blue whale locations extending from the northern tip of Vancouver Island in British Columbia to the Costa Rica Dome area off Central America, and those of fin whales extending from southern Oregon to central Baja California. The area of highest use for blue whales was between Point Dume and Mugu Canyon off southern California, out to approximately 30 km from shore. For fin whales, areas of highest use occurred south of San Miguel Island, approximately 100 km offshore, and approximately 70 to 80 km offshore along the south-central California coast between Arroyo Grande and Big Sur.

Mate et al. (2015) indicated that both blue and fin whales were tracked in the NWTRC and the Southern California Training Range Complex (SOCAL), but neither species traveled into the Gulf of Alaska Temporary Maritime Activities Area. Only one blue whale had locations within area W237 within the NWTRC area (in August, September, and October). Blue whale presence was observed in both SOCAL and NWTRC in August, September, October, and November. Fin whale locations occurred in SOCAL in all 5 months in which they were tracked (August, September, October, November, and December), but in only 2 months in NWTRC (August and September). Eighteen blue whales spent 1 to 48 percent (37 to 414 hours) of their tracking periods within the SOCAL, while four blue whales spent 3 to 45 percent (15 to 1,249 hours) of their tracking periods within the NWTRC. Finally, Mate et al. (2015) concluded that four fin whales spent from less than 1 to 39 percent (1 to 956 hours) of their total tracking periods in SOCAL, and one fin whale spent 51 percent (811 hours) of its tracking period in NWTRC.

Occurrence in the Inland Waters of the NWTT action area

Blue whales are not expected to occur within the inland portion of the action area since it is well inland of the areas normally inhabited by this species.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Blue whales are not expected to occur within the SEAFAC region of the action area since it is well inland of the areas normally inhabited by this species.

Population Structure

For this and all subsequent marine mammal species in this section, the term "population" refers to groups of individuals whose patterns of increase or decrease in abundance over time are determined by internal dynamics (births resulting from sexual interactions between individuals in the group and deaths of those individuals) rather than external dynamics (immigration or emigration). This definition is a reformulation of definitions articulated by Futuymda (1986) and Wells and Richmond (1995) and is more restrictive than those uses of 'population' that refer to groups of individuals that co-occur in space and time but do not have internal dynamics that determine whether the size of the group increases or decreases over time (see review by Wells and Richmond 1995). The definition we apply is important to ESA section 7 consultations because such concepts as 'population decline,' 'population collapse,' 'population extinction,' and 'population recovery' apply to the restrictive definition of 'population' but do not explicitly apply to alternative definitions. As a result, we do not treat the different whale "stocks" recognized by the International Whaling Commission or other authorities as populations unless those distinctions were clearly based on demographic criteria. We do, however, acknowledge those "stock" distinctions in these narratives.

At least three subspecies of blue whales have been identified based on body size and geographic distribution (*B. musculus intermedia,* which occurs in the higher latitudes of the Southern Oceans, *B. m. musculus,* which occurs in the Northern Hemisphere, and *B. m. brevicauda* which

occurs in the mid-latitude waters of the southern Indian Ocean and north of the Antarctic convergence), but this consultation will treat them as a single entity. Readers who are interested in these subspecies will find more information in Gilpatrick et al. (1997), Kato et al. (1995), Omura et al. (1970), and Ichihara (1966).

In addition to these subspecies, the International Whaling Commission's Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan 1991), although there is increasing evidence that there may be more than one blue whale population in the Pacific Ocean (Barlow 1995; Gilpatrick et al. 1997; Mizroch et al. 1984; Ohsumi and Masaki. 1972). For example, studies of the blue whales that winter off Baja California and in the Gulf of California suggest that these whales are morphologically distinct from blue whales of the western and central North Pacific (Gilpatrick et al. 1997), although these differences might result from differences in the productivity of their foraging areas more than genetic differences (Barlow et al. 1997; Calambokidis et al. 1990; Sears 1987).

Little genetic data exist on blue whales globally. Data from Australia indicates that at least populations in this region experienced a recent genetic bottleneck, likely the result of commercial whaling, although genetic diversity levels appear to be similar to other, non-threatened mammal species (Attard et al. 2010). Consistent with this, data from Antarctica also demonstrate this bottleneck but high haplotype diversity, which may be a consequence of the recent timing of the bottleneck and blue whales long lifespan (Sremba et al. 2012). Data on genetic diversity of blue whales in the Northern Hemisphere are currently unavailable. However, genetic diversity information for similar cetacean population sizes can be applied. Stocks that have a total population size of 2,000 to 2,500 individuals or greater provide for maintenance of genetic diversity resulting in long-term persistence and protection from substantial environmental variance and catastrophes. Stocks that have a total population 500 individuals or less may be at a greater risk of extinction due to genetic risks resulting from inbreeding. Stock populations at low densities (less than 100) are more likely to suffer from the 'Allee' effect, where inbreeding and the heightened difficulty of finding mates reduces the population growth rate in proportion with reducing density.

Blue whales from both the eastern and western North Pacific have been heard, tracked, or harvested in waters off Kodiak Island; acoustic detections are made in the Gulf of Alaska from mid-July to mid-December and a peak from August through November (COSEWIC 2002; Ivashin and Rovnin. 1967; Moore et al. 2006; Stafford 2003; Stafford et al. 2007; Yochem and Leatherwood 1985). Although acoustic detections in the Gulf of Alaska were absent since the late 1960s, recordings have increased during 1999 to 2002 and a few sightings have been made in the northern Gulf of Alaska (Calambokidis et al. 2009a; Moore et al. 2006; NOAA 2004; Stafford 2003; Stafford et al. 2007; Stafford and Moore 2005). However, surveys in the western Gulf of Alaska and east of Kodiak Island have not found blue whales (Rone et al. 2010; Zerbini et al. 2006). Blue whales are rarely observed in nearshore Alaskan waters, but seem to prefer continental shelf edge waters; such areas in the Gulf of Alaska were formerly feeding grounds

for blue whales prior to severe depletion (Rice and Wolman. 1982). Call detections of blue whales from the western North Pacific indicate a greater likelihood of these individual occurring southwest of Kodiak Island (Stafford 2003). A population of blue whales that has distinct vocalizations inhabits the northeast Pacific from the Gulf of Alaska to waters off Central America (Gregr et al. 2000; Mate et al. 1998; Stafford 2003). We assume that this population is the one affected by the activities considered in this Opinion.

Abundance Estimate

The global, pre-exploitation estimate for blue whales is approximately 181,200 (IWC 2007). Current estimates indicate approximately 5,000 to 12,000 blue whales globally (IWC 2007). Blue whales are separated into populations by ocean basin in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere. There are three stocks of blue whales designated in United States waters: the Eastern North Pacific Ocean, Central North Pacific Ocean, and Western North Atlantic Ocean. Due to the location of the action, the Eastern North Pacific stock of blue whales is most likely to be in the action area. The minimum population size for eastern North Pacific Ocean blue whales is 1,050; the most recent abundance estimate is 1,496 whales (Carretta et al. 2020).

Natural Threats

Natural causes of mortality in blue whales are largely unknown, but probably include predation and disease (not necessarily in their order of importance). Blue whales are known to become infected with the nematode *Carricauda boopis* (Baylis 1928), which are believed to have caused fin whales to die as a result of renal failure (Lambertsen 1986); see additional discussion under Fin whales). Killer whales and sharks are also known to attack, injure, and kill very young or sick fin and humpback whales and likely hunt blue whales as well (Ford and Reeves 2008; Perry et al. 1999).

Anthropogenic Threats

Two human activities are known to threaten blue whales; whaling and shipping. Historically, whaling represented the greatest threat to every population of blue whales and was ultimately responsible for listing blue whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing blue, fin, and other large whales using a fairly primitive open-water netting technique (Tonnessen and Johnsen 1982). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species.

From 1889 to 1965, whalers killed about 5,761 blue whales in the North Pacific Ocean (Hill et al. 1999). From 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984). Evidence of a population decline was seen in the catch data from Japan. In 1912, whalers captured 236 blue whales; in 1913, 58 blue whales; in 194, 123 blue whales; from 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984).

In the eastern North Pacific, whalers killed 239 blue whales off the California coast in 1926. And, in the late 1950s and early 1960s, Japanese whalers killed 70 blue whales per year off the Aleutian Islands (Mizroch et al. 1984).

Although the International Whaling Commission banned commercial whaling in the North Pacific in 1966, Soviet whaling fleets continued to hunt blue whales in the North Pacific for several years after the ban. Surveys conducted in these former-whaling areas in the 1980s and 1990s failed to find any blue whales (Forney and Brownell Jr. 1996). By 1967, Soviet scientists wrote that blue whales in the North Pacific Ocean (including the eastern Bering Sea and Prince William Sound) had been so overharvested by Soviet whaling fleets that some scientists concluded that any additional harvests were certain to cause the species to become extinct in the North Pacific (Latishev 2007). As its legacy, whaling has reduced blue whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push blue whales closer to extinction. Otherwise, whaling currently does not threaten blue whale populations.

In 1980, 1986, 1987, and 1993, ship strikes have been implicated in the deaths of blue whales off California (Barlow 1997). More recently, Berman-Kowalewski et al. (2010) reported that between 1988 and 2007, 21 blue whale deaths were reported along the California coast, typically one or two cases annually. In addition, several photo-identified blue whales from California waters were observed with large scars on their dorsal areas that may have been caused by ship strikes. Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach, and speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than whales that are not feeding (Sears 1983). Within the St. Lawrence Estuary, blue whales are believed to be affected by large amounts of recreational and commercial vessel traffic. Blue whales in the St. Lawrence appeared more likely to react to these vessels when boats made fast, erratic approaches or sudden changes in direction or speed (Edds and Macfarlane 1987).

Increasing oceanic noise may impair blue whale behavior. Although available data do not presently support traumatic injury from sonar, the general trend in increasing ambient low-frequency noise in the deep oceans of the world, primarily from ship engines, could impair the ability of blue whales to communicate or navigate through these vast expanses (Aburto et al. 1997; Clark 2006). Blue whales off California altered call levels and rates in association with changes in local vessel traffic (Mckenna 2011). There is a paucity of contaminant data regarding blue whales. Available information indicates that organochlorines, including dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCB), benzene hexachloride (HCH), hexachlorobenzene (HCB), chlordane, dieldrin, methoxychlor, and mirex have been isolated from blue whale blubber and liver samples (Gauthier et al. 1997; Metcalfe et al. 2004). Contaminant transfer between mother and calf occurs, meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional

contaminant loads during life and passing higher loads to the next generation (Gauthier et al. 1997; Metcalfe et al. 2004). This is supported by ear plug data showing maternal transfer of pesticides and flame retardants in the first year of life (Trumble et al. 2013). These data also support pulses of mercury in body tissues of the male studied (Trumble et al. 2013).

Status and Trends

Blue whales (including all subspecies) were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Blue whales are listed as endangered on the IUCN Red List of Threatened Animals (IUCN 2010). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA.

The blue whale is endangered as a result of past commercial whaling. In the North Atlantic Ocean, at least 11,000 blue whales were harvested from the late 19th to mid-20th centuries. In the North Pacific Ocean, at least 9,500 whales were killed between 1910 and 1965. Commercial whaling no longer occurs, but blue whales are threatened by vessel strikes, entanglement in fishing gear, pollution, harassment due to whale watching, and reduced prey abundance and habitat degradation due to climate change. Because populations appear to be increasing in size, the species appears to be somewhat resilient to current threats; however, the species has not recovered to pre-exploitation levels.

It is difficult to assess the current status of blue whales globally because (1) there is no general agreement on the size of the blue whale population prior to whaling and (2) estimates of the current size of the different blue whale populations vary widely. We may never know the size of the blue whale population in the North Pacific prior to whaling, although some authors have concluded that their population numbers about 200,000 animals before whaling. Similarly, estimates of the global abundance of blue whales are uncertain. Since the cessation of whaling, the global population of blue whales has been estimated to range from 11,200 to 13,000 animals (Maser et al. 1981). These estimates, however, are more than 20 years old.

The current best available abundance estimate for the eastern North Pacific population of blue whales that occur off California, Oregon, and Washington is 1,496 whales, while the minimum population size for eastern North Pacific Ocean blue whales is 1,050 whales (Carretta et al. 2020). There was a documented increase in the blue whale population size between 1979 and 1994, but there has not been evidence to suggest an increase in the population since then (Barlow 1994; Barlow and Taylor 2001; Carretta et al. 2010). In 2008, Cascadia Research conducted photographic identification surveys to make abundance estimates of blue whales along the U.S. West Coast. The results reflect an upward trend in abundance of blue whales along the U.S. West Coast, although their numbers are highly variable off California, most likely due to the variability of its use as a feeding area (Calambokidis et al. 2009b). Current estimates indicate the Eastern North Pacific stock shows no signs of population growth since the early 1990s, perhaps because the population is nearly at carrying capacity (Carretta et al. 2020). An overall population

growth rate for the species or growth rates for the two other individual U.S. stocks are not available at this time. In the Southern Hemisphere, population growth estimates are available only for Antarctic blue whales, which estimate a population growth rate of 8.2 percent per year (95 percent confidence interval 1.6 to 14.8 percent, Branch 2007).

The information available on the status and trend of blue whales do not allow us to reach any conclusions about the extinction risks facing blue whales as a species, or particular populations of blue whales. The possible exception is the eastern North Pacific blue whale population which many not have been subject to as much commercial whaling as other blue whale populations and which may be recovering to a stable population level since the cessation of commercial whaling in 1971 (Campbell et al. 2015; Monnahan et al. 2014a; Monnahan et al. 2014b). With the limited data available on blue whales, we do not know whether these whales exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself) or if blue whales are threatened more by exogenous threats such as anthropogenic activities (primarily whaling and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate).

Diving and Social Behavior

Blue whales spend more than 94 percent of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5 to 20 times at 12 to 20 sec intervals before a deep dive of 3 to 30 min (Croll et al. 1999a; Leatherwood et al. 1976; Maser et al. 1981; Yochem and Leatherwood 1985). Average foraging dives are 140 meters deep and last for 7.8 min (Croll et al. 2001a). Non-foraging dives are shallower and shorter, averaging 68 meters and 4.9 min (Croll et al. 2001a). However, dives of up to 300 meters are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 meters).

Blue whales occur singly or in groups of two or three (Aguayo 1974; Mackintosh 1965; Nemoto 1964; Pike and Macaskie 1969; Ruud 1956; Slijper 1962). However, larger foraging aggregations, even with other species such as fin whales, are regularly reported (Fiedler et al. 1998; Schoenherr 1991). Little is known of the mating behavior of blue whales.

Life History

The average life span of blue whales is 80 to 90 years. They have a gestation period of ten to 12 months, and calves nurse for six to seven months. Blue whales reach sexual maturity between five and 15 years of age with an average calving interval of two to three years. They winter at low latitudes, where they mate, calve and nurse, and summer at high latitudes, where they feed. Blue whales forage almost exclusively on krill and can eat approximately 3,600 kilograms (7,936.6 pounds) daily. Feeding aggregations are often found at the continental shelf edge, where upwelling produces concentrations of krill at depths of 90 to 120 meters (295.3 to 393.7 feet).

Vocalizations and Hearing

Blue whale vocalizations tend to be long (greater than 20 seconds), low frequency (less than 100 Hz) signals (Richardson et al. 1995b), with a range of 12 to 400 Hz and dominant energy in the infrasonic range of 12 to 25 Hz (Ketten 1998; McDonald et al. 2001; McDonald et al. 1995a; Mellinger and Clark 2003). Vocalizations are predominantly songs and calls.

Calls are short-duration sounds (two to five seconds) that are transient and frequency-modulated, having a higher frequency range and shorter duration than song units and often sweeping down in frequency (20 to 80 Hz), with seasonally variable occurrence. Blue whale calls have high acoustic energy, with reports of source levels ranging from 180 to 195 decibels re: 1 µPa at 1 meter (Aburto et al. 1997; Berchok et al. 2006; Clark and Gagnon 2004; Cummings and Thompson 1971b; Ketten 1998; McDonald et al. 2001; Samaran et al. 2010). Calling rates of blue whales tend to vary based on feeding behavior. For example, blue whales make seasonal migrations to areas of high productivity to feed, and vocalize less at the feeding grounds then during migration (Burtenshaw et al. 2004). Stafford and Moore (2005) recorded the highest calling rates when blue whale prey was closest to the surface during its vertical migration. Wiggins et al. (2005) reported the same trend of reduced vocalization during daytime foraging followed by an increase at dusk as prey moved up into the water column and dispersed. Oleson et al. (2007c) reported higher calling rates in shallow diving whales (less than 30 meters [98.4 feet]), while deeper diving whales (greater than 50 meters [164 feet]) were likely feeding and calling less.

Although general characteristics of blue whale calls are shared in distinct regions (McDonald et al. 2001; Mellinger and Clark 2003; Rankin et al. 2005; Thompson et al. 1996), some variability appears to exist among different geographic areas (Rivers 1997). Sounds in the North Atlantic Ocean have been confirmed to have different characteristics (i.e., frequency, duration, and repetition) than those recorded in other parts of the world (Berchok et al. 2006; Mellinger and Clark 2003; Samaran et al. 2010). Clear differences in call structure suggestive of separate populations for the western and eastern regions of the North Pacific Ocean have also been reported (Stafford et al. 2001); however, some overlap in calls from the geographically distinct regions have been observed, indicating that the whales may have the ability to mimic calls (Stafford and Moore 2005). In Southern California, blue whales produce three known call types: Type A, B, and D. B calls are stereotypic of blue whale population found in the eastern North Pacific (McDonald et al. 2006) and are produced exclusively by males and associated with mating behavior (Oleson et al. 2007a). These calls have long durations (20 seconds) and low frequencies (10 to 100 Hz); they are produced either as repetitive sequences (song) or as singular calls. The B call has a set of harmonic tonals, and may be paired with a pulsed Type A call. D calls are produced in highest numbers during the late spring and early summer and in diminished numbers during the fall, when A-B song dominates blue whale calling (Hildebrand et al. 2011; Hildebrand et al. 2012; Oleson et al. 2007c).

Blue whale songs consist of repetitively patterned vocalizations produced over time spans of minutes to hours or even days (Cummings and Thompson 1971b; McDonald et al. 2001). The songs are divided into pulsed/tonal units, which are continuous segments of sound, and phrases, repeated in combinations of one to five units (Mellinger and Clark 2003; Payne and McVay 1971). Songs can be detected for hundreds, and even thousands of kilometers (Stafford et al. 1998), and have only been attributed to males (McDonald et al. 2001; Oleson et al. 2007a). Worldwide, songs are showing a downward shift in frequency (McDonald et al. 2009). For example, a comparison of recording from November 2003 and November 1964 and 1965 reveals a long-term shift in the frequency of blue whale calling near San Nicolas Island. In 2003, the spectral energy peak was 16 Hz compared to approximately 22.5 Hz in 1964 and 1965, illustrating a more than 30 percent shift in call frequency over four decades (McDonald et al. 2006). McDonald et al. (2009) observed a 31 percent downward frequency shift in blue whale calls off the coast of California, and also noted lower frequencies in seven of the world's ten known blue whale songs originating in the Atlantic, Indian, Pacific, and Southern Oceans. Many possible explanations for the shifts exist but none has emerged as the probable cause.

As with other baleen whale vocalizations, blue whale vocalization function is unknown, although numerous hypotheses exist (maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources) (Edds-Walton 1997; Oleson et al. 2007b; Payne and Webb 1971; Thompson et al. 1992b). Intense bouts of long, patterned sounds are common from fall through spring in low latitudes, but these also occur less frequently while in summer high-latitude feeding areas. Short, rapid sequences of 30 to 90 Hz calls are associated with socialization and may be displays by males based upon call seasonality and structure. The low frequency sounds produced by blue whales can, in theory, travel long distances, and it is possible that such long distance communication occurs (Edds-Walton 1997; Payne and Webb 1971). The long-range sounds may also be used for echolocation in orientation or navigation (Tyack 1999).

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low frequency) and are likely most sensitive to this frequency range (Ketten 1997). Based on vocalizations and anatomy, blue whales are assumed to predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007c; Stafford and Moore 2005). In terms of functional hearing capability, blue whales belong to the low frequency group, which have a hearing range of 7 Hertz to 35 kilohertz (NMFS 2018b).

Critical Habitat

No critical habitat has been designated for the blue whale.

6.2.2 Fin Whale

The fin whale, *Balaenoptera physalus* (Linnæus 1758), is a well-defined, cosmopolitan species of baleen whale (Gambell 1985a) (Figure 34). Fin whales are the second-largest whale species by length. Fin whales are long-bodied and slender, with a prominent dorsal fin set about two-thirds of the way back on the body. The streamlined appearance can change during feeding when the pleated throat and chest area becomes distended by the influx of prey and seawater, giving the animal a tadpole-like appearance. The basic body color of the fin whale is dark gray dorsally and white ventrally, but the pigmentation pattern is complex. The lower jaw is gray or black on the left side and creamy white on the right side. This asymmetrical coloration extends to the baleen plates as well, and is reversed on the tongue. Individually distinctive features of pigmentation, along with dorsal fin shapes and body scars, have been used in photo-identification studies (Agler et al. 1990). Fin whales live 70 to 80 years (Kjeld 1982).

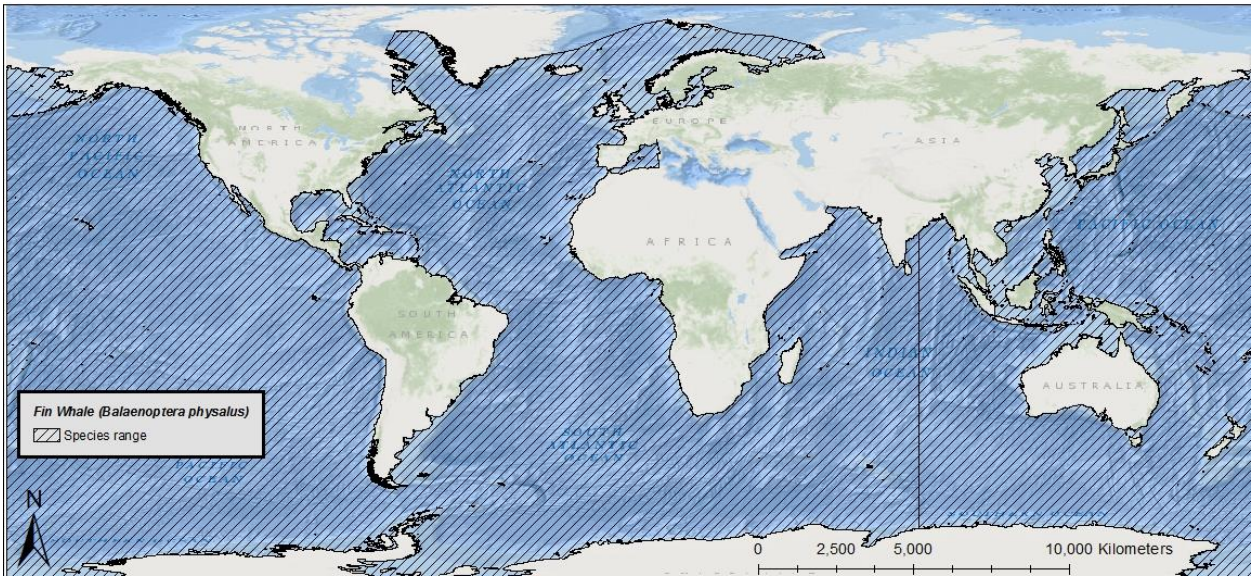


Figure 34. Map identifying the range of the endangered fin whale.

Distribution

Fin whales are distributed widely in every ocean except the Arctic Ocean. In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985a). The overall distribution may be based on prey availability. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Occurrence in the Offshore Portion of the NWTT action area

One fin whale was tagged out of Newport, Oregon in 2018 and was tracked for 35.9 days (Palacios et al. 2020). For the first 12 days, the whale remained off the central Oregon coast, mainly over shelf edge and slope waters. After a 10-day gap, the whale was tracked approximately 250 kilometers west of Queen Charlotte Sound off British Columbia over deep oceanic waters. The whale then headed southwest after 10 days and was last tracked approximately 90 kilometers west of the Hesquiat Peninsula on the central west coast of Vancouver Island (Palacios et al. 2020).

In 42 small boat surveys from Grays Harbor out to the 1,000-meter isobath off Quinault conducted over a five-year period in the summer between 2004 and 2009, there was one sighting of a group of three fin whales (Oleson and Hildebrand 2012). During aerial surveys conducted within the 2,000-meter isobath off southern Washington, Oregon, and Northern California in the spring, summer, and fall of 2011 and 2012, there were six sightings of 13 fin whales during winter and summer 2012 only in offshore waters over the continental slope (Adams et al. 2014b). Between 2014 and 2017, 32 fin whales were instrumented with satellite tags in the waters off the U.S. West Coast (Mate et al. 2017; U.S. Department of the Navy 2018a); all these whales are from the same stock as present in the NWTT action area. Only 4 of the 32 fin whales ventured into the NWTT action area. One of the four traveled only as far north as the California/Oregon border, and another, occurring in waters off Washington, only passed through the NWTT action area briefly on its way farther north into Canadian waters. Across the tag data sample years, fin whale use of the NWTT action area occurred primarily in late summer and fall (Mate et al. 2017; U.S. Department of the Navy 2018a). Consistent with sightings from systematic ship surveys out to 300 nautical miles off the U.S. West Coast and satellite tag data, habitat-based density models built with these data indicate that fin whales are more likely to be present seaward of the continental shelf in the offshore portion of the action area (Barlow 2016; Becker et al. 2016).

Acoustic monitoring has indicated a yearly seasonal pattern of fin whale calls in the action area off Washington and Canada with the absence of calls from approximately May through July (Debich et al. 2014; Kerosky et al. 2013; Oleson and Hildebrand 2012; Soule and Wilcock 2013; Trickey et al. 2015b; Wiggins et al. 2017b). Consistent with those findings and the satellite tag data, a seafloor seismic network at the Strait of Juan de Fuca was used to study fin whale calls and suggested northward movement of transiting fin whale groups from August to October and a southward movement from November to April (Soule and Wilcock 2013).

Occurrence in the Inland Waters of the NWTT action area

Fin whales are not expected to occur within the Inland Waters portion of the action area since fin whales have seldom been documented in the area. Lone fin whales were sighted in the Strait of Juan de Fuca between September and December 2015, in July 2016, and again in October 2017; these were three of only 10 total fin whale sightings in the Salish Sea since 1930 (Cogan 2015; Daugherty 2016; Nichol et al. 2018; Towers et al. 2018).

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Fin whales were observed seven times in the summer during surveys of the inland waters of southeast Alaska from 1991 to 2007 (Dahlheim et al. 2009). Given the limited number of sightings in inland waters such as Clarence Strait and Dixon Entrance (Dahlheim et al. 2009; Nichol et al. 2018) and their more pelagic nature, fin whales are considered rare in the SEAFAC region of the action area.

Population Structure

Archer et al. (2013) examined the genetic structure and diversity of fin whales globally. Full sequencing of the mitochondrial deoxyribonucleic acid (DNA) genome for 154 fin whales sampled in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere, resulted in 136 haplotypes, none of which were shared among ocean basins suggesting differentiation at least at this geographic scale. However, North Atlantic Ocean fin whales appear to be more closely related to the Southern Hemisphere population, as compared to fin whales in the North Pacific Ocean, which may indicate a revision of the subspecies delineations is warranted. Results of a later single-nucleotide polymorphism analysis indicate that distinct mitogenome matrilines in the North Pacific are interbreeding (Archer et al. 2019). Generally speaking, haplotype diversity was found to be high both within oceans basins, and across, with the greatest diversity found in North Pacific fin whales (Archer et al. 2019). Such high genetic diversity and lack of differentiation within ocean basins may indicate that despite some populations having small abundance estimates, the species may persist long-term and be somewhat protected from substantial environmental variance and catastrophes.

Abundance Estimate

The best current abundance estimate for fin whales in California, Oregon, and Washington waters out to 300 nautical miles is 9,029 (CV=0.12) (Nadeem et al. 2016); the minimum population estimate is 8,127 individuals (Carretta et al. 2020). There are currently no reliable abundance estimates for the Northeast Pacific stock (Muto et al. 2020).

Natural Threats

Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggested annual natural mortality rates might range from 0.04 to 0.06 for northeast Atlantic fin whales. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure and may be preventing some fin whale populations from recovering (Lambertsen 1983). Adult fin whales engage in flight responses (up to 40 km/h) to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Killer whale or shark attacks may also result in serious injury or death in very young and sick individuals (Perry et al. 1999).

Anthropogenic Threats

Fin whales have undergone significant exploitation, but are currently protected under the International Whaling Commission (IWC). Fin whales are still hunted in subsistence fisheries off West Greenland. In 2004, five males and six females were killed, and two other fin whales were struck and lost. In 2003, two males and four females were landed and two others were struck and lost (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. However, the scientific recommendation was to limit the number killed to four individuals until accurate populations could be produced (IWC 2005).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (Carretta et al. 2007; Carretta et al. 2017b; Douglas et al. 2008; Lien 1994; NMFS 2018a; Perkins and Beamish 1979; Saez 2018; Waring et al. 2007). Based on reports from 2007 to 2014 for waters off the U.S. West Coast, a total of four fin whales were seriously injured by entanglement in fishing gear(Carretta et al. 2018b). Between 1969 and 1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979). In 1999, one fin whale was reported killed in the Gulf of Alaska pollock trawl fishery and one was killed the same year in the offshore drift gillnet fishery (Angliss and Outlaw 2005; Carretta and Chivers. 2004). According to Waring et al. (2007), four fin whales in the western North Atlantic died or were seriously injured in fishing gear, while another five were killed or injured as a result of ship strikes between January 2000 and December 2004.

Available data from NMFS indicate that, in waters off the U.S. West Coast between 1991 and 2010, there were 11 reported ship strikes involving fin whales (National Marine Fisheries Service 2011), and from 2010 to 2014 along the U.S. West Coast there were nine reported ship strikes to fin whales (Carretta et al. 2016b). Since 2002, 10 out of the 12 stranded fin whales in Washington have showed evidence attributed to a large ship strike (Cascadia Research 2017a). Jensen and Silber (2004) review of the NMFS's ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26 percent of the recorded ship strikes [n = 75/292 records]), with most collisions occurring off the east coast, followed by the west coast of the U.S. and Alaska/Hawai'i. Between 1999 to 2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole et al. 2005; Nelson et al. 2007). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al. 2008). Similarly, 2.4 percent of living fin whales from the Mediterranean show ship strike injury and 16 percent of stranded individuals were killed by vessel collision (Panigada et al. 2006). There are also numerous reports of ship strikes off the Atlantic coasts of France and England (Jensen and Silber 2004). Management measures aimed at reducing the risk of ships hitting right whales should also reduce the risk of collisions with fin whales. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for

seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 27 percent in the Bay of Fundy region.

The organochlorines DDE, DDT, and PCBs have been identified from fin whale blubber, but levels are lower than in toothed whales due to the lower level in the food chain that fin whales feed at (Aguilar and Borrell 1988; Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983; Marsili and Focardi 1996). Females contained lower burdens than males, likely due to mobilization of contaminants during pregnancy and lactation (Aguilar and Borrell 1988; Gauthier et al. 1997). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males (Aguilar and Borrell 1988).

Climate change also presents a potential threat to fin whales, as habitat and prey availability will likely be affected. While fin whales have a larger feeding range than other species and may therefore not be affected as drastically as species with smaller feeding ranges, the potential impacts of climate change on fin whale recovery remain uncertain (NMFS 2010b). Climate change impacts on fin whales are of concern in the Mediterranean Sea, where fin whales appear to rely exclusively upon northern krill as a prey source. These krill occupy the southern extent of their range and increases in water temperature could result in their decline and that of fin whales in the Mediterranean Sea (Gambaiani et al. 2009).

Status and Trends

Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. The pre-exploitation estimate for the fin whale population in the North Pacific Ocean was 42,000 to 45,000. The North Pacific population of fin whales was reduced to 13,620 to 18,680 by 1973 (Ohsumi and Wada 1974). There are three stocks in United States Pacific Ocean waters: Northeast Pacific [minimum 2,554 individuals], Hawaii (approximately 154 individuals [N_{min}=75]) and California/Oregon/Washington (approximately 9,029 [N_{min}=8,127] individuals) (Carretta et al. 2020; Muto et al. 2020; Nadeem et al. 2016). According to whaling records from Canadian Pacific waters, at least 7,605 fin whales were killed between 1908 to 1967 (Gregr et al. 2000).

An overall fin whale population trend in the U.S. Pacific has not been established, but there is evidence that there has been increasing rates in the recent past in different parts of the region. From 1991 to 2014, the estimated average rate of increase for California, Oregon, and Washington waters was 7.5 percent, with the caveat that is unknown how much of that rate could be attributed to immigration rather than birth and death processes (Carretta 2019).

Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, fin whales appear to exist at population sizes that are large enough to avoid demographic phenomena that are known to increase the extinction probability of species

that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself). As a result, we assume that fin whales are likely to be threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) than endogenous threats caused by the small size of their population.

Nevertheless, based on the evidence available, the number of fin whales that are recorded to have been killed or injured in the past 20 years by human activities or natural phenomena, does not appear to be increasing the extinction probability of fin whales, although it may slow the rate at which they recover from population declines that were caused by commercial whaling.

Diving and Social Behavior

The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5 to 20 shallow dives, each of 13 to 20 s duration, followed by a deep dive of 1.5 to 15 min (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale's most common dives last 2 to 6 min (Hain et al. 1992; Watkins 1981b). The most recent data support average dives of 98 meters and 6.3 min for foraging fin whales, while non-foraging dives are 59 meters and 4.2 min (Croll et al. 2001a). However, Lafortuna et al. (1999) found that foraging fin whales have a higher blow rate than when traveling. Foraging dives in excess of 150 meters are known (Panigada et al. 1999). In waters off the U.S. Atlantic Coast, individuals or duos represented about 75 percent of sightings during the Cetacean and Turtle Assessment Program (Hain et al. 1992).

Individuals or groups of less than five individuals represented about 90 percent of the observations. Barlow (2003) reported mean group sizes of 1.1 to 4.0 during surveys off California, Oregon, and Washington.

Life History

Fin whales can live, on average, 80 to 90 years. They have a gestation period of less than one year, and calves nurse for six to seven months. Data from historical whaling records in Hecate Strait and Queen Charlotte Sound indicate that most births in the region occurred between mid-November and mid-March, with a peak in January (DFO 2017). Sexual maturity is reached between six and ten years of age with an average calving interval of two to three years. They mostly inhabit deep, offshore waters of all major oceans. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed, although some fin whales appear to be residential to certain areas. Acoustic recording data in British Columbia indicate that fin whales are present year-round (Koot 2015). Due to the detection of calling males from November through January, researchers assume that breeding occurs in Canadian Pacific waters in Hecate Strait and Queen Charlotte Sound during that time of year (DFO 2017). Fin whales eat pelagic crustaceans (mainly euphausiids or krill) and schooling fish such as capelin, herring, and

sand lice. There is a presumed feeding area along the Juan de Fuca Ridge off northern Washington, based on rates of fin whale calls in the area from fall through February (Muto et al. 2019; Soule and Wilcock 2013).

Vocalization and Hearing

Fin whales produce a variety of low-frequency sounds in the 10 Hz to 200 Hz range (Edds 1988; Thompson et al. 1992a; Watkins 1981a; Watkins et al. 1987). Typical vocalizations are long, patterned pulses of short duration (0.5 to 2 s) in the 18 Hz to 35 Hz range, but only males are known to produce these (Clark et al. 2002; Patterson and Hamilton 1964). Richardson et al. (1995b) reported the most common sound as a 1 second vocalization of about 20 Hz, occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns in winter. Au (Au and Green 2000) reported moans of 14 Hz to 118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34 Hz 150 Hz, and songs of 17 Hz to 25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981a). Source levels for fin whale vocalizations are 140 to 200 dB re 1µPa-m (see also Clark and Gagnon 2004; as compiled by Erbe 2002). The source depth of calling fin whales has been reported to be about 50 meters (Watkins et al. 1987).

Although their function is still in doubt, low-frequency fin whale vocalizations travel over long distances and may aid in long-distance communication (Edds-Walton 1997; Payne and Webb. 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999).

The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997) hypothesized that large mysticetes have acute infrasonic hearing. In a study using computer tomography scans of a calf fin whale skull, Cranford and Krysl (2015) found sensitivity to a broad range of frequencies between 10 hertz and 12 kilohertz and a maximum sensitivity to sounds in the 1 kilohertz to 2 kilohertz range.

Direct studies of fin whale hearing have not been conducted, but it is assumed that fin whales can hear the same frequencies that they produce (low) and are likely most sensitive to this frequency range (Ketten 1997; Richardson et al. 1995b).

Fin whales produce a variety of low frequency (< 1 kilohertz) sounds, but the most typically recorded is a 20 Hz pulse lasting about 1 second, and reaching source levels of 189 ± 4 dB re 1 µPam (Charif et al. 2002; Clark et al. 2002; Edds 1988; Richardson et al. 1995b; Sirovic et al. 2007; Watkins 1981a; Watkins et al. 1987). These pulses frequently occur in long sequenced

patterns, are down swept (e.g., 23 to 18 Hz), and can be repeated over the course of many hours (Watkins et al. 1987). In temperate waters, intense bouts of these patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high latitude feeding areas (Clarke and Charif 1998). The seasonality and stereotypic nature of these vocal sequences suggest that they are male reproductive displays (Watkins 1981a; Watkins et al. 1987); a notion further supported by recent data linking these vocalizations to male fin whales only (Croll et al. 2002). In Southern California, the 20 Hz pulses are the dominant fin whale call type associated both with call-counter-call between multiple animals and with singing (Navy 2010a; Navy 2012). An additional fin whale sound, the 40 Hz call described by Watkins (1981a), was also frequently recorded, although these calls are not as common as the 20 Hz fin whale pulses. Seasonality of the 40 Hz calls differed from the 20 Hz calls, since 40 Hz calls were more prominent in the spring, as observed at other sites across the northeast Pacific (Sirovic et al. 2012). Source levels of Eastern Pacific fin whale 20-Hz calls has been reported as 189 +/- 5.8 dB re 1uPa at 1m (Weirathmueller et al. 2013). Although acoustic recordings of fin whales from many diverse regions show close adherence to the typical 20 Hz bandwidth and sequencing when performing these vocalizations, there have been slight differences in the pulse patterns, indicative of some geographic variation (Thompson et al. 1992a; Watkins et al. 1987).

Responses to conspecific sounds have been demonstrated in a number of mysticetes, and there is no reason to believe that fin whales do not communicate similarly (Edds-Walton 1997). The low-frequency sounds produced by fin whales have the potential to travel over long distances, and it is possible that long-distance communication occurs in fin whales (Edds-Walton 1997; Payne and Webb. 1971). Also, there is speculation that the sounds may function for long range echolocation of large-scale geographic targets such as seamounts, which might be used for orientation and navigation (Tyack 1999).

Although no studies have directly measured the sound sensitivity of fin whales, experts assume that fin whales are able to receive sound signals in roughly the same frequencies as the signals they produce. This suggests fin whales, like other baleen whales, are more likely to have their best hearing capacities at low frequencies, including frequencies lower than those of normal human hearing, rather than at mid- to high-frequencies (Ketten 1997). Several fin whales were tagged during the Southern California-10 Behavioral Response Study (BRS) and no obvious responses to a mid-frequency sound source were detected by the visual observers or in the initial tag analysis (Southall et al. 2011). Results of studies on blue whales (Goldbogen et al. 2013) (Southall et al. 2011), which have similar auditory physiology compared to fin whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. In a study using computer tomography scans of a calf fin whale skull, Cranford and Krysl (2015) found sensitivity to a broad range of frequencies between 10 hertz and 12 kilohertz and a maximum sensitivity to sounds in the 1 to 2 kilohertz range. In terms of functional hearing capability, fin

whales belong to the low-frequency group, which have a hearing range of 7 Hz to 35 kilohertz (NMFS 2018b).

Critical Habitat

NMFS has not designated critical habitat for fin whales.

6.2.3 Humpback Whale – Central America and Mexico DPSs

The humpback whale is a widely distributed baleen whale found in all major oceans. Humpback whales are distinguishable from other whales by long pectoral fins and are typically dark grey with some areas of white. The humpback whale was originally listed as endangered on December 2, 1970 (35 FR 18319). Since then, NMFS has designated 14 DPSs with four identified as endangered (Cape Verde Islands/Northwest Africa, Western North Pacific, Central America, and Arabian Sea) and one as threatened (Mexico) (Figure 35).

Information available from the recovery plan (NMFS 1991), the recent SAR (Carretta 2019), the status review (Bettridge et al. 2015b), and the final listing were used to summarize the life history, population dynamics and status of the species as follows.

Distribution

Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985).

In the North Pacific Ocean, the summer range of humpback whales includes coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Tomlin 1967, Nemoto 1957, Johnson and Wolman 1984 as cited in NMFS 1991). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter.

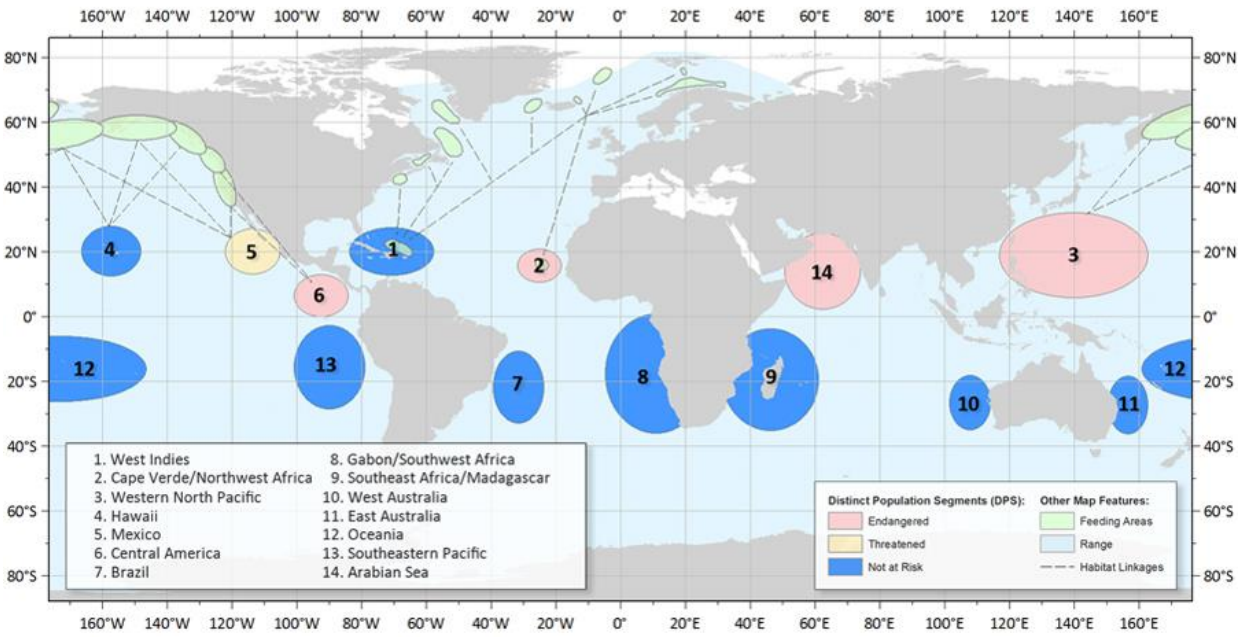


Figure 35. Map showing the distribution of the 14 humpback whale Distinct Population Segments (modified from Bettridge et al. 2015).

Occurrence in the Offshore Portion of the NWTT action area

Humpback whales are expected to be present in the offshore portion of the action area year round. The pattern of increasing humpback whale abundance indicated by previous investigations (Barlow et al. 2011; Calambokidis and Barlow 2004; Calambokidis and Barlow 2013; Calambokidis et al. 2017a) appears consistent with the highest-yet abundances of these species in 2014 (Barlow 2016). Acoustic monitoring over a number of years has demonstrated an overwintering presence of humpback whales and suggests that some portion of the humpback whale population off Washington remain in temperate waters during the winter (Debich et al. 2014; Emmons et al. 2017; Kerosky et al. 2013; Oleson and Hildebrand 2012; Širović et al. 2012; Trickey et al. 2015b). Satellite tag location data from humpback whales within the Offshore portion of the NWTT action area indicate a preference for shallow waters (less than 200 meters depth) consistent with generally known patterns of humpback whale distribution along the Pacific coast (Barlow et al. 2011; Becker et al. 2017; Campbell et al. 2015; Ford et al. 2010a; Forney and Barlow 1998; Mate et al. 2017; U.S. Department of the Navy 2018a).

In 2018, 20 humpback whales were tagged off the coast of Washington, all of which spent time in the NWTT action area during the months of August, September, and October. Humpback whale densities were greatest near Swiftsure Bank, approximately 25 kilometers northwest of Cape Flattery. Four whales traveled as far east as Port Angeles, two traveled as far as Victoria, Vancouver Island, and one traveled into the Strait of Georgia. Humpback whale distances to shore in NWTT averaged 27 kilometers (Palacios et al. 2020). One male humpback whale

frequented foraging areas around Swiftsure Bank and the offshore canyons to the northwest corner of the Olympic Peninsula in 2018 and 2019 (Mate et al. 2020).

In 2018, five humpback whales were tagged off the coast of Oregon, all of which spent time in the NWTT action area during the months of September, October, and November. Humpback whale densities were greatest over the continental shelf edge west of Stonewall Bank, between Newport and Waldport on the central Oregon coast, with a secondary concentration along the shelf edge south of Heceta Bank, between Waldport and Coos Bay. Humpback whale distances to shore in NWTT averaged 39 kilometers (Palacios et al. 2020).

(Palacios et al. 2020) reported that humpback whales exhibited similar diving behavior off Washington and Oregon, with dives lasting from two to seven minutes. Whales off the coast of Washington generally dove to depths less than 100 meters. These dives were deeper and longer in duration during the day, suggesting that the whales were feeding on krill throughout the area. Tagged whales spent around 53 percent of their time at depths less than or equal to 30 meters. Humpback whales were found to spend most of their time foraging in shallower continental shelf waters. These whales were found to spend a smaller proportion of their time transiting through deeper waters beyond the shelf break (Palacios et al. 2020).

Historically, one humpback whale tagged in the waters north of Monterey California was tracked for 85 days moving more than 900 kilometers to waters offshore of Pacific City, Oregon (U.S. Department of the Navy 2018a). While heading north, this individual took an offshore route as far as 200 kilometers from shore and then returned south along a more inshore route. This whale and two others (one tagged off of Newport, Oregon, and the other off Astoria, Oregon) spent portions of time in nearshore shallow waters (less than 200 meters in depth) or in Canadian waters, during which they were outside of the NWTT action area and the locations where Navy training and testing activities occur (Mate et al. 2017; U.S. Department of the Navy 2018a). The remaining two of the five tracked humpback whales were tagged near Cape Blanco, in southern Oregon, and spent most of their time beyond the NWTT action area in continental shelf waters off Trinidad Head and Eureka, California (U.S. Department of the Navy 2017b).

Occurrence in the Inland Waters of the NWTT action area

Data indicate that an increasing number of humpback whales are seasonally present in the Inland Waters portion of the action area and that this trend escalated in 2014 (Cascadia Research 2017b). Locations for 22 humpback whales tagged out of Neah Bay, Washington in 2019 ranged from the northwest corner of Vancouver Island, British Columbia, Canada to Seaside, Oregon (Mate et al. 2020). Most locations occurred between Clallam Bay, 30 kilometers east of Neah Bay, and approximately 100 kilometers west-southwest of Cape Flattery (Mate et al. 2020). Based on opportunistic and informal sighting reports in 2015, it was estimated that there were as many as 15–25 whales present in the Inland Waters portion of the action area during any given day (Cogan 2015).

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Humpback whales are assumed to be present in Behm Canal (U.S. Department of the Navy 1991). In summer, relatively high densities of humpback whales occur throughout much of Southeast Alaska (Muto et al. 2017; Muto et al. 2018b) and Northern British Columbia (Ashe et al. 2013; Best et al. 2015; Ford et al. 2010a; Keen et al. 2018), and they were observed frequently during spring through fall in a series of surveys from 1991 to 2007 in Southeast Alaska (Dahlheim et al. 2009). Humpback whales tagged off Maui were tracked to feeding grounds off British Columbia, Canada and Baranof Island in southeast Alaska (Mate et al. 2019), although these whales were likely from the unlisted Hawaii DPS. Although surveys have not been conducted in the winter months in Southeast Alaska, humpback whales are seen intermittently throughout the winter in various areas of Southeast Alaska, including Sitka Sound and Lynn Canal, indicating that some of these animals do not migrate south and remain in Southeast Alaskan waters to feed on herring (Moran et al. 2009).

Population Structure

During winter months in northern or southern hemispheres, adult humpback whales migrate to specific areas in warmer, tropical waters to reproduce and give birth to calves. During summer months, humpback whales migrate to specific areas in northern temperate or sub-arctic waters to forage. In summer months, humpback whales from different "reproductive areas" will congregate to feed; in the winter months, whales will migrate from different foraging areas to a single wintering area. In either case, humpback whales appear to form "open" populations; that is, populations that are connected through the movement of individual animals.

Separate feeding groups of humpback whales are thought to inhabit western U.S. and Canadian waters, with the boundary between them located roughly at the U.S./Canadian border. The southern feeding ground ranges between 32° to 48°N, with limited interchange with areas north of Washington State (Calambokidis et al. 2004b; Calambokidis et al. 1996). Humpback whales feed along the coasts of Oregon and Washington from May-November, with peak numbers reported May-September, when they are the most commonly reported large cetacean in the region (Calambokidis and Chandler. 2000; Calambokidis et al. 2004b; Dohl 1983; Green et al. 1992). Off Washington State, humpback whales concentrate between Juan de Fuca Canyon and the outer edge of the shelf break in a region called "the Prairie," near Barkley and Nitnat canyons, in the Blanco upwelling zone, and near Swiftsure Bank (Calambokidis et al. 2004b). Humpback whales also tend to congregate near Heceta Bank off the coast of Oregon (Green et al. 1992). Additional data suggest that further subdivisions in feeding groups may exist, with up to six feeding groups present between Kamchatka and southern California (Witteveen et al. 2009).

The Central America DPS is composed of humpback whales that breed along the Pacific coast of Costa Rica, Panama, Guatemala, El Salvador, Honduras, and Nicaragua. This DPS feeds almost

exclusively offshore of California and Oregon in the eastern Pacific Ocean, with only a few individuals identified at the northern Washington – southern British Columbia feeding grounds.

The Mexico DPS is composed of humpback whales that breed along the Pacific coast of mainland Mexico, and the Revillagigedos Islands, and transit through the Baja California Peninsula coast. This DPS feeds across a broad geographic range from California to the Aleutian Islands, with concentrations in California-Oregon, northern Washington-southern British Columbia, northern and western Gulf of Alaska, and Bering Sea feeding grounds (81 FR 62259).

Abundance Estimate

For humpback whales, DPSs that have a total population size of 2,000 to 2,500 individuals or greater provide for maintenance of genetic diversity resulting in long-term persistence and protection from substantial environmental variance and catastrophes. Distinct population segments that have a total population of 500 individuals or less may be at a greater risk of extinction due to genetic risks resulting from inbreeding. Population at low densities (less than one hundred) are more likely to suffer from the 'Allee" effect, where inbreeding and the heightened difficulty of finding mates reduces the population growth rate in proportion with reducing density. Based on surveys from 2004 to 2006, the Central America DPS is estimated to have just below 800 individuals while the Mexico DPS is estimated to have just below 3,000 individuals (Wade 2017). However, sightings of humpbacks off the U.S. west coast have been increasing in more recent years, and these DPS numbers are likely underestimates (Calambokidis 2017).

Natural Threats

Natural sources and rates of mortality of humpback whales are not well known. Based upon prevalence of tooth marks, attacks by killer whales appear to be highest among humpback whales migrating between Mexico and California, although populations throughout the Pacific Ocean appear to be targeted to some degree (Steiger et al. 2008). Juveniles appear to be the primary age group targeted. Humpback whales engage in grouping behavior, flailing tails, and rolling extensively to fight off attacks. Calves remain protected near mothers or within a group and lone calves have been known to be protected by presumably unrelated adults when confronted with attack (Ford and Reeves 2008).

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992b). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period.

Anthropogenic Threats

Three human activities are known to threaten humpback whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for listing several species as endangered.

Entanglement in pot/trap fisheries has been the most common source of injury to humpback whales along the U.S. Pacific coast (Carretta et al. 2016a; Carretta et al. 2017b; NOAA 2017; Saez et al. 2012). There were 54 separate entanglement cases reported for humpback whales along the U.S. West Coast in 2016 (National Oceanic and Atmospheric Administration 2017). For the five-year period between 2011 and 2015 there were 34 cases of entanglement involving pot/trap fisheries and an additional 26 cases of reported interactions with other fisheries (Carretta et al. 2017c). Humpback whales from Mexico and Central America have been identified feeding in Alaska (Bettridge et al. 2015a; Calambokidis et al. 2008). Humpback whales have also been reported seriously injured and killed from entanglement in fishing gear while in their Alaskan feeding grounds (Helker et al. 2017); some proportion of these entanglements could be to whales from the Mexico Distinct Population Segment and from the Central America Distinct Population Segment. An overall minimum estimate of mortality and serious injury due to fisheries in Alaska is 14 humpback whales annually (Muto et al. 2017).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997). Along the U.S. Pacific coast between 2011 and 2015, there were nine ship strikes involving humpback whales; none were Navy vessels (Carretta et al. 2017a; Carretta et al. 2016b). The mean vessel collision mortality and serious injury rate in Alaska is 4.3 humpback whales annually (Muto et al. 2017).

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (Gauthier et al. 1997). Higher PCB levels have been observed in Atlantic waters versus Pacific waters along the United States and levels tend to increase with individual age (Elfes et al. 2010). Although humpback whales in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, overall levels are on par with other baleen whales, which are generally lower than odontocete cetaceans (Elfes et al. 2010). As with blue whales, these contaminants are transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe et al. 2004). Contaminant levels are relatively high in humpback whales as compared to blue whales. Humpback whales feed higher on the food chain, where prey carry higher contaminant loads than the krill that blue whales feed on.

Status and Trends

Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. On April 21, 2015, NMFS proposed to divide the globally listed

endangered humpback whale population into 14 DPSs (80 FR 22304). The humpback whales in the action area potentially belong to one of two listed DPSs: the threatened Mexico DPS or the endangered Central America DPS. All two of these DPSs may feed seasonally in the action area.

According to historical whaling records from five whaling stations in British Columbia, 5,638 humpback whales were killed between 1908 and 1967 (Gregr et al. 2000). We have no way of knowing the degree to which a specific DPS of humpback whale was affected by historical whaling. However, it is likely that individuals from both the Mexico and Central America DPSs were taken, based on where the whalers were hunting off British Columbia (i.e., the purported feeding grounds for these population segments). Prior to commercial whaling, hundreds of thousands of humpback whales existed. Global abundance declined to the low thousands by 1968, the last year of substantial catches (IUCN 2012). Humpback whales may be killed under "aboriginal subsistence whaling" and "scientific permit whaling" provisions of the International Whaling Commission. Additional threats include ship strikes, fisheries interactions (including entanglement), energy development, harassment from whaling watching noise, harmful algal blooms, disease, parasites, and climate change. Due to on-going threats, and the purported low population size, the Central America DPS still faces a risk of extinction. The Mexico DPS has a comparatively larger population than the Central America DPS, but still faces a risk of becoming endangered within the foreseeable future throughout all or a significant portion of its range.

Diving and Social Behavior

Maximum diving depths are approximately 170 m, with a very deep dive (240 meters) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1 to 5.1 min in the north Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). Because most humpback prey is likely found within 300 meters of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen et al. 2008).

During the feeding season, humpback whales form small groups that occasionally aggregate on concentrations of food that may be stable for long-periods of times. Humpbacks use a wide variety of behaviors to feed on various small, schooling prey including krill and fish (Hain et al. 1982; Hain et al. 1995; Jurasz and Jurasz 1979; Weinrich et al. 1992). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994; Clapham 1996; Tyack 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995).

Humpback whales feed on pelagic schooling euphausiids and small fish including capelin, herring and mackerel. Like other large mysticetes, they are a "lunge feeder" taking advantage of dense prey patches and engulfing as much food as possible in a single gulp. They also blow nets, or curtains, of bubbles around or below prey patches to concentrate the prey in one area, then lunge with open mouths through the middle. Dives appear to be closely correlated with the depths of prey patches, which vary from location to location. In the north Pacific (southeast Alaska), most dives were of fairly short duration (<4 min) with the deepest dive to 148 meters (Dolphin 1987), while whales observed feeding on Stellwagen Bank in the North Atlantic dove to <40 meters (Hain et al. 1995). Hamilton et al. (1997) tracked one possibly feeding whale near Bermuda to 240 meters depth.

Life History

Humpback whales can live, on average, 50 years. They have a gestation period of 11 to 12 months, and calves nurse for one year. Sexual maturity is reached between five to 11 years of age. Every one to five years, females five birth to a single calf, with an average calving interval of two to three years. Humpback whales mostly inhabit coastal and continental shelf waters. They winter at lower latitudes, where they calve and nurse, and summer at high latitudes, where they feed. In British Columbia, the highest numbers of humpback whales are found between May and October, however, individuals are observed throughout the year (Ford 2009). Humpback whales exhibit a wide range of foraging behaviors and feed on a range of prey types, including: small schooling fishes, euphausiids, and other large zooplankton (Bettridge et al. 2015b).

Vocalization and Hearing

Humpback whale vocalization is much better understood than hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop et al. 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kilohertz with estimated source levels from 144 to 174 dB (Au et al. 2006a; Frazer and Mercado Iii 2000; McCauley et al. 2000b; Richardson et al. 1995b; Winn et al. 1970). Males also produce sounds associated with aggression, which are generally characterized by frequencies between 50 Hz to 10 kilohertz with most energy below 3 kilohertz (Silber 1986; Tyack 1983). Such sounds can be heard up to 9 km (4.9 nm) away (Tyack 1983). Other social sounds from 50 hertz to 10 kilohertz (most energy below 3 kilohertz) are also produced in breeding areas (Richardson et al. 1995b; Tyack 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kilohertz), pulses (25 to 89 hertz) and songs (ranging from 30 Hz to 8 kilohertz but dominant frequencies of 120 Hz to 4 kilohertz), which can be very loud (175 to 192 dB re: 1 µPa at 1 meter) (Edds-Walton 1997; Oleson et al. 2007b; Payne and Webb 1971; Thompson et al. 1992a). However, humpback whales tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995b).

Humpback whales are known to produce three classes of vocalizations: (1) "songs" in the late fall, winter, and spring by solitary males; (2) social sounds made by calves (Zoidis et al. 2008) or

within groups on the wintering (calving) grounds; and (3) social sounds made on the feeding grounds (Richardson et al. 1995b). The best-known types of sounds produced by humpback whales are songs, which are thought to be reproductive displays used on breeding grounds and sung only by adult males (Clark and Clapham 2004; Gabriele and Frankel. 2002; Helweg et al. 1992; Schevill et al. 1964; Smith et al. 2008). Males may also use songs as a way of mutually assisting other males in mating (Darling et al. 2006) and/or as a long-range sonar to detect other whales from a distance (Mercado III 2018). Singing is most common on breeding grounds during the winter and spring months, but is occasionally heard in other regions and seasons (Clark and Clapham 2004; Gabriele and Frankel. 2002; McSweeney et al. 1989). Au et al. (2006b) noted that humpback whales off Hawaii tended to sing louder at night compared to the day. There is a geographical variation in humpback whale song, with different populations singing a basic form of a song that is unique to their own group. However, the song evolves over the course of a breeding season but remains nearly unchanged from the end of one season to the start of the next (Payne et al. 1983). The song is an elaborate series of patterned vocalizations that are hierarchical in nature, with a series of songs ('song sessions') sometimes lasting for hours (Payne and McVay 1971). Components of the song range from below 20 Hz up to 4 kilohertz, with source levels measured between 151 and 189 dB re: 1 µPa-m and high frequency harmonics extending beyond 24 kilohertz (Au et al. 2006b; Winn et al. 1970). Perazio and Mercado III (2018) found that frequencies from humpback whale songs in the Gulf of Tribugá in the Pacific ranged from 10 Hz to over 10,000 Hz but a frequency band of around 250 to 425 Hz was produced the most often. This suggests that singing humpback whales in this region may prefer to utilize this frequency band. Social calls range from 20 Hz to 10 kilohertz, with dominant frequencies below 3 kilohertz (D'Vincent et al. 1985; Dunlop et al. 2008; Silber 1986; Simao and Moreira 2005). Female vocalizations appear to be simple; Simao and Moreira (2005) noted little complexity.

Humpback whale calves have been shown to produce calls with durations of around 200 to 250 milliseconds, mean bandwidths of around 621 to 2004 hertz, and mean center frequencies of around 500 to 600 hertz (Zoidis et al. 2008). While the significance of these calls are unknown, they may serve as contact calls to the calf's mother (Indeck et al. 2020; Zoidis et al. 2008). Humpback whale calves are likely restricted by their physical immaturity in the types of social calls they can produce; it is thought that their vocal repertoire expands with age, as in North Atlantic right whales (Indeck et al. 2020).

"Feeding" calls, unlike song and social sounds are a highly stereotyped series of narrow-band trumpeting calls. These calls are 20 Hz to 2 kilohertz, less than one second in duration, and have source levels of 162 to 192 dB re: 1 µPa-m (D'Vincent et al. 1985; Thompson et al. 1986). The fundamental frequency of feeding calls is approximately 500 Hz (D'Vincent et al. 1985; Thompson et al. 1986). The acoustics and dive profiles associated with humpback whale feeding behavior in the northwest Atlantic Ocean has been documented with digital acoustic recording tags (DTAGs) (Stimpert et al. 2007). Underwater lunge behavior was associated with nocturnal

feeding at depth and with multiple boats of broadband click trains that were acoustically different from toothed whale echolocation: (Stimpert et al. 2007) termed these sounds "mega-clicks" which showed relatively low received levels at the DTAGs (143 to 154 dB re: 1 µPa), with the majority of acoustic energy below 2 kilohertz.

Recalde-Salas et al. (2020) recorded non-song vocalizations from humpback whales off Western Australia. The frequencies of these sounds ranged from 9 hertz to 6 kilohertz, the majority being under 200 hertz. These sounds lasted from 0.09 to 3.59 seconds. Some of these vocalizations appeared to be similar to social sounds or feeding calls reported in Alaska.

In terms of functional hearing capability, humpback whales belong to low frequency cetaceans which have a hearing range of 7 Hz to 35 kilohertz (NMFS 2018b). Humpback whale audiograms using a mathematical model based on the internal structure of the ear estimate sensitivity is from 700 Hz to 10 kilohertz, with maximum relative sensitivity between 2 kilohertz and 6 kilohertz (Ketten and Mountain 2014). Research by Au et al. (2001) and Au et al. (2006b) off Hawaii indicated the presence of high frequency harmonics in vocalizations up to and beyond 24 kilohertz. While recognizing this was the upper limit of the recording equipment, it does not demonstrate that humpback whales can actually hear those harmonics, which may simply be correlated harmonics of the frequency fundamental in the humpback whale song. The ability of humpback whales to hear frequencies around 3 kilohertz may have been demonstrated in a playback study. Maybaum (1990) reported that humpback whales showed a mild response to a handheld sonar marine mammal detection and location device with frequency of 3.3 kilohertz at 219 dB re: 1 µPa-m or frequency sweep of 3.1 to 3.6 kilohertz. In addition, the system had some low frequency components (below 1 kilohertz) which may have been an artifact of the acoustic equipment. This possible artifact may have affected the response of the whales to both the control and sonar playback conditions.

Critical Habitat

On October 9, 2019, NMFS proposed to designate critical habitat for Central America and Mexico DPS humpback whales (84 FR 54354). Critical habitat for the Central America DPS would include 48,459 square nautical miles of marine habitat off the coasts of Washington, Oregon, and

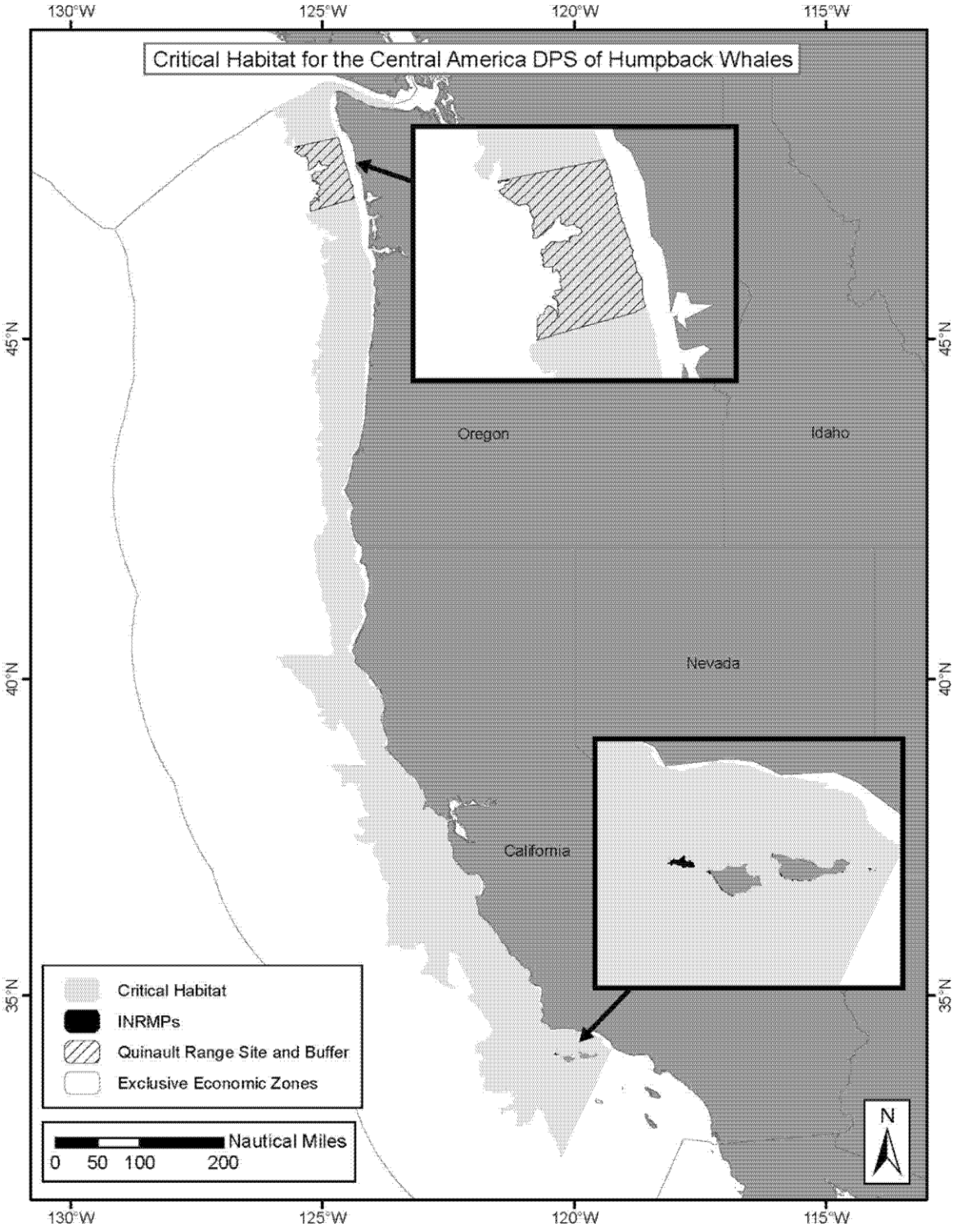


Figure 36. Proposed critical habitat for the humpback whale Central America Distinct Population Segment.

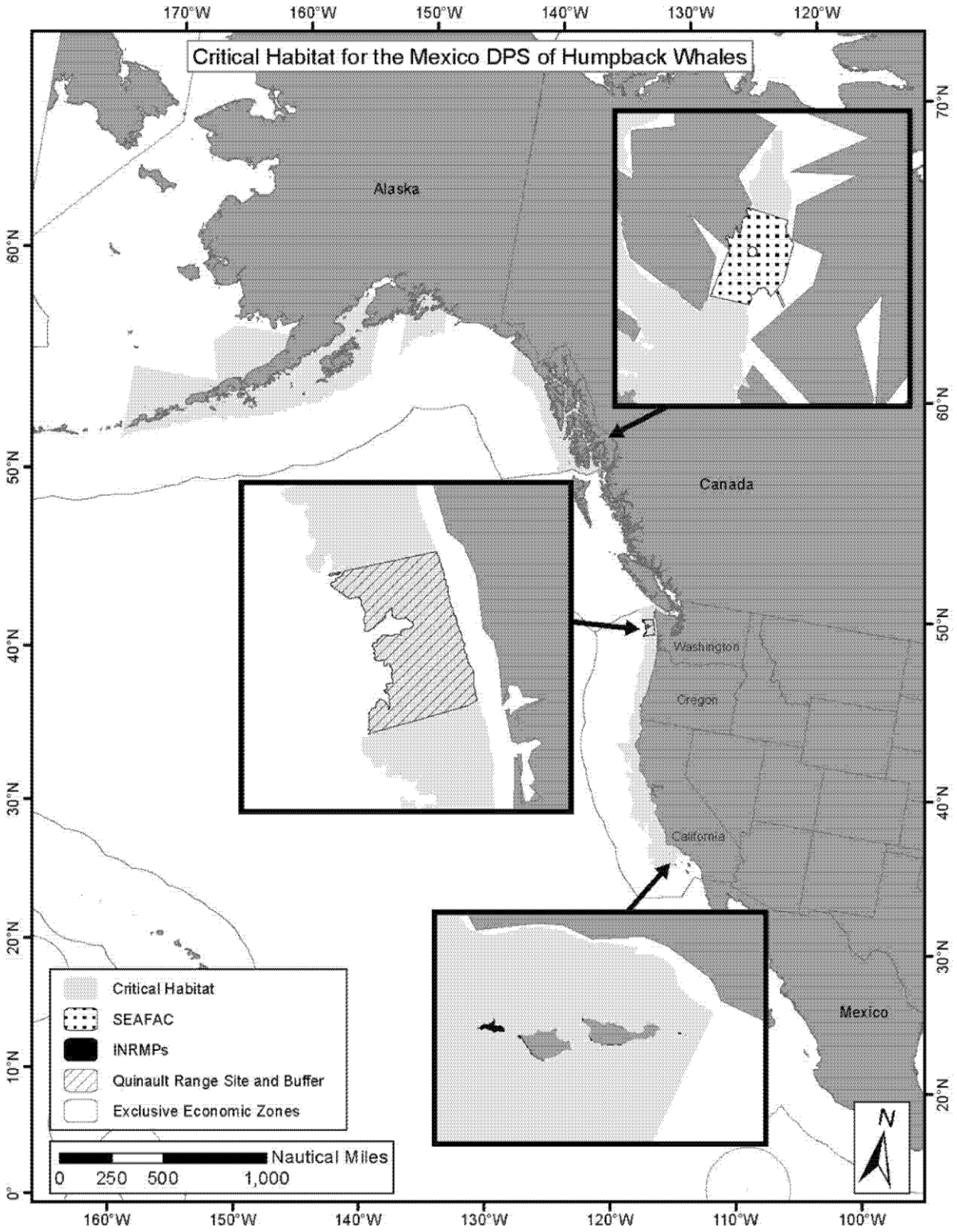


Figure 37. Proposed critical habitat for the humpback whale Mexico Distinct Population Segment.

California (Figure 36). Critical habitat for the Mexico DPS would include 175,812 nm^2 of marine habitat off the coasts of Alaska, Washington, Oregon, and California (Figure 37). About 1,570 nm^2 of marine habitat are being proposed for exclusion from the critical habitat designation as a result of potential national security impacts. These areas include the Quinault Range Site and Buffer and SEAFAC. These critical habitat designations include the physical and biological feature of prey species, defined as "primarily euphausiids and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth" (84 FR 54354).

6.2.4 Killer Whale – Southern Resident DPS

Killer whales are distributed worldwide, but populations are isolated by region and ecotype. Killer whales have been divided into distinct population segments on the basis of differences in genetics, ecology, morphology and behavior. They also have a conspicuous white patch above and behind the eye and a highly variable gray or white saddle behind the dorsal fin. The Southern Resident killer whales DPS of killer whale can be found along the Pacific Coast of the United States and Canada, and in the Salish Sea, Strait of Juan de Fuca, and Puget Sound (Figure 38). The Southern Resident killer whale DPS was listed as endangered under the ESA) on November 18, 2005 (70 FR 69903).

Life History

Killer whales show considerable sexual size "dimorphism". Adult males develop larger pectoral flippers, dorsal fins, tail flukes, and girths than females. Male adult killer whales can reach up to 32 feet (9.8 meters) in length and can weigh nearly 22,000 pounds (10,000 kg); females can reach 28 feet (8.5 meters) in length and can weigh up to 16,500 pounds (7,500 kg). Males become sexually mature at ten to 17 years of age. Females reach maturity at 12 to 16 years of age and produce an average of 5.4 surviving calves during a reproductive life span of approximately 25 years. Mothers and offspring maintain highly stable, life-long social bonds, and this natal relationship is the basis for a matrilineal social structure. Life expectancy for wild female killer whales is approximately 50 years, with maximum longevity estimated at 80 to 90 years. Male killer whales typically live for about 30 years, with maximum longevity estimated at 50 to 60 years.

Recent studies have utilized movement variables of subsurface behavior to identify prey capture events in order to explore sex differences in foraging ecology of fish-eating killer whales. Results show that males made more prey capture dives than females, presumably to support the energetic requirements of a larger body size (Tennessen et al. 2019a; Tennessen et al. 2019b). Females were more likely than males to engage in non-foraging behavior, and spent notably more time in surface respiration, intermediate depth or traveling states than males, who spent substantially more time searching for and pursuing prey. Although limited by small sample sizes, these studies suggest that female foraging behavior may be temporally compartmentalized, due

to the time demands of caring for calves or other offspring or kin, and due to the potentially increased costs of transport for females with calves.

The Southern Resident killer whales DPS includes three large, stable pods or familial groups (J, K, and L), which occasionally interact (Parsons et al. 2009). Most mating was thought to occur outside natal pods, during temporary associations of pods, or as a result of the temporary dispersal of males (Pilot et al. 2010). However, recent genetic paternity analyses using single nucleotide polymorphisms and microsatellites indicate that mating within Southern Resident killer whale pods is common and inbreeding is occurring in the population (Ford et al. 2018). Four cases of strong inbreeding were detected (two between parent and offspring, one between paternal half-siblings, and one between an uncle and half-niece), and two males (J1 and L41) were inferred to have sired 52 percent of all sampled progeny born since 1990 (Ford et al. 2011; Ford et al. 2018).

Southern Resident killer whale habitat utilization is dynamic, and specific breeding, calving or resting areas are not currently documented (NMFS 2006a). Southern Resident killer whales are highly mobile and can travel up to 160 km in a 24-hour period (Baird and Whitehead 2000), allowing rapid movements between areas. The DPS requires open waterways that are free from obstruction to move between important habitat areas, find prey and fulfill other life history requirements (NMFS 2006a). Individual knowledge of productive feeding areas and other special habitats is probably an important determinant in the selection of locations visited and is likely a learned tradition passed from one generation to the next (Ford et al. 1998).

Southern Resident killer whales are large mammals requiring abundant food sources to sustain metabolic processes throughout the year (NMFS 2019e). Prey availability changes seasonally, and Southern Resident killer whales appear to depend on different prey species/populations and habitats throughout the year. The seasonal timing of salmon returns to different river systems likely influences their movements (NMFS 2019e). Whales may travel significant distances to locate prey aggregations sufficient to support their numbers.

The daily prey energy requirements for individual females and males range from 41,376 to 269,458 kcal/day and 41,376 to 217,775 kcal/day, respectively (Noren 2011). The daily prey energy requirements can be converted to the number of fish required each year if the caloric densities of the fish (kcal/fish) consumed are known. However, caloric density of fish can vary because of multiple factors including differences in species, age and/or size, percent lipid content, geographic region and season (L. Barre, NMFS, personal communication to R. Salz, NMFS, May 27, 2020). Noren (2011) estimated the daily consumption rate of a population with 82 individuals over the age of one that consumes solely Chinook salmon would consume 289,131–347,000 fish/year by assuming the caloric density of Chinook salmon was 16,386 kcal/fish (i.e., the average value for adults from Fraser River). Williams et al. (2011b) and Chasco et al. (2017a) modeled annual SRKW prey requirements and found that the whole population requires approximately 211,000 to 364,100 and 190,000 to 260,000 Chinook salmon

per year, respectively. These estimates provide a general indication of how many Chinook salmon need to be available and consumed to meet the biological needs of the whales. These estimates can vary based on several underlying assumptions including the size of the whale population and the caloric density of the salmon.

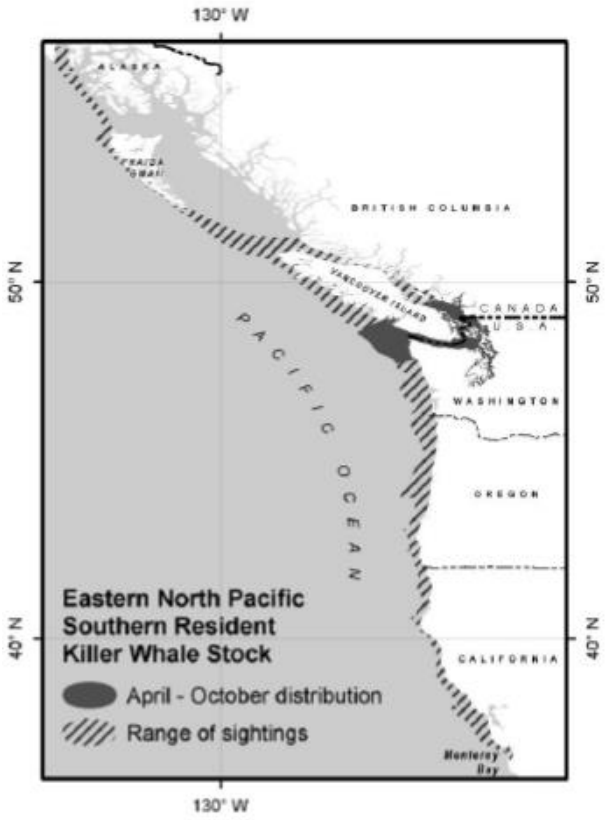


Figure 38. Map identifying the range of the endangered Southern Resident killer whales distinct population segment of killer whale. Approximate April through October distribution of the Southern Resident killer whales distinct population segment of killer whale (shaded area) and range of sightings (diagonal lines) (Carretta et al. 2016c).

Diet data indicate that Chinook salmon is the primary prey of Southern Resident killer whales despite the much lower abundance of this species in some areas and during certain time periods compared to other salmonids (Ford et al. 2016; Hanson et al. 2010; NMFS 2019e). Fecal DNA analysis indicate that greater than 98 percent of Southern Resident killer whale diet in Inland Waters is made up of salmonids, with Chinook salmon comprising around 80 percent of the overall diet (Ford et al. 2016). This confirms previous studies that used visual observations of foraging events and collection of prey remains to identify prey items (Hanson et al. 2010). Fecal DNA analysis also found that coho salmon make up as much as 15 percent of the diet, with

increased consumption of coho salmon (more than 40 percent of the diet) in late summer during seasonal downward shifts in Chinook salmon abundance.

Factors that might influence this preference include Chinook salmon's large size, high fat and energy content, and year-round occurrence in the whales' geographic range (Ford et al. 2010b). Chinook salmon have the highest value of total energy content compared to other salmonids because of their larger body size and higher energy density (kcal/kg) (O'Neill et al. 2014). Southern Resident killer whales consume greater proportions of the larger (older) fish (particularly the 4- and 5-year-olds) relative to the overall numbers of Chinook salmon present (Hilborn et al. 2012).

Genetic analysis of the Hanson et al. (2010) samples indicate that when Southern Resident killer whales are in Inland Waters from May to September, they consume Chinook salmon stocks that originate from regions including the Fraser River (including Upper Fraser, Mid Fraser, Lower Fraser, North Thompson, South Thompson and Lower Thompson), Puget Sound (North and South Puget Sound), the Central British Columbia Coast and West and East Vancouver Island.

Data collected during the winter months and in coastal areas provide further support for the importance of Chinook salmon in most seasons and outer coastal areas (Ford et al. 2010b; Hanson In prep). Results of 56 prey samples indicate that, as is the case in Inland Waters, Chinook salmon are the primary species consumed on the outer coast, although steelhead, chum salmon, halibut, and ling cod were also consumed (Hanson In prep) . Most of the Chinook salmon prey samples collected on the outer coast were determined to have originated from the Columbia River basin, including Lower Columbia Springs, Middle Columbia Tule, and Upper Columbia Summer/Fall. However, the Chinook salmon stocks included fish from as far north at the Taku River and as far south as the Central Valley California (Hanson In prep) .

Vocalizations and Hearing

Killer whales have advanced vocal communication and also use vocalizations to aid in navigation and foraging (NMFS 2008e). Their vocalizations typically have both a low frequency component (250 Hz to 1.5 kilohertz) and a high frequency component (5 to 12 kilohertz) (NMFS 2008e). Killer whale vocalizations consist of three main types, echolocation clicks, which are primarily used for navigation and foraging, and tonal whistles and pulse calls, which are thought to be used for communication (NMFS 2008e).

Their clicks are relatively broadband, short (0.1 to 25 milliseconds (msec)), and range in frequency from 8 to 80 kilohertz with an average center frequency of 50 kilohertz and an average bandwidth of 40 kilohertz (Au et al. 2004). Killer whales apparently use these signals to sense objects in their environment, such as prey; whales foraging on salmon produce these signals at peak-to-peak source levels ranging from 195 to 225 dB re: 1 µPa-m (Au et al. 2004).

Resident killer whales off British Columbia produce whistles for long-range communication during foraging and slow traveling, and social interactions with the clan and between different

groups (Riesch et al. 2006; Thomsen et al. 2002). These whistles are tonal signals that have longer duration (0.06 to 18 sec) and frequencies ranging from 0.5 to 10.2 kilohertz (Thomsen et al. 2001). Northern resident killer whale whistles have source levels ranging from 133 to 147 dB re: 1 µPa-m (Miller 2006). Individual Southern Resident killer whale pods have distinct call repertoires, with each pod being recognizable by its acoustic dialect (NMFS 2008e).

Killer whale pulsed calls are the most commonly observed type of signal associated with killer whales (Ford 1989). With both northern and Southern Resident killer whales, these signals are relatively long (600 to 2,000 msec) and range in frequency between 1 and 10 kilohertz; but may contain harmonics up to 30 kilohertz (Ford 1989). The variable calls of killer whales have source levels ranging from 133 to 165 dB while stereotyped calls have source levels ranging from 135 to 168 dB re: 1 µPa-m (Miller 2006). Killer whales use these calls when foraging and traveling (Ford 1989).

Killer whale hearing is one of the most sensitive of any odontocete, with a hearing range of 600 Hz to 114 kilohertz, with the most sensitive range being between 5 and 81 kilohertz (Branstetter et al. 2017a). The most sensitive frequency is 20 kilohertz, which corresponds with the approximate peak energy of the species' echolocation clicks (Szymanski et al. 1999).

Social Structure

Four levels of social structure have been identified among resident killer whales. The basic and most important social unit is the matriline, which is a highly stable hierarchical group of individuals linked by maternal descent (Baird 2000, Ford et al. 2000, Ford 2002, Ford and Ellis 2002). A matriline is usually composed of a female, her sons and daughters, and offspring of her daughters, spanning one to five (mean = 3) generations. Members maintain extremely strong bonds and individuals seldom separate from the group for more than a few hours. Permanent dispersal of individuals from resident matrilines has never been recorded (Bigg et al. 1990, Baird 2000, Ford et al. 2000, Barrett-Lennard and Ellis 2001) and the two recent separations of calves (A73 and L98) from their natal pods are considered anomalous. Matriarchal females likely hold important social knowledge that guides the behavior of individual matrilines (Boran and Heimlich 1999, McComb et al. 2001).

Groups of related matrilines are known as pods. Matrilines within pods share a common maternal ancestor from the recent past, making them more closely related to one another than to those of other pods (Baird 2000, Ford et al. 2000). Pods are less cohesive than matrilines and member matrilines may travel apart for periods of weeks or months. Nonetheless, matrilines associate more often with others from their pod than with matrilines from other pods. Most pods are comprised of one to four matrilines, but one Southern Resident killer whales pod (L pod) holds 12 matrilines. As noted above, currently, the Southern Resident killer whales DPS is separated into three matriarchal pods, J, K and L. As of the summer census for 2019 conducted by the Center for Whale Research, the population numbers of each pod were 22, 17 and 34 members, respectively. Since the 2019 census was conducted the Center for Whale Research has

reported an additional whale missing and presumed dead, bringing the population number to 72. Two new J-pod calves were born in September 2020.

Clans are the next level of social structure and are composed of pods with similar vocal dialects and a common but older maternal heritage (Ford 1991, Ford et al. 2000, Yurk et al. 2002). Those pods with similar dialects are presumably more closely related to one another than those with greater differences in their dialects (Ford 1991). However, vocalizations known as pulsed calls are not shared between different clans, indicating a lack of recent common ancestry between clans. Clans overlap in their geographic ranges and pods from different clans frequently intermingle. Pods (and clans) that regularly associate with one another are known as communities, which represent the highest level of social organization in resident killer whale societies (Ford et al. 2000, Ford 2002).

Distribution

Three kinds of killer whales occur along the Pacific Coast of the United States: Eastern North Pacific (ENP) Southern Resident killer whales, ENP offshore killer whales, and ENP transient killer whales. Of these only the Southern Resident killer whales are listed as endangered or threatened under the ESA. The Southern Resident killer whales DPS of killer whales is geographically, matrilineally, and behaviorally distinct from other killer whale populations. Southern Resident killer whales primarily occur in the inland waters of Washington State and southern Vancouver Island, although individuals from this population have been observed as far north as Southeast Alaska and to the south off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (NMFS 2005c). Southern Resident killer whales are the only marine mammal that begin and end their lives almost entirely within the action area.

Southern Resident killer whales spend a significant portion of the year in the inland waterways of the Georgia Strait, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods regularly occur in the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca (Felleman et al. 1991; Heimlich-Boran 1988; Olson 1998; Osborne 1999). The K and L pods typically arrive in May or June and remain in this core area until October or November, although both pods make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000). The J pod will occur intermittently in the Georgia Basin and Puget Sound during late fall, winter and early spring. During the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Felleman et al. 1991; Ford et al. 2000; Heimlich-Boran 1988; Olson 1998).

The local movement of Southern Resident killer whales usually follows the distribution of salmon, which are their preferred prey (Heimlich-Boran 1988; Heimlich-Boran 1986; Nichol and Shackleton 1996). Areas that are major corridors for migrating salmon, and therefore, for Southern Resident killer whales, include Haro Strait and Boundary Passage, the southern tip of

Vancouver Island, Swanson Channel off North Pender Island, and the mouth of the Fraser River delta, which is visited by all three pods in September and October (Felleman et al. 1991; Ford et al. 2000).

Occurrence in the Offshore Portion of the NWTT action area

To better predict the pattern of distribution of the endangered Southern Resident killer whales off the Washington, Oregon, and Northern California coasts, researchers integrated visual sightings, location data obtained between 2012 and 2016 from satellite-tagged Southern Resident killer whales, and acoustic detections from underwater hydrophones obtained from 6 to 13 recorders deployed from 2011 to 2015 off the Washington, Oregon, and California coast (Hanson et al. 2018; Hanson et al. 2017b; U.S. Department of the Navy 2018a). Along the Pacific coast, the distribution of satellite-tag locations confirms that Southern Resident killer whales generally inhabit nearshore waters and over multiple years have spent the highest amount of time near the mouth of the Columbia River and Westport, Washington (Hanson et al. 2018; Hanson et al. 2017a; U.S. Department of the Navy 2018a); their presence generally inshore of the action area boundary located 12 nautical miles from the coast.

Although less is known about the whales' movements in outer coastal waters, satellite tagging, opportunistic sighting, and acoustic recording data suggest that Southern Resident killer whales spend nearly all of their time on the continental shelf, within 34 km (21.1 mi) of shore in water less than 200 meters (656.2 feet) deep (Hanson et al. 2017b). According to Hanson et al. (2017b), almost all (96.5 percent) outer coastal locations of satellite-tagged Southern Resident killer whales occurred in continental shelf waters of 200 meters (656.2 feet) depth or less, 77.7 percent were in waters less than 100 meters (328.1 feet) depth, and only 5.3 percent were in waters less than 18 meters (59 feet). Overall, the tagged whales' locations were typically (95 percent) within 34 km (21.1 mi) of shore (with 83 percent within 20 km [12.4 mi] of shore and 54 percent within 10 km [6.2 mi] of shore), only 5 percent of locations were in waters within 2 km (1.3 mi) of shore, and only 5 percent were beyond 34 km (21.1 mi) from shore. (Hanson et al. 2017b). Farthest offshore detection of tagged SRKWs was 47 miles. Southern Resident killer whales were also acoustically detected by the monitoring hydrophones as far as 38.5 miles off Cape Flattery, at the northern extreme of the NWTT action area off Washington (Hanson et al. 2018; U.S. Department of the Navy 2018a), which is also the area where there appears to be the semi-permanent and highly productive eddy associated with the outflow from the Strait of Juan de Fuca (Dalla-Rosa et al. 2012; MacFadyen et al. 2008).

Satellite tagging also identified high-use offshore areas, primarily off the Washington outer coast. Although this area only represented 16.2 percent of the total satellite tag derived range, the tagged whales spent 53.1 percent of their time there. Acoustic data suggest that K and L pods spend a relatively large amount of time off the outer coast of Washington, with detections in every month of the year. Their time was further concentrated between Grays Harbor and the Columbia River (19.1 percent of their total time) (Hanson et al. 2017b). Within the NWTT action area, tagged Southern Resident killer whales used about 9.7 percent of area W-237 (see Figure

30 above) and spent about 15 percent of their time (though the amount varies between pods) (Hanson et al. 2017b). Off Oregon and California, more than 75 percent of the tagged whales' locations were closer to shore than the OPAREA boundary, which is located around 12 nautical miles from shore (Hanson et al. 2017b). Consistent with visual sightings, J-pod occurred frequently near the western entrance of the Strait of Juan de Fuca but spent relatively little time in other outer coastal areas. K and L pods, however, used the outer coastal waters along Washington and Oregon during winter months with more regularity (Hanson et al. 2017b).

Occurrence in the Inland Waters of the NWTT action area

Southern Resident killer whales spend large amounts of time in the Inland Waters portion of the action area. In spring and summer months, the Southern Resident killer whales stock is most frequently seen in the San Juan Islands region with intermittent sightings in Puget Sound (Whale Museum 2012). In the fall and early winter months, the Southern Resident killer whales are seen more frequently in Puget Sound, where returning chum salmon and Chinook salmon are concentrated (Osborne et al. 1988). The J pod is seen most frequently along the western shore of San Juan Island and is the only pod observed regularly in Puget Sound throughout winter (Heimlich-Boran 1988; Osborne 1999). The K pod is most frequently observed during May and June when they occur along the western shore of San Juan Island while searching for salmon. The L pod is the largest of the three pods (Ford et al. 1994) and frequently breaks off into separate subgroups.

The Southern Resident killer whales stock inhabits both inland Washington and southern British Columbia waters and offshore waters along the coast of North America (Carretta et al. 2018a). Photo-identification of individual whales through the years, as well as more recent satellite tagging and passive acoustic monitoring, has resulted in a substantial understanding of this stock's structure, behaviors, and movements in inland waters (Wiles 2016; Wright et al. 2017b). In spring and summer months, the Southern Resident killer whales stock is most frequently seen in the San Juan Islands region with intermittent sightings in Puget Sound (Olson and Osborne 2017; Olson et al. 2018; Shields et al. 2018), which is consistent with the "summer core area" identified during the establishment of the critical habitat for the species. In the fall and early winter months, the Southern Resident killer whales are seen more frequently in Puget Sound, where returning chum salmon, steelhead, and Chinook salmon are concentrated; Chinook salmon are targeted preferentially when available (Ford et al. 2009; Ford et al. 2016; Hanson et al. 2018). By winter, they spend progressively less time in the inland marine waters and more time off the coast of Washington, Oregon, and California (Black 2011; Cogan 2015; Hanson et al. 2017b; National Marine Fisheries Service 2016; Olson and Osborne 2017). There was a confirmed Southern Resident killer whales occurrence along the Washington State Ferries route between Bremerton and Seattle in December 2007, but the exact location of the sighting is not known (Orca Network 2012). The use of the Inland Waters portion of the NWTT action area by Southern Resident killer whales has declined in recent years as they shift their range in response

to reduced prey availability in Puget Sound (Olson and Osborne 2017; Olson et al. 2018; Shields et al. 2018).

While Southern Resident killer whales are frequently sighted in the main basin of Puget Sound, their presence near Navy installations varies from not present at all to infrequent sightings, depending on the season (Olson and Osborne 2017; Olson et al. 2018). There are no confirmed sightings of Southern Resident killer whales inside Hood Canal (NMFS 2019e). Southern Resident killer whales (J pod) were historically documented in Hood Canal by sound recordings in 1958 (Ford 1991), a photograph from 1973, and also anecdotal accounts of historical use, but these latter sightings may be transient whales (NMFS 2008g). Near NAVBASE Kitsap Bremerton and Keyport, the Southern Resident killer whale is also rare, with the last confirmed sighting in Dyes Inlet in 1997 (it is assumed transients will occasionally be present in these areas). Southern Resident killer whales have been observed in Saratoga Passage and Possession Sound near Naval Air Station Whidbey Island and Naval Station Everett, respectively, and have also been observed in southern Puget Sound in the Carr Inlet area.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Individuals belonging to the Alaska Resident stock are the killer whales most likely to occur in the SEAFAC region of the action area, and are more likely from spring through fall (Dahlheim et al. 2009). Southern Resident killer whales (L pod, 30 individuals) were photographically identified in Chatham Strait, Southeast Alaska (northwest of Behm Canal), in June 2007. Southern Resident killer whales were previously thought to only range as far north as the Queen Charlotte Islands, B. C.; however, this sighting extends their known range about 200 miles to the north (Carretta 2019). However, Alaska Resident stock of killer whales are more likely to be encountered in and around the Behm Canal than Southern Resident killer whales. We would expect encounters with Southern Resident killer whales near SEAFAC to be very rare. The effects of NWTT activities occurring in the Western Behm Canal portion of the action area on Southern Resident killer whales are extremely unlikely to occur and, therefore, are considered discountable.

Abundance Estimate

Photo-identification of individual whales through the years has resulted in a substantial understanding of this stock's structure, behaviors, and movements in inland waters. Recent abundance estimates for the Southern Resident killer whales DPS were 75 whales in 2019, and previously 75 whales in 2018 (Carretta 2019). As discussed above, based on the Center for Whale Research census, as of August 6, 2019, the population was comprised of 72 individuals, but two newborns were observed in September 2020, bringing the DPS to 74 individuals.

Natural Threats

Southern Resident killer whales, like many wild animal populations, experience highest mortality in the first year age class (Krahn et al. 2002; Olesiuk et al. 1990), although the reasons

for these mortalities are still uncertain. The causes could include poor mothering, infectious or non-infectious diseases, and infanticide (Gaydos et al. 2004). Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive Southern Resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. They did, however, identify pathogens in sympatric odontocete species that could threaten the long-term viability of the small Southern Resident killer whales population. Franks (2020) developed a stochastic individual-based model to predict the severity of a potential disease outbreaks in Southern Resident killer whales. He found that this population would be highly vulnerable to disease outbreaks, with cetacean morbillivirus, in particular, likely to produce outbreaks in which over half the population becomes infected. High levels of contaminants (discussed below) may increase the susceptibility of cetaceans to morbillivirus infection, or play an etiological role in the resulting hepatocytic damage (Reijnders et al. 1999).

As discussed above, strong inbreeding has been documented and inbreeding depression, or fitness effects of inbreeding (e.g., lower survival or fecundity), may be a concern for the Southern Resident killer whales DPS (Ford et al. 2018). Ruggerone et al. (2019) reported on an unprecedented, synchronized biennial pattern of birth and mortality in the Southern Resident killer whale population. From 1998−2017, mortality of newborn and older whales was 3.6 times higher (61 versus 17 whales) and successful births 50 percent lower (16 versus 32 whales) in even years than in odd years as the population decreased from 92 to only 76 whales. The authors suggest this biennial pattern may be linked to an extreme biennial pattern of pink salmon abundance. However, pink salmon are not an important prey species for Southern Resident killer whales, and the causal mechanism of the proposed indirect link between these biennial patterns remain unknown.

Anthropogenic Threats

Several human activities have contributed to the decline of Southern Resident killer whales. Southern Resident killer whales were once shot deliberately in Washington and British Columbia (Baird 2001; Olesiuk et al. 1990). Between 1967 and 1973, 43 to 47 killer whales were removed from the population for displays in oceanaria. As a result of those removals, the Southern Resident killer whale population declined by about 30 percent. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and the current low of 74 individuals, including two calves born in September 2020. Over the same time interval, Southern Resident killer whales have been exposed to changes in the distribution and abundance of their prey base (primarily Pacific salmon) which has reduced their potential forage base, potential competition with salmon fisheries, which reduces their realized forage base, disturbance from vessels, and persistent toxic chemicals in their environment.

Prey Availability

A growing body of evidence documents how Southern Resident killer whales are affected by limitations of their primary prey, Chinook salmon (Matkin et al. 2017). When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Southern Resident killer whales individuals are occasionally found in poor condition, which may indicate nutritional stress. Individuals who have been identified as being in poor condition have a higher probability of dying than individuals who have not been so identified (Hilborn et al. 2012). Southern Resident killer whale fecundity, death rates and rates of population increase have shown statistical correlations with some indices of Chinook salmon abundance (Hilborn et al. 2012). However, Hilborn et al. (2012) note that "considerable caution is warranted in interpreting the correlative results as confirming a linear causal relationship between Chinook salmon abundance and SRKW vital rates." These relationships are likely non-linear, the relationships may be influenced by small sample sizes of killer whale births and deaths, and the relationships may arise from uncertainties in the indices of Chinook salmon abundance used for fisheries management (PFMC 2020).

Salmon have declined because of land alteration throughout the Pacific Northwest associated with agriculture, timber harvest practices, the construction of dams, and urbanization, fishery harvest practices, and hatchery operations. Many of the salmon populations that were once abundant historically have declined to the point where they have been listed as endangered or threatened with extinction.

Competition from other salmonid predators, including other resident killer whales, pinnipeds, and fisheries may also be limiting the growth of the Southern Resident killer whale population (Chasco et al. 2017b). Results from Chasco et al. (2017b) suggest that, at least in recent years, competition with other marine mammals is a more important factor limiting the growth of this endangered population than competition with human fisheries. Primarily due to increases in harbor seal abundance, since 1970 predation on Chinook salmon runs within Puget Sound has increased approximately nine-fold in terms of numbers and doubled in terms of biomass (Chasco 2019). Availability of Chinook salmon to Southern Residents may also be impacted by sound that interferes with the whales' echolocation from vessels or other sounds sources.

Killer whale populations inhabiting Alaskan waters have a much broader range of salmon populations available as prey (Chasco et al. 2017b). In contrast, Southern Resident killer whales, distributed in the Salish Sea and U.S. west coast, is the most at-risk population with a long-term growth rate close to zero, and a much smaller diversity of salmon populations available as prey. This narrower selection of Chinook salmon stocks available to Southern Resident killer whales may be a competitive disadvantage compared to higher latitude killer whale populations (Chasco et al. 2017b).

Availability of Southern Resident killer whales prey along the outer coast is likely limited at particular times of year due to the small run sizes of some important Chinook salmon stocks, as

well as the distribution of preferred adult Chinook salmon that may be relatively spread out prior to their aggregation when returning to their natal rivers. Nutritional demands on SRKW are presumed to be greatest in winter when their salmonid prey are more widely dispersed, smaller in size and other non-salmonid prey appear to be a larger fraction of the diet. Thermoregulatory demands may also influence nutritional demands during winter. SRKW then transition to spring, eventually subsisting on a diminishing number of spring/summer run adult Chinook salmon approaching river mouths inside and outside the Salish Sea until the Fraser River Chinook salmon runs peak in mid- to late-August. Variable levels of the thyroid hormone triiodothyronine (or T3) in Southern Resident killer whales during late spring and summer may indicate nutritional stress during the period spent in the Salish Sea (Wasser et al. 2017). Wasser et al. (2017) suggest that the increasingly common occurrence of Southern Resident killer whales births outside the typical winter calving period may be an indication of the increased unpredictability of diminishing fish runs, along with the corresponding high rate of late reproductive loss.

NMFS and Washington Department of Fish and Wildlife (WDFW) developed a framework to identify Chinook salmon stocks that are important to Southern Resident killer whales based on three evaluation factors: 1) stock is observed as part of Southern Resident killer whale diet based on prey tissues/scales and fecal samples; 2) stock is consumed during times of potential reduced body condition and increased diet diversity; and 3) degree of spatial and temporal overlap between fish stock and Southern Resident killer whales (WDFW 2018). Based on this conceptual model, the highest priority Chinook salmon stocks for Southern Resident killer whales, in order of importance, are as follows: 1) Northern Puget Sound (Fall) and Southern Puget Sound (Fall), 2) Lower Columbia (Fall) and Georgia Strait (Fall), 3) Upper Columbia (Fall) and Snake River (Fall), Fraser (Spring), and Lower Columbia (Spring), 4) Middle Columbia (Fall), and 5) Snake River (Spring-Summer) and Northern Puget Sound (Spring). Results of another Southern Resident killer whales feeding study suggest that recovering Fraser River (summer-early fall) and Columbia River Chinook salmon (early spring) runs should be among the highest priorities for managers aiming to recover this endangered population of killer whales (Wasser et al. 2017).

Resident killer whales selectively prey upon Chinook salmon, particularly the oldest and largest individuals (Ford et al. 1998; Hanson et al. 2010; Herman et al. 2005a). About 90 percent of Chinook salmon eaten by residents are 4–6 years old (Ford and Ellis 2006). Thus, in addition to declining numbers of available Chinook salmon prey, Southern Resident killer whales can be adversely affected by changes in the size-structure and age-structure of Chinook salmon populations over time. Ohlberger et al. (2018) found that the size-structure and age-structure of Chinook salmon have changed considerably across the Northeast Pacific Ocean since the late 1970s. While changes in age proportions showed some region-specific trends, many of the populations coast-wide have experienced declines in the proportion of ocean age 4 and 5 fish, and proportions of ocean age 2 fish have generally increased. Furthermore, the size-at-age of ocean age 1 and 2 fish has increased over time, at least in most hatchery populations, whereas the

size-at-age of ocean age 4 and 5 fish has declined considerably, especially since about 2000 (Ohlberger et al. 2018).

Vessel Impacts

Since the 1970s commercial shipping, whale watching, ferry operations, and recreational boat traffic have increased in Puget Sound and the coastal islands of southern British Columbia. Vessel traffic exposes Southern Resident killer whales to several threats that have consequences for the species' likelihood of survival and recovery. First, heavy vessel traffic increases the risk of Southern Resident killer whales being struck by vessels. In 2005, a Southern Resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. In 2006, an adult male Southern Resident killer whale, L98, was killed in a collision with a tug boat; given the gender imbalances in the Southern Resident killer whale population, we assume that the death of this adult male would have reduced the demographic health of this population. In fall 2016 another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that the whale died of blunt force trauma to the head "the animal had injuries consistent with blunt trauma to the dorsal side, and a hematoma indicating that it was alive at the time of injury and would have survived the initial trauma for a period of time prior to death" (Fisheries and Oceans Canada 2019). The injuries are consistent with those incurred during a vessel strike.

Second, the number and proximity of vessels, particularly whale-watch vessels in the areas occupied by Southern Resident killer whales, represents a source of chronic disturbance and stress for this population. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Cotton 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Several investigators have studied the effects of whale watch vessels on marine mammals (Amaral and Carlson 2005a; Au and Green 2000; Corkeron 1995; Erbe 2002; Felix 2001; Magalhaes et al. 2002; Richter et al. 2006; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002a). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. The whales' responses changed with these different variables and, in some circumstances, the whales did not respond to the vessels. In other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

In addition to the disturbance associated with the presence of vessels, vessel traffic affects the acoustic ecology of Southern Resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of Southern Resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15 percent during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2009) reported that Southern Resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales need to increase the duration of their vocalization to avoid masking by the boat noise. Holt et al. (2015) found that an increase in vocal effort by bottlenose dolphins, manifested as an increase in the acoustic energy of sounds produced, resulted in an increase in metabolic rate relative to resting. Vessels in the path of the whales can also interfere with important social behaviors such as prey sharing (Ford and Ellis 2006) or nursing (Kriete 2007). With the disruption of feeding behavior that has been observed, it is estimated that the presence of vessels could result in an 18 percent decrease in energy intake, a consequence that could have a significant negative effect on an already prey-limited species (Williams et al. 2006, Lusseau et al. 2009).

Federal vessel regulations were established in 2011 to prohibit vessels from approaching killer whales within 200 yards (182.9 meters (m)) and from parking in the path of the whales within 400 yards (365.8 meters). These regulations apply to all vessels in inland waters of Washington State with exemptions to maintain safe navigation and for government vessels in the course of official duties, ships in the shipping lanes, research vessels under permit, and vessels lawfully engaged in commercial or treaty Indian fishing that are actively setting, retrieving, or closely tending fishing gear (76 FR 20870, April, 14, 2011). In December 2017, NMFS completed a technical memorandum evaluating the effectiveness of regulations adopted in 2011 to help protect SRKWs from the impacts of vessel traffic and noise (Ferrara et al. 2017). In the assessment, Ferrara et al. (2017) used five measures: education and outreach efforts, enforcement, vessel compliance, biological effectiveness, and economic impacts. For each measure, the trends and observations in the 5 years leading up to the regulations (2006-2010) were compared to the trends and observations in the 5 years following the regulations (2011-2015). The memo finds that the regulations have benefited the whales by reducing impacts without causing economic harm to the commercial whale-watching industry or local communities. The authors also find room for improvement in terms of increasing awareness and enforcement of the regulations, which would help improve compliance and further reduce biological impacts to the whales. In 2019, a new Washington State law was signed that increases vessel viewing distances from 200 to 300 yards (183 to 274 meters) to the side of the whales and reduces vessel speed within one-half nautical mile of the whales to seven knots over ground. A new effort based in Canada, the Enhancing Cetacean Habitat and Observation (ECHO) Program, is a Vancouver Fraser Port Authority-led initiative aimed at better understanding and managing

the impact of shipping activities on at-risk whales throughout the southern coast of British Columbia.

Contaminants

The absence of contaminants or other agents of a type and/or amount that would inhibit reproduction, impair immune function, result in mortalities, or otherwise impede the growth and recovery of the Southern Resident killer whale population is a habitat feature essential for the species' recovery. Water quality in Puget Sound, in general, is degraded as described in the Puget Sound Partnership 2018-2022 Action Agenda and Comprehensive (Puget Sound Partnership 2018). For example, toxicants in Puget Sound persist and build up in marine organisms including SRKWs and their prey resources, despite bans in the 1970s of some harmful substances and cleanup efforts. Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants has been documented in Southern Resident killer whales (Krahn et al. 2007; Ross et al. 2000). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007).

Sonar

The impacts of military mid-frequency sonar on killer whales have not been directly studied, but observations are available from an event that occurred in the Strait of Juan de Fuca and Haro Strait on 5 May 2003, when members of J pod were present off southwestern San Juan Island. A U.S. Navy guided-missile destroyer (USS Shoup) passed through the strait while operating its mid-frequency AN/SQS-53C sonar during a training exercise. Members of J pod were present in the strait and unusual behaviors by whales in response to the sound were reported by local researchers (NMFS 2004d, U.S. Navy, Pacific Fleet 2004). NOAA assessed the acoustic exposures and reported that it was unlikely that the whales experienced either temporary or permanent hearing loss. Based on the duration and received levels, and levels known to cause behavioral reactions in other cetaceans, J pod received exposure levels likely to cause behavioral disturbance, which is consistent with eyewitness accounts (NMFS 2004d)

The Navy will not use active sonar during training or testing within the Puget Sound and Strait of Juan de Fuca Mitigation Area, unless a required element necessitates that the activity be conducted in NWTT Inland Waters during (1) Unmanned Underwater Vehicle Training, (2) Civilian Port Defense – Homeland Security Anti-Terrorism/Force Protection Exercises, (3) activities conducted at designated Naval Sea Systems Command testing sites, and (4) pierside sonar maintenance or testing at designated locations. Only a few Navy vessels operating in the greater Puget Sound area are equipped with mid-frequency active sonar. Typical Navy mid-

frequency active sonar use in Puget Sound is limited to pier-side system maintenance and training on designated ranges. As a precautionary measure, any ship, submarine or unit wanting to use active mid-frequency sonar in Puget Sound, including the Strait of Juan de Fuca, is required to obtain prior permission from the appropriate designated Command authority. See Section 3.5 above for complete details on mitigation measures in place to minimize impacts of Navy sonar on Southern Resident killer whales.

Status and Trends

In the mid- to late-1800s, Southern Resident killer whales were estimated to have numbered around 200 individuals. The current population estimate of 74, including two new born calves in September 2020, represents a decline from the recent past, when in 2012, there were 85 whales. Population abundance has fluctuated over time with a maximum of 99 whales in 1995 (Carretta 2019), with an increase between 1974 and the mid-90s, from 76 to 93 individuals. As compared to stable or growing populations, the DPS reflects lower fecundity and has demonstrated little to no growth in recent decades (NMFS 2016k). For the period between 1974 and the mid-1990s, when the population increased from 76 to 93 animals, the population growth rate was 1.8 percent (Ford et al. 1994). More recent data indicate the population is now in decline (Carretta 2019). Prior to 2019, there had been no Southern Resident killer whales born since 2015[17]. In 2019, two whales were born, one in L pod, and one in J pod. Four whales died or were presumed dead following the 2018 census, as of July 1, 2019 (NMFS 2019h). Nutritional stress in the forms of lack of prey, toxin loads, and vessel disturbance is thought to be a possible contributing factor to low offspring production for Southern Resident killer whales. Analysis of fecal hormones has indicated several miscarriages in recent years, particularly in late pregnancy (Wasser et al. 2017). The number of effective breeders in the population is about 26 (Ford et al. 2018).

At population sizes between 70 and 90 individuals, we would expect Southern Resident killer whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox 2007) —including stochastic sex determination (Lande et al. 2003)—and the effects of these phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson et al. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008). For example, the small number of adult male Southern Resident killer whales might represent a stable condition for this species or it might reflect the effects of stochastic sex

[17] http://www.orcanetwork.org/Main/index.php?categories_file=Births%20and%20Deaths; accessed 2/13/2020.

determination. Regardless, a high mortality rates among adult males in a population with a smaller percentage of males would increase the imbalance of male-to-female gender ratios in this population and increase the importance of the few adult males that remain.

At these population sizes, populations experience higher extinction probabilities because stochastic sexual determination leaves them with harmful imbalances between the number of male or female animals in the population (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson et al. 2006). In general, an individual's contribution to the growth (or decline) of the population it represents depends, in part, on the number of individuals in the population: the smaller the population, the more the performance of a single individual is likely to affect the population's growth or decline (Coulson et al. 2006). Given the small size of the Southern Resident killer whale population, the performance (= "fitness," measured as the longevity of individuals and their reproductive success over their lifespan) of individual whales would be expected to have appreciable consequences for the growth or decline of the Southern Resident killer whale population.

After thorough genetic study, the Biological Review Team concluded that Southern Resident killer whales DPS of killer whales were discrete from other killer whale groups (NMFS 2008). Despite the fact that their ranges overlap, Southern Resident killer whales DPS of killer whales do not intermix with Northern Resident killer whales. Low genetic diversity within a population is believed to be in part due to the matrilineal social structure (NMFS 2008e). Inbreeding is a concern for the Southern Resident killer whales; four cases of inbreeding have been recorded, two between parent and offspring, one between paternal half-siblings, and one between an uncle and a half-niece; the fitness consequences of inbreeding in this population are unknown (Ford et al. 2018).

These phenomena would likely increase the extinction probability of Southern Resident killer whales and amplify the potential consequences of human-related activities on this species. Based on their population size and life history (that is, slow-growing mammals that give birth to single calves with several years between births), we assume that Southern Resident killer whales would have elevated extinction probabilities due to a combination of exogenous anthropogenic threats (as discussed above and in Section 7 *Environmental Baseline*), natural phenomena, and endogenous threats resulting from their small population size. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer Southern Resident killer whales remain in these circumstances, the greater their extinction probability becomes.

Critical Habitat

In 2006, NMFS designated critical habitat for the Southern Resident killer whales DPS of killer whale (71 FR 69054). The three specific areas in Washington are: (1) the Summer Core Area in

Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca, which comprise approximately 2,560 mi^2 (6,630 km^2) of marine habitat (Figure 39). As noted above, Hood Canal was not included in the inland critical habitat designation due to the lack of confirmed sightings of Southern Resident killer whales inside Hood Canal.

NMFS determined that the national security benefits of exclusion outweighed the conservation benefits of designation for 18 DOD owned or controlled sites. These sites, covering 112 square miles, include: shore-based facilities; several nearshore areas around structures such as docks and piers; Naval Undersea Warfare Center, Keyport; Strait of Juan de Fuca naval air-to-surface weapon range restricted area; Strait of Juan de Fuca and Whidbey Island naval restricted areas; Admiralty Inlet naval restricted area; Port Gardner Naval Base restricted area; Port Orchard Passage naval restricted area; Sinclair Inlet naval restricted area; Carr Inlet naval restricted area; Townsend/Indian Island/Walan Point naval restricted area; and Crescent Harbor Explosive Ordnance Units Training Area.

The physical and biological features (previously referred to as *primary constituent elements* or PCEs) of inland designated critical habitat that are essential to the conservation of Southern Resident killer whales DPS of killer whales are: (1) water quality to support growth and development; (2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and (3) inter-area passage conditions to allow for migration, resting, and foraging.

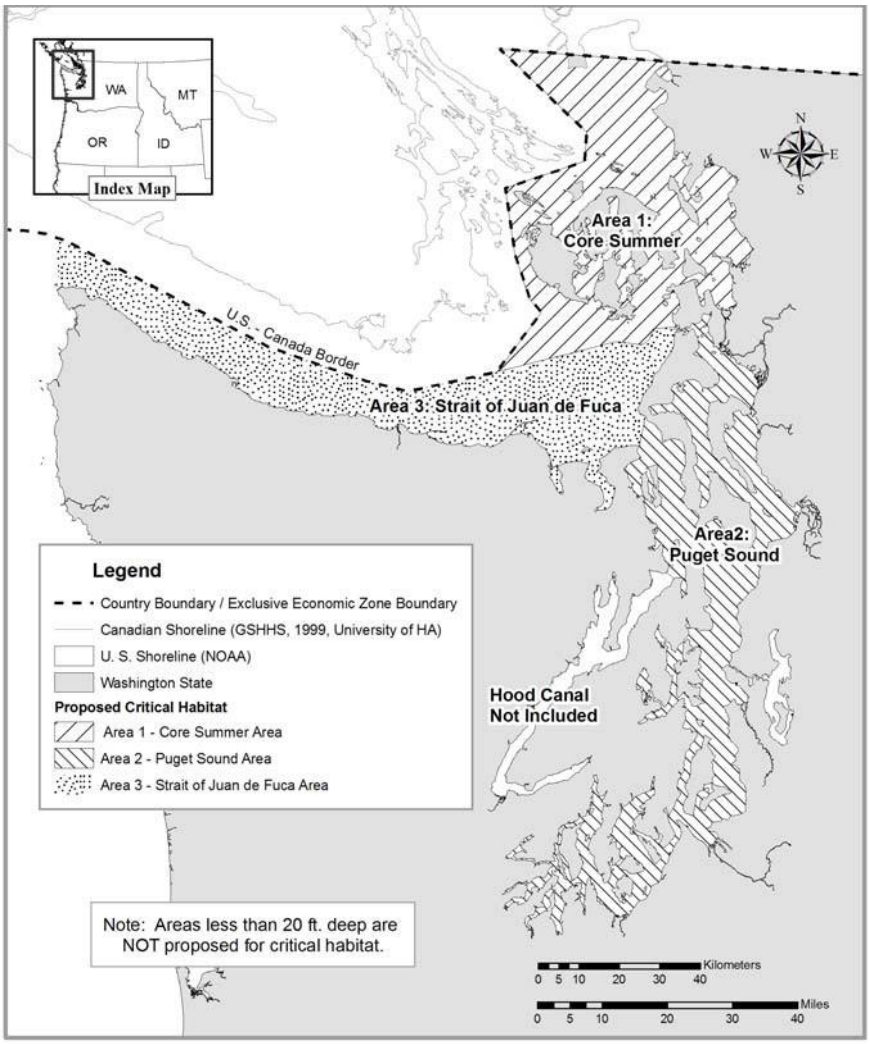


Figure 39. Map identifying inland designated critical habitat for the endangered Southern Resident killer whales DPS of killer whale

On September 19, 2019, NMFS proposed to revise the critical habitat designation for Southern Resident killer whales by expanding it to include six new areas along the U.S. West Coast (84 FR 49214), while keeping the current designated critical habitat area in Washington (Figure 40). The proposed new areas along the U.S. West Coast include roughly 16,167 mi^2 (41,873.8 km^2) of marine waters between the 6.1 meter depth contour and the 200 meter (652.2 feet) depth contour from the U.S. international border with Canada south to Point Sur, California (Table 66). Area 1 (Coastal Washington and Northern Oregon Inshore Area) and Area 2 (Coastal Washington and Northern Oregon offshore area) have the same northern and southern boundaries, but are separated longitudinally at the 50 meter (164.0 foot) isobath. Areas 1 and 2 are both considered high-use areas for Southern Resident killer whales based on J, K, and L pod presence documented through sightings, acoustic recordings, and satellite tag data. Prey is the primary essential feature of Areas 1 and 2.

NMFS has proposed excluding the Quinault Range Site (see map below) from the critical habitat designation. The total area proposed for exclusion is 1,146.9 mi^2 (2,970.5 km^2) or 6.6 percent of potential coastal critical habitat.

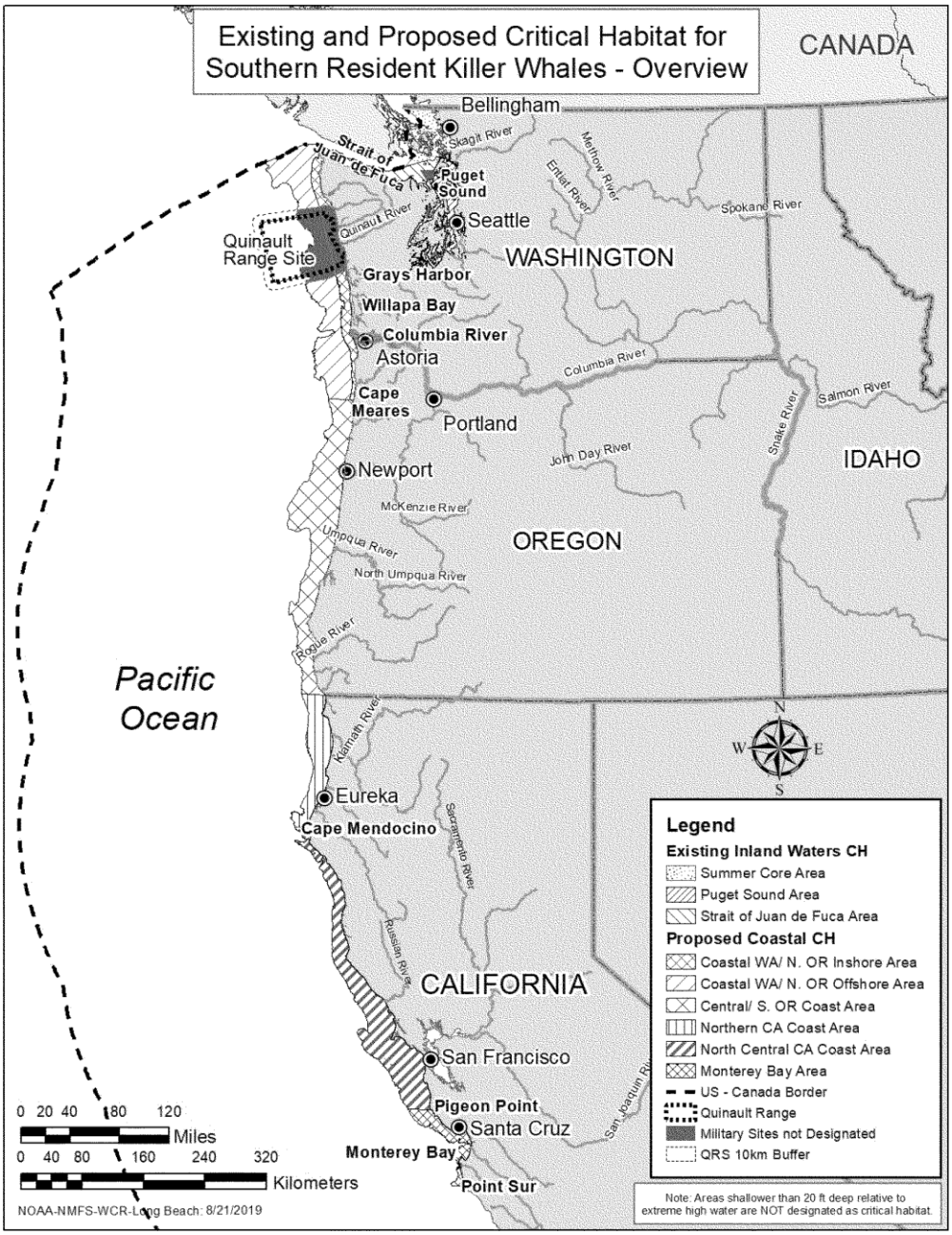


Figure 40. Existing and proposed critical habitat for Southern Resident killer whales.

The physical and biological features of proposed coastal critical habitat that are essential to the conservation of Southern Resident killer whales DPS of killer whales are: (1) water quality to

support growth and development; (2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and (3) inter-area passage conditions to allow for migration, resting, and foraging.

Table 66. Southern Resident killer whale proposed coastal critical habitat specific area descriptions.

Area	Boundaries	Size	Essential feature (primary feature in bold)
1 - Coastal Washington/Northern Oregon Inshore Area	U.S. ocean waters west of a line connecting Cape Flattery, Washington (48°23'10'' N/124°43'32'' W), Tatoosh Island, Washington (48°23'30'' N/124°44'12'' W), and Bonilla Point, British Columbia (48°35'30'' N/124°43'00'' W), from the U.S. international border with Canada south to Cape Meares (45°29'12'' N), between the 6.1-m and 50-m isobath contours.	1,441.9 mi^2 (3,734.6 km^2)	**Prey**, passage, water quality
2 - Coastal Washington/Northern Oregon Offshore Area	U.S. ocean waters west of a line connecting Cape Flattery, Washington (48°23'10'' N/124°43'32'' W), Tatoosh Island, Washington (48°23'30'' N/124°44'12'' W), and Bonilla Point, British Columbia (48°35'30'' N/124°43'00'' W), from the U.S. international border with Canada south to Cape Meares (45°29'12'' N), between the 50-m and 200-m isobath contours.	4,617.2 mi^2 (11,958.6 km^2)	**Prey**, passage, water quality
3 - Central/Southern Oregon Coast Area	Cape Meares (45°29'12'' N) south to the OR/CA border (42°00'00'' N), between the 6.1-m and 200-m isobath contours.	4,962.6 mi^2 (12,853.1 km^2)	**Passage**, prey, water quality
4 - Northern California Coast Area	OR/CA border (42°00'00'' N) south to Cape Mendocino, CA (40°26'19'' N), between the 6.1- m and 200-m isobath contours.	1,606.8 mi^2 (4,161.5 km^2)	**Prey**, passage, water quality
5 – North Central California Coast Area[a]	Cape Mendocino, CA (40°26'19'' N) south to Pigeon Point, CA (37°11'00'' N), between the 6.1-m and 200-m isobath contours.	3,976.2 mi^2 (10,298.4 km^2)	**Passage**, prey, water quality
6 - Monterey Bay Area[a]	Pigeon Point, CA (37°11'00'' N) south to Point Sur, CA (36°18'00'' N), between the 6.1-m and 200-m isobath contours.	710.1 mi^2 (1,839.2 km^2)	**Prey**, passage, water quality

[a] Note: proposed coastal critical habitat areas 5 and 6 are outside of the NWTT action area.

Recovery Goals

The Southern Resident killer whale recovery program includes actions to address the following goals:

1. Prey Availability: Support salmon restoration efforts in the region including habitat, harvest and hatchery management considerations and continued use of existing NMFS authorities under the ESA and Magnuson-Stevens Fishery Conservation and Management Act to ensure an adequate prey base
2. Pollution/Contamination: Clean up existing contaminated sites, minimize continuing inputs of contaminants harmful to killer whales, and monitor emerging contaminants.
3. Vessel Effects: Continue with evaluation and improvement of guidelines for vessel activity near Southern Resident killer whales and evaluate the need for regulations or protected areas.
4. Oil Spills: Prevent oil spills and improve response preparation to minimize effects on Southern Resident killer whales and their habitat in the event of a spill.
5. Acoustic Effects: Continue agency coordination and use of existing ESA and MMPA mechanisms to minimize potential impacts from anthropogenic sound.
6. Education and Outreach: Enhance public awareness, educate the public on actions they can participate in to conserve killer whales and improve reporting of Southern Resident killer whale sightings and strandings.
7. Response to Sick, Stranded, Injured Killer Whales: Improve responses to live and dead killer whales to implement rescues, conduct health assessments, and determine causes of death to learn more about threats and guide overall conservation efforts.
8. Transboundary and Interagency Coordination: Coordinate monitoring, research, enforcement, and complementary recovery planning with Canadian agencies, and Federal and State partners.
9. Research and Monitoring: Conduct research to facilitate and enhance conservation efforts. Continue the annual census to monitor trends in the population, identify individual animals, and track demographic parameters.

Delisting criteria are as follows: 1) killer whales have sustained an average growth of 2.3 percent per year for 28 years, 2) population parameters are consistent with a healthy growing population, and 3) threats have been addressed. Criteria for downlisting are: 1) interim downlisting criteria of average growth of 2.3 percent per year for 14 years, and 2) progress toward addressing threats represent sustained growth to indicate that the population could be downlisted to threatened. For more details on recovery goal and delisting/downlisting criteria see the 2008 Final Recovery Plan for the Southern Resident killer whale (NMFS 2008e).

6.2.5 Sei Whale

The sei whale is a widely distributed baleen whale found in all major oceans (Figure 41). Sei whales are distinguishable from other whales by a long, sleek body that is dark bluish-gray to

black in color and pale underneath, and a single ridge located on their rostrum. The sei whale was originally listed as endangered on December 2, 1970.

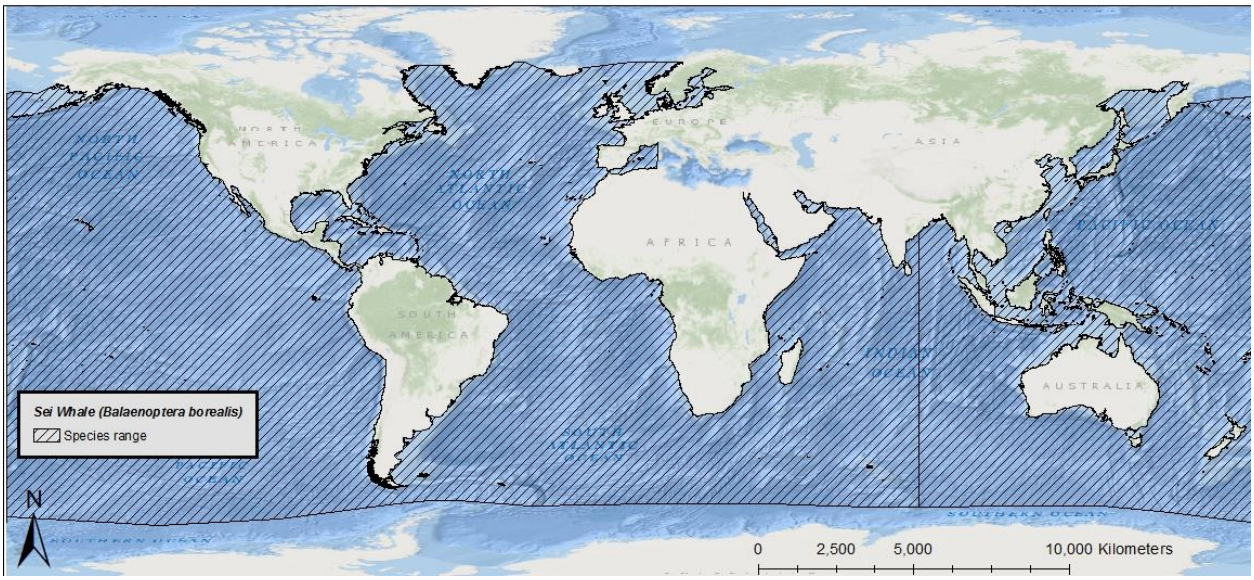


Figure 41. Map identifying the range of the endangered sei whale.

Information available from the recovery plan (NMFS 2011e), recent SAR (Carretta 2019), and status review (NMFS 2012d) were used to summarize the life history, population dynamics, and status of the species as follows.

Distribution

The sei whale occurs in all oceans of the world except the Arctic. The migratory pattern of this species is thought to encompass long distances from high-latitude feeding areas in summer to low-latitude breeding areas in winter; however, the location of winter areas remains largely unknown (Perry et al. 1999). Sei whales are often associated with deeper waters and areas along continental shelf edges (Hain et al. 1985). This general offshore pattern is disrupted during occasional incursions into shallower inshore waters (Waring et al. 2004). The species appears to lack a well-defined social structure and individuals are usually found alone or in small groups of up to six whales (Perry et al. 1999). When on feeding grounds, larger groupings have been observed (Gambell 1985b).

In the western Atlantic Ocean, sei whales occur from Nova Scotia and Labrador in the summer months and migrate south to Florida, the Gulf of Mexico, and the northern Caribbean (Gambell 1985b). In the eastern Atlantic Ocean, sei whales occur in the Norwegian Sea (as far north as Finnmark in northeastern Norway), occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Gambell 1985b).

In the North Pacific Ocean, sei whales occur from the Bering Sea south to California (on the east) and the coasts of Japan and Korea (on the west). During the winter, sei whales are found from 20° to 23° N (Gambell 1985b; Masaki 1977).

Sei whales occur throughout the Southern Ocean during the summer months, although they do not migrate as far south to feed as blue or fin whales. During the austral winter, sei whales occur off Brazil and the western and eastern coasts of Southern Africa and Australia. During the winter, sei whales are found from 20° to 23°N (Gambell 1985b; Masaki 1977). Sasaki et al. (2013) demonstrated that sei whale in the North Pacific are strongly correlated with sea surface temperatures between 13.1 and 16.8°C.

Occurrence in the Offshore Portion of the NWTT action area

Sei whales are expected to be present in the offshore area of the action area (Barlow 2016; Williams and Thomas 2007). Sei whales are distributed offshore in waters off the U.S. West Coast (Carretta et al. 2012). They are generally found feeding along the California Current, preferring deep water habitat along the continental shelf break (Perry et al. 1999). During six systematic ship surveys conducted between 1991 and 2008 in waters off the U.S. West Coast to approximately 300 nautical miles offshore, there were a total of 10 sei whale sightings, four of which were in waters off Oregon and Washington (Barlow 2010). There were no sei whale sightings during more coastal (out to about the 660 foot [200 meter] isobath) ship surveys off the northern Washington coast between 1995 and 2002 (Calambokidis et al. 2004a). Sei whales migrate into the Action area in summer and fall and are found farther south in winter (Perry et al. 1999).

Occurrence in the Inland Waters of the NWTT action area

Sei whales are considered rare in the Inland Waters portion of the action area, including Puget Sound. A sei whale washed ashore west of Port Angeles in the Strait of Juan de Fuca during September 2003 (Preston 2003), but this is considered an unusual event.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Sei whales are not expected to occur within the SEAFAC region of the action area since it is well inland of the areas normally inhabited by this species.

Population Structure

Two subspecies of sei whale are recognized, *B. b. borealis* in the Northern Hemisphere and *B. b. schlegellii* in the Southern Hemisphere. Based on genetic analyses, there appears to be some differentiation between sei whale populations in different ocean basins. An early study of allozyme variation at 45 loci found some genetic differences between Southern Ocean and the North Pacific sei whales (Wada and Numachi 1991). However, more recent analyses of mtDNA control region variation show no significant differentiation between Southern Ocean and the North Pacific sei whales, though both appear to be genetically distinct from sei whales in the North Atlantic (Baker and Clapham 2004; Huijser et al. 2018). Within ocean basin, there appears to be intermediate to high genetic diversity and little genetic differentiation despite there being different managed stocks (Danielsdottir et al. 1991; Huijser et al. 2018; Kanda et al. 2011; Kanda et al. 2006; Kanda et al. 2015; Kanda et al. 2013).

Sei whales are distributed worldwide, occurring in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere. Very little is known about the distribution of sei whales in the northeast Pacific. Generally, the species occupies pelagic habitats, and is very rarely seen inshore; over 3,700 sei whales were killed by whales offshore of the west coast of Vancouver Island. In the recent past, two sei whales have been sighted in Canadian Pacific waters, one in 2004 off southeastern Haida Gwaii, and the other in 2008 near Learmonth Bank in Dixon Entrance (Nichol 2011).

Some mark-recapture, catch distribution, and morphological research indicate more than one population may exist in the North Pacific Ocean—one between 155° and 175° W, and another east of 155° W (Masaki 1976; Masaki 1977). Sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska and south to California to the east and Japan and Korea to the west (Leatherwood et al. 1982b; Nasu 1974). Sightings have also occurred in Hawaiian waters (Smultea et al. 2010). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998). Whaling data suggest that sei whales do not venture north of about 55° N (Gregr et al. 2000). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July to September, although other researchers question these observations because no other surveys have reported sei whales in the northern and western Bering Sea. Harwood (1987) evaluated Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea. Harwood (1987) reported that 75 to 85 percent of the North Pacific population resides east of 180°. During winter, sei whales are found from 20° to 23° N (Gambell 1985b; Masaki 1977). Considering the many British Columbia whaling catches in the early to mid-1900s, sei whales have clearly utilized this area in the past (Gregr et al. 2000; Pike and Macaskie 1969).

Sei whales appear to prefer to forage in regions of steep bathymetric relief, such as continental shelf breaks, canyons, or basins situated between banks and ledges (Best and Lockyer 2002; Gregr and Trites 2001; Kenney and Winn 1987), where local hydrographic features appear to help concentrate zooplankton, especially copepods. In their foraging areas, sei whales appear to associate with oceanic frontal systems (Horwood 1987). In the north Pacific, sei whales are found feeding particularly along the cold eastern currents (Perry et al. 1999).

In the early to mid-1900s, sei whales were hunted off the coast of British Columbia (Gregr et al. 2000; Pike and Macaskie 1969). Masaki (1977) presented sightings data on sei whales in the North Pacific from the mid-1960s to the early 1970s. Over that time interval sei whales did not appear to occur in waters of Washington State and southern British Columbia in May or June, their densities increased in those waters in July and August (1.9 to 2.4 and 0.7 to 0.9 whales per 100 mi of distance for July and August, respectively), then declined again in September. More recently, sei whales have become known for an irruptive migratory habit in which they appear in an area then disappear for time periods that can extend to decades. Based on a sei whale that stranded near Port Angeles and the sei whales observed by Forney and her co-workers (Forney

2007), we know that these whales still occur in waters off Washington, Oregon, and northern California.

Abundance Estimate

Models indicate that total abundance declined from 42,000 to 8,600 individuals between 1963 and 1974 in the North Pacific Ocean. More recently, the North Pacific Ocean population was estimated to be 29,632 (95 percent confidence intervals 18,576 to 47,267) between 2010 and 2012 (IWC 2016; Thomas et al. 2016). The best abundance estimate for sei whales for the waters of the U.S. West Coast is 519 (CV=0.40); the current minimum population estimate is 374 (Carretta et al. 2020).

Natural Threats

Andrews (1916) suggested that killer whales attacked sei whales less frequently than fin and blue whales in the same areas. Sei whales engage in a flight responses to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Endoparasitic helminths (worms) are commonly found in sei whales and can result in pathogenic effects when infestations occur in the liver and kidneys (Rice 1977).

Anthropogenic Threats

Human activities known to threaten sei whales include whaling, commercial fishing, and maritime vessel traffic. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting may occur in some areas.

Sei whales, because of their offshore distribution and relative scarcity in U.S. Atlantic and Pacific waters, probably have a lower incidence of entrapment and entanglement than fin whales. Data on entanglement and entrapment in non-U.S. waters are not reported systematically. Heyning and Lewis (1990) made a crude estimate of about 73 rorquals killed/year in the southern California offshore drift gillnet fishery during the 1980s. Some of these may have been fin whales instead of sei whales. Some balaenopterids, particularly fin whales, may also be taken in the drift gillnet fisheries for sharks and swordfish along the Pacific coast of Baja California, Mexico (Barlow et al. 1997). Heyning and Lewis (1990) suggested that most whales killed by offshore fishing gear do not drift far enough to strand on beaches or to be detected floating in the nearshore corridor where most whale-watching and other types of boat traffic occur. Thus, the small amount of documentation may not mean that entanglement in fishing gear is an insignificant cause of mortality. Observer coverage in the Pacific offshore fisheries has been too low for any confident assessment of species-specific entanglement rates (Barlow et al. 1997). The offshore drift gillnet fishery is the only fishery that is likely to "take" sei whales from this stock, but no fishery mortalities or serious injuries to sei whales have been observed. Sei whales, like other large whales, may break through or carry away fishing gear. Whales carrying gear may

die later, become debilitated or seriously injured, or have normal functions impaired, but with no evidence recorded.

Sei whales are occasionally killed in collisions with vessels. One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Waring et al. 2009). From 1986-2019 there has been one confirmed vessel collision with a sei whale off California.

Sei whales are known to accumulate DDT, DDE, and PCBs (Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring.

Status and Trends

The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973.

Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000 to 38,000 whales by 1967, and reduced again to 20,600 to 23,700 whales by 1973. From 1910 to 1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Harwood and Hembree. 1987; Perry et al. 1999). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales: 300 to 600 sei whales were killed per year from 1911 to 1955. The sei whale catch peaked in 1959, when 1,340 sei whales were killed. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters. Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968 to 1969, after which the sei whale population declined rapidly (Mizroch et al. 1984). When commercial whaling for sei whales ended in 1974, the population in the North Pacific had been reduced to 7,260 to 12,620 animals (Tillman 1977). There have been no direct estimates of sei whale populations for the eastern Pacific Ocean (or the entire Pacific). Between 1991 and 2001, during aerial surveys, there were two confirmed sightings of sei whales along the U.S. Pacific coast.

Population growth rates for sei whales are not available at this time as there are little to no systematic survey efforts to study sei whales.

Diving and Social Behavior

Generally, sei whales make 5 to 20 shallow dives of 20 to 30 sec duration followed by a deep dive of up to 15 min (Gambell 1985b). The depths of sei whale dives have not been studied; however the composition of their diet suggests that they do not perform dives in excess of 300 meters. Sei whales are usually found in small groups of up to 6 individuals, but they commonly form larger groupings when they are on feeding grounds (Gambell 1985b).

Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2007). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Mizroch et al. 1984; Rice 1977). Sei whales in the North Pacific feed on euphausiids

and copepods, which make up about 95 percent of their diets (Calkins 1986). The dominant food for sei whales off California during June-August is northern anchovy, while in September-October whales feed primarily on krill (Rice 1977). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollack, capelin, and Atka mackerel (Nemoto and Kawamura 1977). In the Southern Ocean, analysis of stomach contents indicates sei whales consume Calanus spp. and small-sized euphasiids with prey composition showing latitudinal trends (Kawamura 1974). Evidence indicates that sei whales in the Southern Hemisphere reduce direct interspecific competition with blue and fin whales by consuming a wider variety of prey and by arriving later to feeding grounds (Kirkwood 1992). Rice (1977) suggested that the diverse diet of sei whales may allow them greater opportunity to take advantage of variable prey resources, but may also increase their potential for competition with commercial fisheries.

Little is known about the actual social system of these animals. Groups of two to five individuals are typically observed, but sometimes thousands may gather if food is abundant. However, these large aggregations may not be dependent on food supply alone, as they often occur during times of migration. Norwegian workers call the times of great sei whale abundance "invasion years." During mating season, males and females may form a social unit, but strong data on this issue are lacking.

Life History

Sei whales can live, on average, between 50 and 70 years. They have a gestation period of ten to 12 months, and calves nurse for six to nine months. Sexual maturity is reached between six and 12 years of age with an average calving interval of two to three years. Sei whales mostly inhabit continental shelf and slope waters far from the coastline. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed on a range of prey types, including: plankton (copepods and krill) small schooling fishes, and cephalopods.

Vocalization and Hearing

Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100 to 600 Hz range with 1.5 second duration and tonal and upsweep calls in the 200 to 600 Hz range of one to three second durations (McDonald et al. 2005). Vocalizations from the North Atlantic Ocean consisted of paired sequences (0.5 to 0.8 seconds, separated by 0.4 to 1.0 seconds) of 10 to 20 short (4 milliseconds) frequency modulated sweeps between 1.5 to 3.5 kilohertz (Richardson et al. 1995b). Source levels of 189 ±5.8 dB re: 1 µPa-m have been established for sei whales in the northeastern Pacific Ocean (Weirathmueller 2013).

Direct studies of sei whale hearing have not been conducted, but it is assumed that they can hear the same frequencies that they produce (low) and are likely most sensitive to this frequency range (Ketten 1997). This suggests sei whales, like other baleen whales, are more likely to have their best hearing capacities at low frequencies, including frequencies lower than those of normal human hearing, rather than mid- to high-frequencies (Ketten 1997). In terms of functional

hearing capability, sei whales belong to the low-frequency group, which have a hearing range of 7 Hz to 35 kilohertz (NMFS 2018b).

Critical Habitat

NMFS has not designated critical habitat for sei whales.

6.2.6 Sperm Whale

The sperm whale is a widely distributed species found in all major oceans (Figure 42).

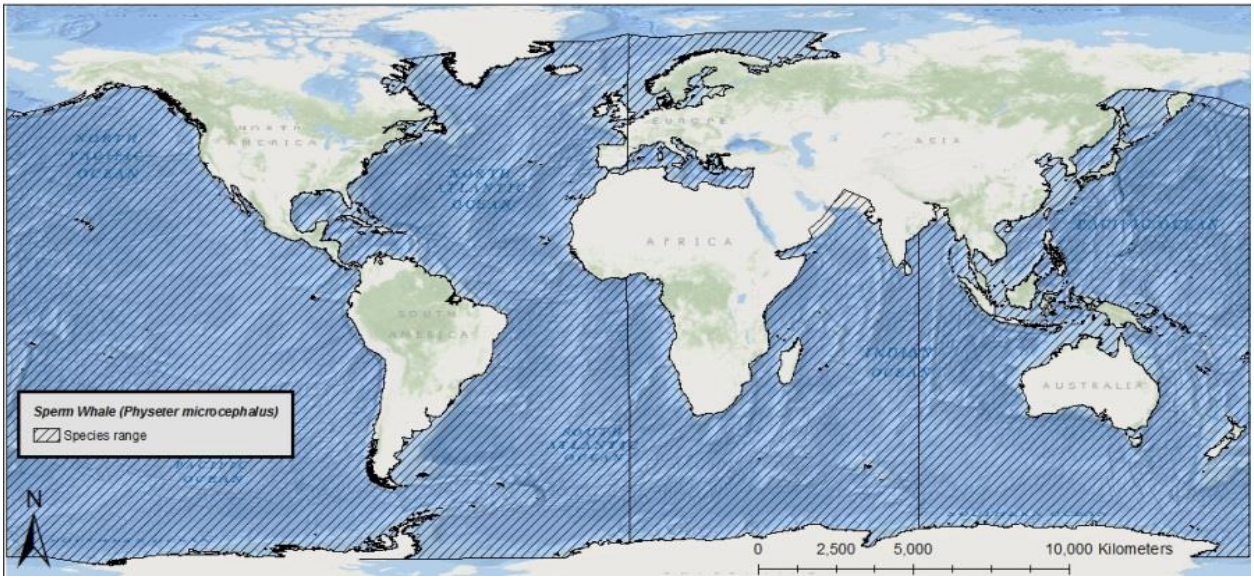


Figure 42. Map identifying the range of the endangered sperm whale.

Sperm whales are the largest toothed whale and distinguishable from other whales by its extremely large head, which takes up to 25 to 35 percent of its total body length and a single blowhole asymmetrically situated on the left side of the head near the tip. The sperm whale was originally listed as endangered on December 2, 1970.

Information available from the recovery plan (NMFS 2010a), recent SARs (Carretta 2019), and status review (NMFS 2015f) were used to summarize the life history, population dynamics, and status of the species as follows.

Distribution

Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring 1993) where adult males join them to breed.

Occurrence in the Offshore Portion of the NWTT action area

No sperm whales were detected during systematic surveys of waters between the British Columbia border with Alaska and Washington (Williams and Thomas 2007). In aerial surveys conducted in waters off Washington, Oregon, and Northern California covering the approximate nearshore half of the action area in the spring, summer, and fall of 2011 and 2012, sperm whales were encountered only twice, in deep water off the coast from Grays Harbor (Adams et al. 2014b). During the NMFS 2014 summer shipboard survey in the action area, there were a total of five sperm whale sightings (Barlow 2016). The variable presence of sperm whales in the area is reflected in the acoustic monitoring record of sperm whale click detections. In 2008, sperm whales were present in the acoustic record between April through November and in the following year from February through May (Oleson and Hildebrand 2012). In similar acoustic monitoring efforts between 2010 and 2013, sperm whales were found to be present from November through June (Debich et al. 2014; Kerosky et al. 2013; Klinck et al. 2015; Širović et al. 2012).

Occurrence in the Inland Waters of the NWTT action area

Given their documented preference for deep offshore waters, sperm whales are unlikely to occur within the Inland Waters portion of the action area.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Given their documented preference for deep offshore waters, sperm whales are unlikely to occur within the SEAFAC region of the action area since it is characterized by coastal waters removed from the continental shelf break.

Population Structure

Ocean-wide genetic studies indicate sperm whales have low genetic diversity, suggesting a recent bottleneck, but strong differentiation between matrilineally related groups (Lyrholm and Gyllensten 1998). Consistent with this, two studies of sperm whales in the Pacific Ocean indicate low genetic diversity (Mesnick et al. 2011; Rendell et al. 2012). As none of the stocks for which data are available have high levels of genetic diversity, the species may be at some risk to inbreeding and 'Allee' effects, although the extent to which is currently unknown.

Sperm whales have a global distribution and can be found in relatively deep waters in all ocean basins. While both males and females can be found in latitudes less than 40°, only adult males venture into the higher latitudes near the poles. Sperm whale distribute widely throughout the North Pacific Ocean, with movements over 5,000 kilometers, likely driven by changes in prey abundance. Males appear to range more broadly than females (Mizroch and Rice 2013).

Sperm whales are seasonal migrants to waters off the coast of Washington and Oregon where their densities are highest during spring and summer; they do not appear to occur in these waters during the winter. Sperm whales also tend to occur in the deeper water at the western edge of the

action area. In surveys of waters off Oregon and Washington conducted by Green et al. (1992), no sperm whales were encountered in waters less than 200 meters deep, 12 percent of the sperm whales were encountered in waters 200 to 2,000 meters deep (the continental slope), and the remaining 88 percent of the sperm whales were encountered in waters greater than 2,000 meters deep. In surveys conducted by Forney and her co-workers (Forney 2007), sperm whales were reported from the Olympic Coast Slope transects (west of the Olympic Coast National Marine Sanctuary), but not from surveys conducted over the National Marine Sanctuary or the area immediately west of Cape Flattery.

In the Gulf of Alaska, sperm whales have been sighted along the Aleutian Trench as well as over deeper waters and have been detected acoustically throughout the year (Forney and Brownell Jr. 1996; Mellinger et al. 2004a). Occurrence is higher from July through September than January through March (Mellinger et al. 2004a; Moore et al. 2006). The vast majority of individuals in the region are likely male based upon whaling records and genetic studies; the area is a summer foraging area for these individuals (Allen and Angliss 2010; Reeves et al. 1985; Straley and O'Connell 2005; Straley et al. 2005). Mean group size has been reported to be 1.2 individuals (Wade et al. 2003; Waite 2003). However, female groups may rarely occur at least up to the central Aleutian Islands (Fearnbach et al. 2012).

Abundance Estimate

The sperm whale is the most abundant of the large whale species, with a global population of between 300,000 and 450,000 individuals (Whitehead 2009). The higher estimates may be approaching population sizes prior to commercial whaling. In the northeast Pacific Ocean, the abundance of sperm whales was estimated to be between 26,300 and 32,100 in 1997 (NMFS 2015b). Population estimates are also available for the California/Oregon/Washington stock, estimated to consist of 1,997 individuals (N_{min}=1,270) (Carretta et al. 2020). There are currently no reliable population abundance estimates for the North Pacific stock (Muto et al. 2020).

Natural Threats

Sperm whales are known to be occasionally predated upon by killer whales (Jefferson et al. 1991; Pitman et al. 2001) by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989; Weller et al. 1996; Whitehead et al. 1997) and large sharks (Best et al. 1984) and harassed by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989; Weller et al. 1996; Whitehead et al. 1997). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (Goold et al. 2002; Wright 2005), direct widespread causes remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

Anthropogenic Threats

Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959 to 1983). However, other estimates have included 436,000 individuals killed between 1800 and 1987 (Carretta et al. 2005). However, all of these estimates are likely underestimates due to illegal killings and inaccurate reporting by Soviet whaling fleets between 1947 and 1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (Yablokov 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber 2004). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006).

In U.S. waters in the Pacific, sperm whales have been incidentally taken only in drift gillnet operations, which killed or seriously injured an average of nine sperm whales per year from 1991 to 1995 (Barlow et al. 1997).

Interactions between sperm whales and longline fisheries in the Gulf of Alaska have been reported since 1995 and are increasing in frequency (Hill and DeMaster 1998; Hill et al. 1999; Rice 1989). Between 2002 and 2006, there were three observed serious injuries (considered mortalities) to sperm whales in the Gulf of Alaska from the sablefish longline fishery (Angliss and Outlaw 2008). Sperm whales have also been observed in Gulf of Alaska feeding off longline gear (for sablefish and halibut) at 38 of the surveyed stations (Angliss and Outlaw 2008). Recent findings suggest sperm whales in Alaska may have learned that fishing vessel propeller cavitation (as gear is retrieved) are an indicator that longline gear with fish is present as a predation opportunity (Thode et al. 2007).

Contaminants have been identified in sperm whales, but vary widely in concentration based upon life history and geographic location, with northern hemisphere individuals generally carrying higher burdens (Evans et al. 2004). Contaminants include dieldrin, chlordane, DDT, DDE, PCBs, HCB and HCHs in a variety of body tissues (Aguilar 1983; Evans et al. 2004), as well as several heavy metals (Law et al. 1996). However, unlike other marine mammals, females appear to bioaccumulate toxins at greater levels than males, which may be related to possible dietary differences between females who remain at relatively low latitudes compared to more migratory males (Aguilar 1983; Wise et al. 2009). Chromium levels from sperm whales skin samples worldwide have varied from undetectable to 122.6 μg Cr/g tissue, with the mean (8.8 μg Cr/g

tissue) resembling levels found in human lung tissue with chromium-induced cancer (Wise et al. 2009). Older or larger individuals did not appear to accumulate chromium at higher levels.

Status and Trends

The sperm whale is endangered as a result of past commercial whaling. According to historical whaling records from five whaling stations in British Columbia, 6,158 sperm whales were killed between 1908 and 1967 (Gregr et al. 2000). Although the aggregate abundance worldwide is probably at least several hundred thousand individuals, the extent of depletion and degree of recovery of populations are uncertain. Commercial whaling is no longer allowed, however, illegal hunting may occur at biologically unsustainable levels. Continued threats to sperm whale populations include ship strikes, entanglement in fishing gear, competition for resources due to overfishing, population, loss of prey and habitat due to climate change, and noise. The species' large population size shows that it is somewhat resilient to current threats. There is insufficient data to evaluate trends in abundance and growth rates of sperm whales at this time.

Diving and Social Behavior

Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km down and durations in excess of 2 hours (Clarke 1976; Watkins 1985; Watkins et al. 1993). However, dives are generally shorter (25 to 45 min) and shallower (400 to 1,000 meters). Dives are separated by 8 to 11 min rests at the surface (Gordon 1987; Jochens et al. 2006; Papastavrou et al. 1989; Watwood et al. 2006). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003a). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Unlike other cetaceans, there is a preponderance of dive information for this species, most likely because it is the deepest diver of all cetacean species so generates a lot of interest. Sperm whales feed on large and medium-sized squid, octopus, rays and sharks, on or near the ocean floor (Clarke 1986; Whitehead 2002). Some evidence suggests that they do not always dive to the bottom of the sea floor (likely if food is elsewhere in the water column), but that they do generally feed at the bottom of the dive. Davis et al. (2007) report that dive-depths (100 to 500 meters) of sperm whales in the Gulf of California overlapped with depth distributions (200 to 400 meters) of jumbo squid, based on data from satellite-linked dive recorders placed on both species, particularly during daytime hours. Their research also showed that sperm whales foraged throughout a 24-hour period, and that they rarely dove to the sea floor bottom (>1000 meters). The most consistent sperm whale dive type is U-shaped, during which the whale makes a rapid descent to the bottom of the dive, forages at various velocities while at depth (likely while chasing prey) and then ascends rapidly to the surface. There is some evidence that male sperm whales, feeding at higher latitudes during summer months, may forage at several depths

including <200 m, and utilize different strategies depending on position in the water column (Teloni et al. 2007).

Movement patterns of Pacific female and immature male groups appear to follow prey distribution and, although not random, movements are difficult to anticipate and are likely associated with feeding success, perception of the environment, and memory of optimal foraging areas (Whitehead 2008). However, no sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred miles are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several days. However, general transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

Gaskin (1973) proposed a northward population shift of sperm whales off New Zealand in the austral autumn based on reduction of available food species and probable temperature tolerances of calves.

Sperm whales have a strong preference for waters deeper than 1,000 meters (Reeves and Whitehead 1997; Watkins and Schevill 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 meters in depth (Clarke 1956; Rice 1989). Sperm whales have been observed near Long Island, New York, in water between 40 and 55 meters deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf.

Sperm whales are frequently found in locations of high productivity due to upwelling or steep underwater topography, such as continental slopes, seamounts, or canyon features (Jaquet 1996; Jaquet and Whitehead 1996). Cold-core eddy features are also attractive to sperm whales in the Gulf of Mexico, likely because of the large numbers of squid that are drawn to the high concentrations of plankton associated with these features (Biggs et al. 2000; Davis et al. 2000; Davis et al. 2002). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet and Whitehead 1996; Waring et al. 1993). Sperm whales over George's Bank were associated with surface temperatures of 23.2 to 24.9°C (Waring et al. 2004).

Local information is inconsistent regarding sperm whale tendencies. Gregr and Trites (2001) reported that female sperm whales off British Columbia were relatively unaffected by the

surrounding oceanography. However, Tynan et al. (2005) reported increased sperm whales densities with strong turbulence associated topographic features along the continental slope near Heceta Bank. Two noteworthy strandings in the region include an infamous incident (well publicized by the media) of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. In addition, a mass stranding of 47 individuals in Oregon occurred during June 1979 (Norman et al. 2004; Rice et al. 1986).

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6 to 12 individuals) versus the Pacific (25 to 30 individuals) (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

Life History

The average lifespan of sperm whales is estimated to be at least 50 years (Whitehead 2009). They have a gestation period of one to one and a half years, and calves nurse for approximately two years. Sexual maturity for sperm whales in the North Pacific is reached between 7 and 13 years of age for females with an average calving interval for four to six years. Male sperm whales reach full sexual maturity between ages 18 and 21, after which they undergo a second growth spurt, reaching full physical maturity at around age 40 (Mizroch and Rice 2013). Data from historical whaling station records from 1908 to 1967 indicate that sperm whales mated in April through June, and calved in July to August in the offshore waters of British Columbia (Gregr et al. 2000). Sperm whales mostly occur far offshore, inhabiting areas with a water depth of 600 meters (1,968 feet) or more, and are uncommon in waters less than 300 meters (984 feet) deep. However, if there are shelf breaks or submarine canyons close to land, sperm whales can occur there. They winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed primarily on squid; other prey includes octopus and demersal fish (including teleosts and elasmobranchs). An analysis of commercial whaling records from the Coal Harbor whaling station in northern Vancouver from 1963 to 1967 looked at sperm whale stomach contents. The samples came late spring through summer (April through September). North Pacific giant squid (*Moroteuhis robusta*) was the most abundant prey item for both males and females, but the secondary prey item differed between sexes. After giant squid, males consumed rockfish (*Sebastes spp*.), while females ate ragfish (*Icosteus spp*.) and other fish (Flinn et al. 2002).

Vocalization and Hearing

Sound production and reception by sperm whales are better understood than in most cetaceans. Recordings of sperm whale vocalizations reveal that they produce a variety of sounds, such as clicks, gunshots, chirps, creaks, short trumpets, pips, squeals, and clangs (Goold 1999). Sperm whales typically produce short duration repetitive broadband clicks with frequencies below 100 Hz to greater than 30 kilohertz (Watkins 1977) and dominant frequencies between 1 to 6 kilohertz and 10 to 16 kilohertz. Another class of sound, "squeals," are produced with frequencies of 100 Hz to 20 kilohertz (e.g., Weir et al. 2007). The source levels of clicks can reach 236 dB re: 1 µPa-m, although lower source level energy has been suggested at around 171 dB re: 1 µPa-m (Goold and Jones 1995; Mohl et al. 2003; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). Most of the energy in sperm whale clicks is concentrated at around 2 to 4 kilohertz and 10 to 16 kilohertz (Goold and Jones 1995; Weilgart and Whitehead 1993). The clicks of neonate sperm whales are very different from typical clicks of adults in that they are of low directionality, long duration, and low frequency (between 300 Hz and 1.7 kilohertz) with estimated source levels between 140 to 162 dB re: 1 µPa-m (Madsen et al. 2003). The highly asymmetric head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Norris and Harvey 1972).

Long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Miller et al. 2004; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997; Whitehead and Weilgart 1991). Creaks (rapid sets of clicks) are heard most frequently when sperm whales are foraging and engaged in the deepest portion of their dives, with inter-click intervals and source levels being altered during these behaviors (Laplanche et al. 2005; Miller et al. 2004). Clicks are also used during social behavior and intragroup interactions (Weilgart and Whitehead 1993). When sperm whales are socializing, they tend to repeat series of group-distinctive clicks (codas), which follow a precise rhythm and may last for hours (Watkins and Schevill 1977). Codas are shared between individuals in a social unit and are considered to be primarily for intragroup communication (Rendell and Whitehead 2004; Weilgart and Whitehead 1997). Research in the South Pacific Ocean suggests that in breeding areas the majority of codas are produced by mature females (Marcoux et al. 2006). Coda repertoires have also been found to vary geographically and are categorized as dialects (Pavan et al. 2000; Weilgart and Whitehead 1997). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean Sea and those in the Pacific Ocean (Weilgart and Whitehead 1997). Three coda types used by male sperm whales have recently been described from data collected over multiple years: these codas are associated with dive cycles, socializing, and alarm (Frantzis and Alexiadou 2008).

Our understanding of sperm whale hearing stems largely from the sounds they produce. The only direct measurement of hearing was from a young stranded individual from which auditory evoked potentials were recorded (Carder and Ridgway 1990). From this whale, responses support a hearing range of 2.5 to 60 kilohertz and highest sensitivity to frequencies between 5 to

20 kilohertz. Other hearing information consists of indirect data. For example, the anatomy of the sperm whale's inner and middle ear indicates an ability to best hear high-frequency to ultrasonic hearing (Ketten 1992). The sperm whale may also possess better low-frequency hearing than other odontocetes, although not as low as many baleen whales (Ketten 1992). Reactions to anthropogenic sounds can provide indirect evidence of hearing capability, and several studies have made note of changes seen in sperm whale behavior in conjunction with these sounds. For example, sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins et al. 1985; Watkins and Schevill 1975). In the Caribbean Sea, Watkins et al. (1985) observed that sperm whales exposed to 3.25 to 8.4 kilohertz pulses (presumed to be from submarine sonar) interrupted their activities and left the area. Similar reactions were observed from artificial sound generated by banging on a boat hull (Watkins et al. 1985). André et al. (1997) reported that foraging whales exposed to a 10 kilohertz pulsed signal did not ultimately exhibit any general avoidance reactions: when resting at the surface in a compact group, sperm whales initially reacted strongly, and then ignored the signal completely. Thode et al. (2007) observed that the acoustic signal from the cavitation of a fishing vessel's propeller (110 dB re: 1 μPa^2-s between 250 Hz and 1 kilohertz) interrupted sperm whale acoustic activity and resulted in the animals converging on the vessel. Sperm whales have also been observed to stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995). Because they spend large amounts of time at depth and use low frequency sound, sperm whales are likely to be susceptible to low frequency sound in the ocean (Croll et al. 1999b). Nonetheless, sperm whales are considered to be part of the mid-frequency marine mammal hearing group, with a hearing range between 150 Hz and 160 kilohertz (NMFS 2018b).

Critical Habitat

NMFS has not designated critical habitat for sperm whales.

6.2.7 Guadalupe Fur Seal

Guadalupe fur seals are medium sized, sexually dimorphic otariids that are generally asocial with their conspecifics and other species (Belcher and T.E. Lee 2002; Reeves et al. 2002). Except for adult males, members of this species resemble California sea lions and northern fur seals. Distinguishing characteristics of the Guadalupe fur seal include the digits on their hind flippers (all of similar length), large, long foreflippers, unique vocalizations, and a characteristic behavior of floating vertically with their heads down in the water and their hind flippers exposed for cooling (Reeves et al. 2002).

Distribution

Guadalupe fur seals' historic range included the Gulf of Farallones, California to the Revillagigedo Islands, Mexico (Belcher and T.E. Lee 2002; Rick et al. 2009). Currently, they breed mainly on Guadalupe Island, Mexico, 155 miles off of the Pacific Coast of Baja California.

A smaller breeding colony, discovered in 1997, appears to have been established at Isla Benito del Este, Baja California, Mexico (Belcher and T.E. Lee 2002) (Figure 43).

There are reports of individuals being sighted in the California Channel Islands, Farallone Islands, Monterey Bay, and other areas of coastal California and Mexico (Belcher and T.E. Lee 2002; Carretta et al. 2002; Reeves et al. 2002). A single female gave birth to a pup on the Channel Islands in 1997. In recent years, a small number of pups (less than 30 per year) have been born at San Benito Archipelago (reviewed in McCue et al. 2020). Guadalupe fur seals are known to travel great distances, with sightings occurring thousands of kilometers away from the main breeding colonies (Aurioles-Gamboa et al. 1999). Their presence along the U.S. west coast has increased; sightings have occurred from central Mexico to southern British Columbia, Canada and rarely in Alaska (reviewed in McCue et al. 2020).

The Guadalupe fur seal population is slowly recovering from the brink of extinction. The current population abundance is approximately 31,000 animals. Of all the fur seal species, this one is the least studied due to their limited geographic locations. The Guadalupe fur seal population does appear to be increasing annually.

Before intensive hunting decreased their numbers, Guadalupe fur seals ranged from Monterey Bay, California, to the Revillagigedo Islands, Mexico (Aurioles-Gamboa and Camacho-Rios 2007), but have occasionally been identified from strandings (Northwest Region Stranding Database; Wilkinson 2013) or in archaeological contexts as far north as northern California, Oregon, and Washington (Etnier 2002; Rick et al. 2009). As described in Section 7.4, Guadalupe fur seals have experienced multiple unusual mortality events (UMEs) since 1989, one of which is still ongoing.

Sightings of live animals off Washington and Oregon are more limited, although there is photo documentation of apparently healthy Guadalupe fur seals in offshore waters of Washington and British Columbia in recent years during summer and early autumn (Lambourn et al. 2012). Of the 35 adult and juvenile Guadalupe fur seals that were tagged in 2018 at Guadalupe Island in support of marine mammal monitoring efforts in Navy training and testing areas in the North Pacific, one adult female was tracked in the southernmost region of the NWTT action area, 250 to 350 kilometers offshore of northern California (Norris and Elorriaga-Verplancken 2019). Given the increased number of strandings in the Pacific Northwest, coupled with their increasing population, it is possible that Guadalupe fur seals are returning to their historic pelagic migration range suggested by the archaeological findings (Etnier 2002; Lambourn et al. 2012; Rick et al. 2009).



Figure 43. Guadalupe fur seal historic range.

Occurrence in the Offshore Portion of the NWTT action area

Based on their rookeries occurring in Baja California, Mexico, the species is predominantly distributed off Mexico, but with annual strandings in Oregon and Washington, Guadalupe fur seals are considered "seasonal" migrants within the offshore portion of the action area. Given the lack of at-sea sightings by the National Marine Fisheries Service and their documented coastal strandings (Lambourn et al. 2012), it would be anticipated Guadalupe fur seals would be more coastal and near-shore in distribution in the action area as compared to other more pelagic pinnipeds such as northern fur seals.

Occurrence in the Inland Waters of the NWTT action area

Guadalupe fur seals are not expected to occur within the Inland Waters portion of the action area. Strandings from the offshore portion of the action area have been documented as noted above, but they are considered extralimital in the inland waters.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Guadalupe fur seals are not expected to occur in the SEAFAC portion of the action area.

Population Structure

All Guadalupe fur seals represent a single population, with two known breeding colonies in Mexico, and a purported breeding colony in the United States. Gallo-Reynoso (1994) calculated that the population of Guadalupe fur seals in Mexico from thirty years of population and counts and concluded the population was increasing; with an average annual growth rate of 13.3 percent on Guadalupe Island. The 2000 NMFS SAR for Guadalupe fur seals also indicated the breeding colonies in Mexico were increasing; and more recent evidence indicates that this trend is continuing (Aurioles-Gamboa et al. 2010; Esperon-Rodriguez and Gallo-Reynoso 2012). From 1984 to 2013 at Guadalupe Island, the Guadalupe fur seal population increased at an average annual growth rate of 5.9 percent (range 4.1 to 7.7 percent) (García-Aguilar et al. 2018). Other estimates of the Guadalupe fur seal population of the San Benito Archipelago (from 1997-2007) indicate that it is increasing as well at an annual rate of 21.6 percent (Esperon-Rodriguez and Gallo-Reynoso 2012), and that this population is at a phase of exponential increase (Aurioles-Gamboa et al. 2010). However, these estimates are considered too high, and likely result from immigration at Guadalupe Island (Carretta 2019). Based on direct counts of animals from 1955 and 1993, the estimated annual population growth rate is 13.7 percent (Carretta 2019).

Abundance Estimate

It is difficult to obtain an accurate abundance estimate of Guadalupe fur seals due in part to their tendency to stay in caves and remain at sea for extended lengths of time, making them unavailable for counting. At the time of listing in 1985, the population was estimated at 1,600 individuals, compared to approximately 30,000 before hunting occurred in the 18th and 19th centuries. A population was "rediscovered" in 1928 with the capture of two males on Guadalupe Island; from 1949 on, researchers reported sighting Guadalupe fur seals at Isla Cedros (near the San Benito Archipelago), and Guadalupe Island (Bartholomew Jr. 1950; Peterson et al. 1968). In 1994, the population at Guadalupe Island was estimated at 7,408 individuals (Gallo-Reynoso 1994). There have been other, more recent population abundance estimates for Guadalupe Island, with a considerable amount of variation between them: 20,000 in 2010 (García-Capitanachi et al. 2017), and between 34,000 and 44,000 in 2013 (García-Aguilar et al. 2018). Guadalupe fur seals are also found on San Benito Island, likely immigrants from Guadalupe Island, as there are relatively few pups born on San Benito Island (Aurioles-Gamboa et al. 2010). There were an estimated 2,504 seals on San Benito Island in 2010 (García-Capitanachi et al. 2017). Based on information presented by (García-Aguilar et al. 2018), and using a population size to pup count ratio of 3.5, the minimum population estimate is 31,019 (Carretta et al. 2020).

Natural Threats

Guadalupe fur seals are known to be preyed on by sharks and killer whales (Belcher and Lee 2002; Jefferson et al. 2008).

Anthropogenic Threats

Although a number of human activities may have contributed to the current status of this species, historic commercial hunting was likely the most devastating. Even with population surveys occurring on an irregular basis in subsequent years, these surveys provide evidence that the Guadalupe fur seal has been increasing after suffering such a significant decline. Although commercial hunting occurred in the past, and has since ceased, the effects of these types of exploitations persist today. Other human activities, such as entanglements from commercial fishing gear, are ongoing and continue to affect these species.

Status and Trends

Commercial sealers in the 19th century decimated the Guadalupe fur seal population, taking as many 8,300 fur seals from San Benito Island (Townsend 1924). Numbers on the total number of fur seals harvested are difficult to ascertain because of the difficulty the hunters had in distinguishing species while hunting (Seagars 1984). These harvests were devastating for the Guadalupe fur seal population, so much so that in 1892, only seven individuals were observed on Guadalupe Island, the location of one of the larger known breeding colonies (Bartholomew Jr. 1950); two years later, a commercial sealer took all 15 remaining individuals that could be found (Townsend 1899).

The species was presumed extinct, until 1926, when a small herd was found on Guadalupe Island by commercial fishermen, who later returned and killed all the seals they could find. In 1928, the Mexican government declared Guadalupe Island as a pinniped sanctuary. In 1954, during a survey of the island, Hubbs (1956) discovered at least 14 individuals. The government of Mexico banned the hunting of Guadalupe fur seals in 1967. Although population surveys occurred on an irregular basis in subsequent years, evidence shows that the Guadalupe fur seal population has been increasing ever since.

The Guadalupe fur seal clearly experienced a precipitous decline due to commercial exploitation, and may have undergone a population bottleneck. Bernardi et al. (1998) compared the genetic divergence in the nuclear fingerprint of samples taken from 29 Guadalupe fur seals, and found an average similarity of 0.59 of the DNA profiles. This average is typical of outbreeding populations. When comparing the amount of unique character fragments found in Guadalupe fur seals to that of other pinnipeds that have experienced bottlenecks (e.g., Hawaiian monk seals), that amount is much higher (0.14 vs. 0.05) in Guadalupe fur seals than Hawaiian monk seals. By using mitochondrial DNA sequence analysis in comparing the genetic diversity of Guadalupe fur seals to northern elephant seals (which did experience a severe bottleneck), Guadalupe fur seals had more haplotypes and a higher number of variable sites. The authors hypothesized that the

numbers of Guadalupe fur seals left after harvest may have been underestimated, and the population may not have actually experienced a bottleneck, or the bottleneck may have been of short duration and not severe enough to suppress genetic diversity. Although the relatively high levels of genetic variability are encouraging, it is important to note that commercial harvest still influenced the population. Later studies comparing mt DNA found in the bones of pre-exploitation Guadalupe fur seals against the extant population showed a loss of genotypes, with twenty-five genotypes in pre-harvest fur seals, and seven present today (Weber et al. 2004).

The population has also been influenced by factors leading to strandings and unusual mortality events (see Section 7.4). Of the 169 documented strandings in Washington and Oregon from 2005 through 2016, 139 were yearlings. Strandings were highly seasonal, with most occurring in June consistently throughout the years examined. The three major causes of death could be categorized as emaciation, trauma (fishery-related, blunt force, bullet wounds, and shark attack), and infectious disease from coccidian parasites, including *Toxoplasma gondii* and *Sarcocystis neurona*. These increased strandings may be resulting from increased use of these coastal habitats by a population of Guadalupe fur seals that is reaching a healthy size (D'Agnese et al. 2020).

While some incidental breeding takes place on the San Benito Islands and the Channel Islands, the Guadalupe Island breeding colony supports the population (García-Aguilar et al. 2018). The current abundance of the Guadalupe fur seal represents about one-fifth of the estimated historical population size, and although the population has continued to increase, the species has not expanded its breeding range, potentially affecting its recovery (García-Aguilar et al. 2018). Because that over the last fifty years the population has been increasing since being severely depleted, we believe that the Guadalupe fur seal population is resilient to future perturbations.

Life History

Guadalupe fur seals prefer rocky habitats and can be found in natural recesses and caves (Fleischer 1978), using sheltered beaches and rocky platforms for breeding (Arias-del-Razo et al. 2016). Breeding occurs in June through August. Adult males return to the colonies in early June. Female Guadalupe fur seals arrive on beaches in June, with births occurring between mid-June to July (Pierson 1978); the pupping season is generally over by late July (Fleischer 1978). Breeding adult males are polygamous, and may mate with up to 12 females during a single breeding season. Females stay with pups for seven to eight days after parturition, and then alternate between foraging trips at sea and lactation on shore; nursing lasts about eight months (Figureroa-Carranza 1994). Guadalupe fur seals feed mainly on squid species (Esperon-Rodriguez and Gallo-Reynoso 2013); the Gulf of Ulloa on the Pacific side of the Baja California peninsula is an important feeding area (Aurioles-Gamboa and Szteren 2019). Based on a stable isotope analysis of male Guadalupe fur seal carcasses, there appears to be some niche segregation between coastal and oceanic males, possibly based on individual age and size (Aurioles-Gamboa and Szteren 2019). Foraging trips can last between four to twenty-four days (average of fourteen

days). Tracking data show that adult females spend seventy-five percent of their time sea, and twenty-five percent at rest (Gallo-Reynoso et al. 1995).

Vocalization and Hearing

Pinnipeds produce sounds both in air and water that range in frequency from approximately 100 Hz to several tens of kilohertz and it is believed that these sounds serve social functions such as mother-pup recognition and reproduction. Source levels for pinniped vocalizations range from approximately 95–190 dB re 1 μPa (Richardson et al. 1995b).

Underwater hearing in otariid seals is adapted to low frequency sound and less auditory bandwidth than phocid seals. Hearing in otariid seals has been tested in California sea lions (Kastak and Schusterman 1998) and northern fur seals (Babushina et al. 1991; Moore and Schusterman 1987). Based on these studies, Guadalupe fur seals would be expected to hear sounds within the ranges of 50 Hz–75 kilohertz in air and 50 Hz–50 kilohertz in water.

Critical Habitat

NMFS has not designated critical habitat for Guadalupe fur seals.

6.2.8 Leatherback Sea Turtle

The leatherback sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide (Figure 44). Because only leatherbacks originating from the Western Pacific nesting beaches may be found in the action area, this biological opinion will focus on the effects of the proposed action on the West Pacific population.

Leatherbacks are the largest living turtle, reaching lengths of six feet long, and weighing up to one ton. Leatherback sea turtles have a distinct black leathery skin covering their carapace with pinkish white skin on their belly. The species was first listed under the Endangered Species Conservation Act (35 FR 8491) and listed as endangered under the ESA since 1973.

Life History

Leatherback age at maturity has been difficult to ascertain, with estimates ranging from five to twenty-nine years (Avens et al. 2009; Spotila et al. 1996). Females lay up to seven clutches per season, with more than sixty-five eggs per clutch and eggs weighing greater than eighty grams (Reina et al. 2002; Wallace et al. 2007). Leatherback sex determination is affected by nest temperature, with higher temperatures producing a greater proportion of females (Mrosovsky 1994; Witzell et al. 2005). A significant female bias has been reported in several leatherback populations (Binckley et al. 1998; James et al. 2007; Plotkin 1995). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50 percent worldwide (Eckert et al. 2012). Females nest every one to seven years. Natal homing, at least within an ocean basin, results in reproductive isolation between five broad geographic regions: eastern and western Pacific, eastern and western Atlantic, and Indian Ocean.

Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight. Leatherbacks weigh about 33 percent more on their foraging grounds than at nesting, indicating that they probably catabolize fat reserves to fuel migration and subsequent reproduction (James et al. 2005; Wallace et al. 2006). Sea turtles must meet an energy threshold before returning to nesting beaches. Therefore, their remigration intervals (the time between nesting) are dependent upon foraging success and duration (Hays 2000; Price et al. 2004).

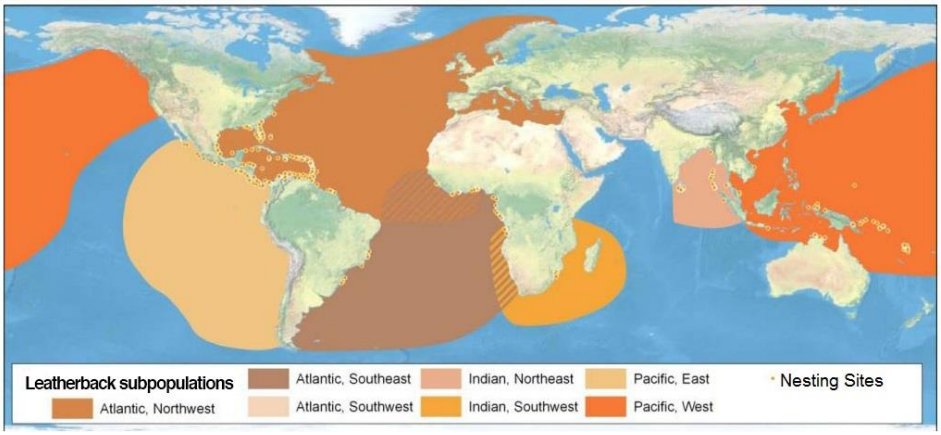


Figure 44. Map identifying the range of the endangered leatherback sea turtle. From NMFS http://www.nmfs.noaa.gov/pr/species/turtles/leatherback.html, adapted from (Wallace et al. 2010).

Diving

The leatherback sea turtle is one of the deepest divers in the ocean, with dives as deep as 3,937 feet (1,200 meters), although it spends most of its time feeding at a depth of less than 328 feet (100 meters). Leatherback turtles primarily feed on gelatinous zooplankton such as cnidarians (jellyfish and siphonophores) and tunicates (salps and pyrosomas) (Bjorndal 1997; NMFS and USFWS 1998b). The leatherback dives continually and spends short periods of time on the surface between dives (Eckert et al. 1989; Southwood et al. 1999). Typical dive durations averaged 6.9 to 14.5 min per dive, with a maximum of 42 min (Eckert et al. 1996). Sea turtles typically remain submerged for several minutes to several hours depending upon their activity state (Standora et al. 1984). Long periods of submergence hamper detection and confound census efforts. During migrations or long distance movements, leatherbacks maximize swimming efficiency by traveling within 15 feet (5 meters) of the surface (Eckert 2002).

Nesting Social Behavior

Male leatherbacks do not return to land after they hatch from their nests whereas mature females return to land only to lay eggs (Spotila 2004). Aside from this brief terrestrial period, which lasts

approximately two to three months during egg incubation and hatching, leatherback turtles are rarely encountered out of the water. Hatchling leatherbacks are pelagic, but nothing is known about their distribution during the first 4 years of life (Musick and Limpus 1997).

Vocalization and Hearing

Sea turtles do not have an external ear pinnae or eardrum. Instead, they have a cutaneous layer and underlying subcutaneous fatty layer that function as a tympanic membrane. The subcutaneous fatty layer receives and transmits sounds to the middle ear and into the cavity of the inner ear (Ridgway et al. 1969). Sound also arrives by bone conduction through the skull. Sound arriving at the inner ear via the columella (homologous to the mammalian stapes or stirrup) is transduced by the bones of the middle ear.

Sea turtle auditory sensitivity is not well studied, though a few preliminary investigations suggest that it is limited to low frequency bandwidths, such as the sounds of waves breaking on a beach. The role of underwater low-frequency hearing in sea turtles is unclear. It has been suggested that sea turtles may use acoustic signals from their environment as guideposts during migration and as a cue to identify their natal beaches (Lenhardt et al. 1983).

Lenhardt et al. (1983) applied audio frequency vibrations at 250 Hz and 500 Hz to the heads of loggerheads and Kemp's ridleys submerged in salt water to observe their behavior, measure the attenuation of the vibrations, and assess any neural-evoked response. These stimuli (250 Hz, 500 Hz) were chosen as representative of the lowest sensitivity area of marine turtle hearing (Wever and Vernon 1956). At the maximum upper limit of the vibratory delivery system, the sea turtles exhibited abrupt movements, slight retraction of the head, and extension of the limbs in the process of swimming. Lenhardt et al. (1983) concluded that bone-conducted hearing appears to be a reception mechanism for at least some of the sea turtle species, with the skull and shell acting as receiving surfaces. Sensitivity even within the optimal hearing range was low as threshold detection levels in water are relatively high at 160 to 200 dB re 1 µPa-m, which is the standard reference measure for underwater sound energy in this regard (Lenhardt et al. 1994). Piniak (2012) measured hearing of leatherback turtle hatchlings in water and in air, and observed reactions to low frequency sounds, with responses to stimuli occurring between 50 Hz and 1.6 kilohertz in air between 50 Hz and 1.2 kilohertz in water (lowest sensitivity recorded was 93 dB re: 1 µPa at 300 Hz).

Ridgway et al. (1969) used aerial and mechanical stimulation to measure the cochlea in three specimens of green turtle, and concluded that they have a useful hearing span of perhaps 60 to 1,000 Hz, but hear best from about 200 Hz up to 700 Hz, with their sensitivity falling off considerably below 200 Hz. The maximum sensitivity for one animal was at 300 Hz, and for another was at 400 Hz. At the 400 Hz frequency, the green turtle's hearing threshold was about 64 dB in air (approximately 126 dB in water). At 70 Hz, it was about 70 dB in air (approximately 132 dB in water). No audiometric data are available for the leatherback turtle, but based on other sea turtle hearing capabilities, they probably also hear best in the low frequencies.

For exposures to impulsive sound, a recent study on the effects of air guns on sea turtle behavior also suggests that sea turtles are most likely to respond to low-frequency sounds (McCauley et al. 2000b). Loggerhead sea turtles will avoid air-gun arrays at 2 km and at 1 km, with received levels of 166 dB re 1 µPa-m and 175 dB re 1 µPa, respectively (McCauley et al. 2000b). The sea turtles' response was consistent: above a level of about 166 dB re 1 µPa, the sea turtles noticeably increased their swimming activity. Above 175 dB re 1 µPa, their behavior became more erratic, possibly indicating that they were agitated (McCauley et al. 2000b).

Currently it is believed that the range of maximum sensitivity for sea turtles is 200 to 800 Hz, with an upper limit of about 2,000 Hz (Lenhardt 1994; Moein et al. 1994). Green turtles are most sensitive to sounds between 200 and 700 Hz, with peak sensitivity at 300 to 400 Hz (Ridgway et al. 1969). They possess an overall hearing range of approximately 60 to 1,000 Hz (Ridgway et al. 1969). Juvenile loggerhead turtles hear sounds between 250 and 1,000 Hz and, therefore, often avoid low-frequency sounds (Bartol et al. 1999). Finally, sensitivity even within the optimal hearing range is apparently low—threshold detection levels in water are relatively high at 160 to 200 dB re 1 µPa-m (Lenhardt 1994). Given the lack of audiometric information for leatherback turtles, the potential for TTS among leatherback turtles must be classified as unknown but would likely follow those of other sea turtles. In terms of sound emission, nesting leatherback turtles produce sounds in the 300 to 500 Hz range (Mrosovsky 1972).

Distribution

Leatherback sea turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico, Nicaragua, and Costa Rica (eastern Pacific) and Indonesia, the Solomon Islands, and Papua New Guinea (western Pacific). Genetic studies have been used to identify two discrete leatherback populations in the Pacific Ocean (Dutton 2006): an eastern Pacific Ocean population, which nests between Mexico and Ecuador; and a western Pacific Ocean population, which nests in numerous countries, including Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu. The West Pacific DPS nests throughout four countries with a broad, diverse foraging range. It exhibits metapopulation dynamics and fine-scale population structure (NMFS 2020d).

Leatherback sea turtles have been documented in Alaska waters as far north as approximately 60º latitude and as far west in the Gulf of Alaska as the Aleutian Islands (Eckert 1993). In contrast with other sea turtles, leatherback sea turtles have physiological traits that allow for the conservation of body heat which enable them to maintain body core temperatures well above the ambient water temperatures (Eckert 1993; Greer et al. 1973; Pritchard 1971). Carapaces of adult leatherbacks are 4 cm (1.5 inches) thick on average, contributing to the leatherback's thermal

tolerance that enables this species to forage in water temperatures far lower than the leatherback's core body temperature (Bostrom et al. 2010). In an analysis of available sightings (Eckert 2002), researchers found that leatherback turtles with carapace lengths smaller than 100 cm (39 in) were sighted only in waters 79 ºF or warmer, while adults were found in waters as cold as 32 ºF to 59 ºF off Newfoundland (Goff and Lien 1988). As a result, they are more capable of surviving for extended periods of time in cooler waters than the hard-shelled sea turtles (Bleakney 1965; Lazell Jr. 1980).

In the Pacific Ocean, leatherback turtles have the most extensive range of any living reptile and have been reported in all pelagic waters of the Pacific between 71° N and 47° S latitude and in all other major pelagic ocean habitats (NMFS and USFWS 1998b). Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Few quantitative data are available concerning the seasonality, abundance, or distribution of leatherbacks in the central northern Pacific Ocean. Satellite tracking studies and occasional incidental captures of the species in the Hawaii-based longline fishery indicate that deep ocean waters are the preferred habitats of leatherback turtles in the central Pacific Ocean (NMFS and USFWS 2007b). The primary migration corridors for leatherbacks are across the North Pacific Subtropical Gyre, with the eastward migration route possibly to the north of the westward migration.

Occurrence in the Offshore Portion of the NWTT Action Area

Any leatherback sea turtles found within the NWTT action area would be from the Western Pacific population. Leatherback turtles of the West Pacific DPS nest in tropical and subtropical latitudes primarily in Indonesia, Papua New Guinea, and Solomon Islands, and to a lesser extent in Vanuatu. Few quantitative data are available concerning the seasonality, abundance, or distribution of leatherbacks in the NWTT action area. The movements of adult leatherback sea turtles appear to be linked to the seasonal availability of their prey and the requirements of their reproductive cycles (Collard 1990; Davenport and Balazs 1991). Leatherbacks prefer convergence zones and upwelling areas in the open ocean, along continental margins, or near large archipelagos. Leatherbacks from both eastern (rare) and western (primary) Pacific Ocean nesting populations migrate to northern Pacific Ocean foraging grounds, where longline fisheries operate (Dutton et al. 1998). Leatherbacks from nesting beaches in the Indo-Pacific region have been tracked migrating thousands of km from nesting areas to summer foraging grounds off the coast of northern California (Benson et al. 2007), including a 6,385 mi. (10,276 km) migration from a nesting beach in Papua New Guinea to foraging grounds off the coast of Oregon (Benson et al. 2007). The waters off the Oregon and California coasts have been repeatedly recognized by scientists and agencies as comprising one of the most important leatherback foraging areas in the Pacific (NMFS and USFWS 1998b).

Leatherback turtles are regularly seen off the western coast of the United States. Off the California coast, the highest densities of leatherback sea turtles were found off central California

(Benson et al. 2007). Telemetry studies have shown areas of concentration along the central California coast and in the waters of Oregon and Washington (Benson et al. 2011). Stinson (1984) concluded that the leatherback was the most common sea turtle in U.S. waters north of Mexico. Aerial surveys off Washington, Oregon, and California indicate that most leatherbacks occur in waters over the continental slope, with a few over the continental shelf (Eckert 1993). Green et al. (1992) conducted a study between 1989 and 1990 to assess the presence and abundance of federally listed species along the coasts of Washington and Oregon. During the study, 16 sea turtles were observed; all sightings were of leatherback sea turtles and all occurred between June and September, with most sightings (10) occurring in July (Green et al. 1992). The data suggest that leatherback sea turtles are most likely to occur in the NWTT action area in summer and early fall when water temperatures are warmer.

New population modeling conducted by Gaspar and Lalire (2017) compare Pacific juvenile leatherback predicted distributions with passive dispersion (juvenile turtles drifting or following currents) and active dispersion, where juvenile turtles respond to habitat cues (e.g., water temperature) and actively swim to foraging grounds often counter to prevailing currents. This modeling effort suggests that oceanic currents broadly shape the dispersal area of leatherbacks within the North Central Pacific Basin, and habitat-driven movements strongly influence the spatial and temporal distribution of juveniles within this area. Specifically, these habitat-driven movements lead juveniles to gather in the North Pacific Transition Zone and to undertake seasonal north-south migrations. The modeling effort also suggest that juveniles in the North Pacific Transition Zone migrate westward, counter to prevailing currents, thereby increasing residence time. This likely exposes leatherbacks in the Pacific to increased risk of interactions with fisheries, in the central and eastern part of the North Pacific basin. Habitat-driven movements modeled by Gaspar and Lalire (2017) would also reduce the risk of cold-induced mortality. This risk appears to be larger among the juveniles that rapidly circulate into the Kuroshio Current than in other, more southern latitude currents.

Occurrence in the Inland Waters of the NWTT action area

Leatherback sea turtles are occasionally sighted within the Strait of Juan de Fuca, but are rare in Puget Sound (NMFS and USFWS 1998b; NMFS and USFWS 2007b). While leatherback sea turtles are capable of foraging in inland waters, their preferred foraging habitat is offshore. Given their rare occurrence in these areas, it is extremely unlikely that leatherback sea turtles would overlap spatially with Navy activities in the inland portion of the action area. Any effects resulting from the proposed action on leatherback sea turtles in the inland waters are considered discountable. Therefore, the leatherback sea turtle is not analyzed further for military activities occurring in the Inland Waters portion of the NWTT action area.

Occurrence in the Western Behm Canal, Alaska - NWTT action area

Stinson's (1984) archival search of 363 sea turtles sighted along the Pacific coast from Baja California, Mexico to the Gulf of Alaska from 1917 to 1982 indicated that only 6 of the recorded

leatherback sightings occurred in Alaska waters. From 1960 to 1968, 19 sightings of leatherback turtles in Alaska waters were recorded (11 were sightings, 3 were netted and released, 3 were netted and killed, and 2 were carcasses) (Hodge and Wing 2000). While leatherback sea turtles are physically capable of foraging in Alaska waters, they are rare in the area and prefer offshore waters to the south. Given their rare occurrence in Alaska, it is extremely unlikely that leatherback sea turtles would overlap spatially with Navy activities in Western Behm Canal. Any effects resulting from the proposed action on leatherback sea turtles in Western Behm Canal are considered discountable. Therefore, the leatherback sea turtle is not analyzed further for activities occurring in the Western Behm Canal portion of the action area.

Diversity

The West Pacific DPS exhibits genetic diversity, with six haplotypes identified in 106 samples from Solomon Islands, Papua Barat Indonesia, and Papua New Guinea (Dutton 2006). The population also exhibits temporal nesting diversity, with various proportions of the population nesting during different times of the year (summer versus winter) which helps to increase resilience to environmental impacts (NMFS 2020d). The foraging strategies are also diverse, with turtles using seven ecoregions of the Pacific Ocean (NMFS 2020d), which likely provide some resilience against local reductions in prey availability or catastrophic events, such as oil spills or typhoons, by limiting exposure to only a portion of the DPS (NMFS 2020d). Overall, diversity within the West Pacific DPS likely provides it with some resilience to threats (NMFS 2020d).

Abundance Estimate

There are no known nesting habitats for the leatherback sea turtle in the action area. There are 28 known nesting sites for the western Pacific Ocean stock ranging across the western tropical Pacific Ocean, from Australia and Melanesia (Papua New Guinea, Solomon Islands, Fiji, and Vanuatu) to Indonesia, Thailand, and China (Chaloupka et al. 2004; Chua 1988; Dutton 2006; Hirth et al. 1993; Suarez et al. 2000). The major nesting populations of the eastern Pacific Ocean stock occur in Mexico, Costa Rica, and Nicaragua (Chaloupka et al. 2004; Dutton et al. 1999; Eckert and Sarti 1997; Márquez 1990; Sarti M. 1996; Spotila et al. 1996), with the largest ones in Mexico and Costa Rica.

NMFS and USFWS (2020) estimated the total index of nesting female abundance of the West Pacific DPS to be 1,277 females. This number represents an index of nesting female abundance for this DPS because it only includes available data from recently (as of 2014) and consistently monitored (over the remigration interval) nesting beaches: Jamursba-Medi and Wermon, Indonesia. It does not include nesting females from other beaches of Indonesia, Papua New Guinea, Solomon Islands or Vanuatu because these areas have not been consistently monitored for nesting in recent years. Therefore, actual nesting female abundance could be higher, given the potential for unidentified or unmonitored nesting beaches.

Martin et al. (2020) estimated the abundance of Western Pacific leatherbacks for the two index beaches in Indonesia, which represent approximately 75 percent of all nesting individuals. Using the median value for imputed nest counts they estimated 790 total nesters (95% CI: 666–942). Jones et al. (2018) used model-estimated annual female distributions for 2015 to 2017 to estimate an index of current total reproductive female abundance for the western Pacific leatherback population. This was computed as a 3-year run sum based on an assumed 3-year remigration interval. The estimates for 2015-2017 annual females ranged from 340 to 439 and the summed total reproductive female estimate was 1,180 (95% CI: 949–1,479) (Jones et al. 2018). Using this estimate, and assuming a 3:1 ratio of females to males, NMFS (2019g) estimated the current adult portion of the population is 1,851 (1,488-2,320). NMFS (2019g) used the proportion or change in the estimates derived from the information contained in Jones et al. (2018) to estimate the current population size of the West Pacific Ocean leatherback sea turtle. The total West Pacific Ocean population estimate is 175,000 leatherback sea turtles, but may range between 68,000 and 360,000 individuals (NMFS 2019g).

Status and Trends

The leatherback turtle was listed under the Endangered Species Act as endangered throughout its range in 1970. The Pacific Ocean leatherback population is generally smaller in size than that in the Atlantic Ocean. With only four major rookeries remaining in the Western Pacific Ocean and two in the Eastern Pacific Ocean, the Pacific leatherback is at an extremely high risk of extinction. This Western Pacific DPS exhibits low hatching success and decreasing nest and population trends due to past and current threats, which are likely to further lower abundance and increase the risk of extinction (NMFS 2020d).

Leatherback sea turtles are endangered by several human activities, including fisheries interactions, entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), direct harvest, egg collection, the destruction and degradation of nesting and coastal habitat, boat collisions, and ingestion of marine debris (Chaloupka et al. 2004; Eckert and Sarti 1997; Sarti M. 1996). The primary threat to the West Pacific DPS is the legal and illegal harvest of leatherback turtles and their eggs (NMFS 2020d).

The index of nesting female abundance of the West Pacific DPS (n = 1,277) places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (NMFS 2020d). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb impacts to habitat or losses in individuals (NMFS 2020d). Low site fidelity and dispersal of nests among various beaches may help to reduce population level impacts from threats which may disproportionately affect one area over another. However, due to its small size, the DPS has restricted capacity to buffer such losses (NMFS 2020d). The median trend in annual nest counts estimated for Jamursba Medi (data collected from 2001 to 2017) was −5.7 percent annually (NMFS 2020d). For Wermon (data collected from 2006 to 2017, excluding

2013–2015 due to low or insufficient effort), the median trend was −2.3 percent annually (NMFS 2020d).

Critical Habitat

On January 26, 2012, NMFS issue a final rule to revise the current critical habitat for the leatherback sea turtle by designating additional areas within the Pacific Ocean (77 FR 4169; effective February 27, 2012). This designation includes approximately 16,910 mi^2 (43,798 km^2) stretching along the California coast from Point Arena to Point Arguello east of the 3,000 meters depth contour; and 25,004 mi^2 (64,760 km^2) stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meters depth contour (Figure 45). Only the northern portion of designated critical habitat (i.e., Cape Flattery to Cape Blanco) overlaps with the NWTT action area.

The leatherback sea turtle Critical Habitat Review Team identified one essential physical and biological feature (formerly referred to as Primary Constituent Elements or PCEs) that is essential for the conservation of leatherbacks in marine waters of the U.S. West Coast:

The occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (e.g., Chrysaora, Aurelia, Phacellophora, and Cyanea), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

All areas of the Pacific Coast critical habitat, including the portion that overlaps with the action area, scored a high conservation value, which is consistent with the scientific literature and observations of a high level of leatherback foraging in these areas (NMFS 2012b). The Critical Habitat Review Team identified the following activities within the action area that may impact the prey essential feature describe above: point source pollution, pesticide application, oil spill response, tidal, wave and wind energy projects, and LNG.

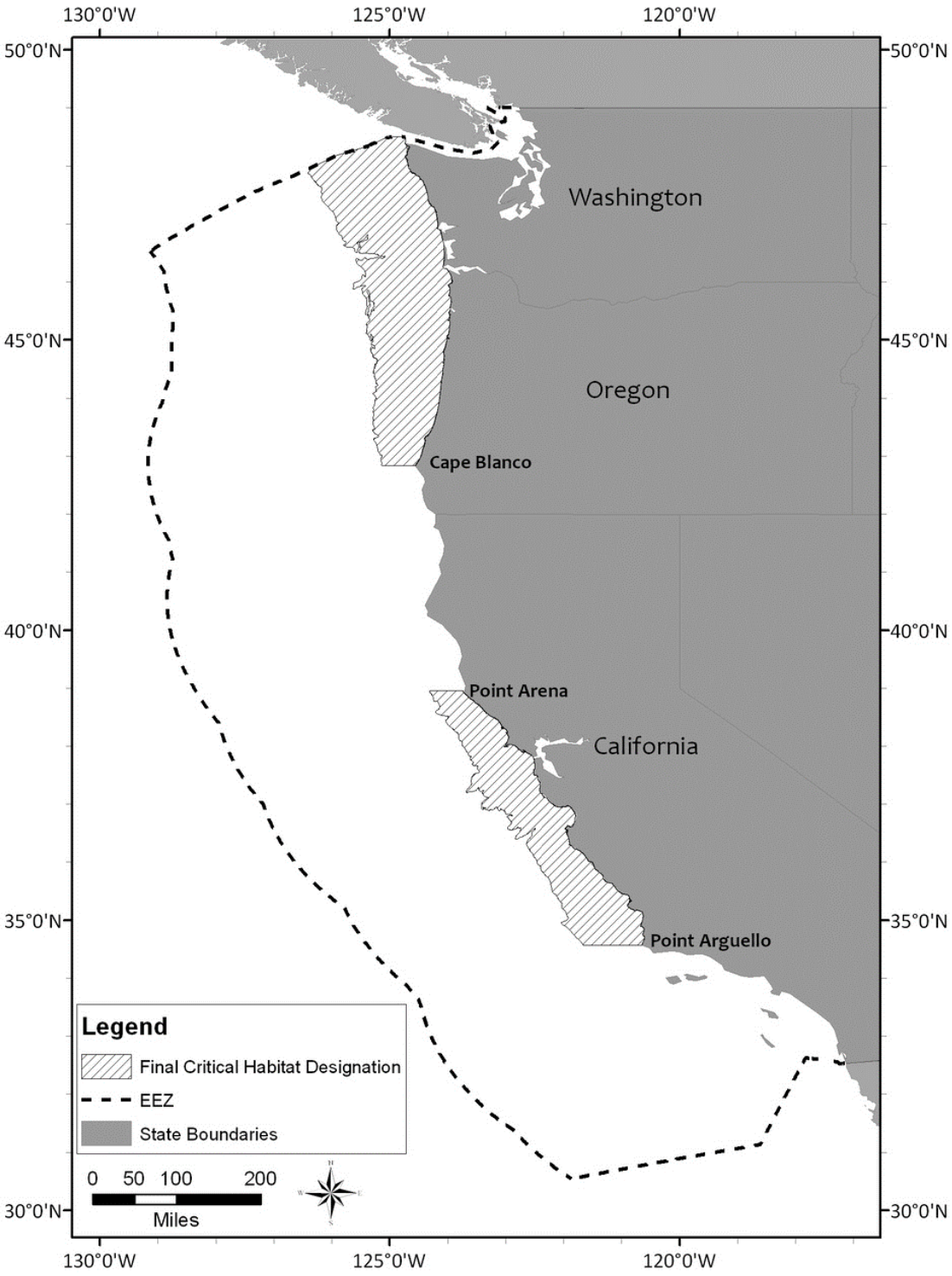


Figure 45. Leatherback sea turtle designated critical habitat on the Pacific Coast.

Recovery Goals

See the U.S. Pacific Recovery Plan (NMFS and USFWS 1998a) for leatherback sea turtles for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top five recovery actions identified to support in the Leatherback Five Year Action Plan (NOAA 2016):

- Reduce fisheries interactions
- Improve nesting beach protection and increase reproductive output
- International cooperation
- Monitoring and research
- Public engagement

6.2.9 Chinook Salmon – California Coastal Evolutionarily Significant Unit (ESU)

Chinook salmon, also referred to as king salmon in California, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002). On September 16, 1999, NMFS listed the California Coastal ESU of Chinook salmon as a "threatened" species (FR 64 50394). On June 28, 2005, NMFS confirmed the listing of California Coastal Chinook salmon as threatened under the ESA and also added seven artificially propagated populations from the following hatcheries or programs to the listing. The California Coastal Chinook salmon Evolutionarily Significant Unit (ESU) includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River (Humboldt County, CA.) to the Russian River (Sonoma County, CA) (Figure 46).

Life History

California Coastal Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt et al. 2005). The different populations vary in run timing depending on latitude and hydrological differences between watersheds. Entry of California Coastal Chinook salmon into the Russian River depends on increased flow from fall storms, usually in November to January. Juveniles of this ESU migrate downstream from April through June and may reside in the estuary for an extended period before entering the ocean.

The length of time required for embryo incubation and emergence from the gravel is dependent on water temperature. For maximum embryo survival, water temperatures reportedly must be between 41°F and 55.4°F and oxygen saturation levels must be close to maximum. Under those conditions, embryos hatch in 40 to 60 days and remain in the gravel as alevins (the life stage between hatching and egg sack absorption) for another 4 to 6 weeks before emerging as fry.

Juveniles may reside in freshwater for 12 to 16 months, but some migrate to the ocean as young-of-the- year in the winter or spring months within eight months of hatching.

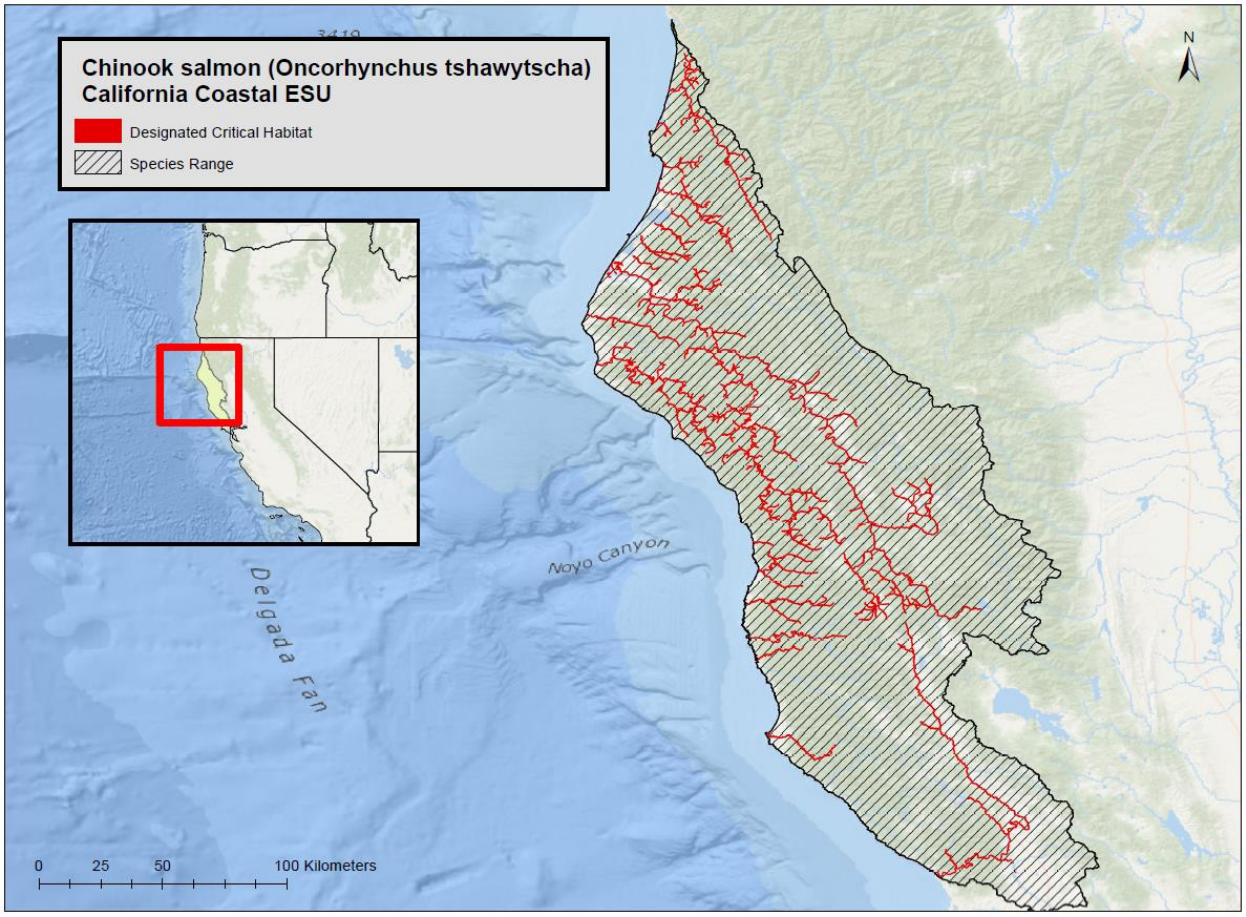


Figure 46. Geographic range and designated critical habitat of California coastal ESU Chinook salmon.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the California Coastal ESU of Chinook salmon.

Comparison of historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many basins (NMFS 2019d). Current abundance estimates for adult and juvenile California Coastal Chinook salmon are estimated to be 7,034 and 1,278,078 individuals, respectively (See Table 67).

Table 67. Average abundance for California Coastal Chinook salmon natural-origin spawners (NMFS 2019d).

Population	Years	Spawners	Expected Number of Outmigrants[ab]
Redwood Creek	2009-2013	1,745	317,067
Mad River	2010-2015	71	12,900
Freshwater Creek	2010-2015	6	1,090
Eel River mainstem	2010-2015	1,198	217,677
Eel River (Tomki Creek)	2010-2015	70	12,719
Eel River (Sproul Creek)	2010-2015	103	18,715
Mattole River	2007-2009, 2012, 2013	648	117,742
Russian River	2009 - 2014	3,137	569,993
Ten Mile River	2009 - 2014	6	1,090
Noyo River	2009 - 2014	14	2,544
Big River	2009 - 2014	13	2,362
Albion River	2009 - 2014	15	2,726
Navarro River	2009 - 2014	3	545
Garcia River	2009 - 2014	5	909
ESU Average		**7,034**	**1,278,078**

[a]Expected number of outmigrants = total spawners*50 percent proportion of females*3,634 eggs per female*10 percent survival rate from egg to outmigrant.

[b]Based upon number of natural-origin spawners.

The available data, a mixture of short-term (six-year or less) population estimates or expanded red (nest) estimates and longer-term partial population estimates and spawner/red indexes, provide no indication that any of the independent populations (likely to persist in isolation) are approaching viability targets. Overall, there is a lack of compelling evidence to suggest that the status of these populations has improved or deteriorated appreciably since the previous status review (Williams et al. 2011c).

At the ESU level, the loss of the spring-run life history type represents a significant loss of diversity within the ESU, as has been noted in previous status reviews (Williams et al. 2011c). Concern remains about the extremely low numbers of Chinook salmon in most populations of the North-Central Coast and Central Coast strata, which diminishes connectivity across the ESU. However, the fact that Chinook salmon have regularly been reported in the Ten Mile, Noyo, Big, Navarro, and Garcia rivers represents a significant improvement in our understanding of the status of these populations in watersheds where they were thought to have been extirpated. These

observations suggest that spatial gaps between extant populations are not as extensive as previously believed.

The California Coastal Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River to the Russian River, California (64 FR 50394; September 16, 1999). Seven artificial propagation programs are considered to be part of the ESU: The Humboldt Fish Action Council (Freshwater Creek), Yager Creek, Redwood Creek, Hollow Tree, Van Arsdale Fish Station, Mattole Salmon Group, and Mad River Hatchery fall-run Chinook salmon hatchery programs. These artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the ESU (NMFS 2016c).

Status

The California Coastal Chinook salmon ESU was historically comprised of 38 populations which included 32 fall-run populations and 6 spring-run populations across four Diversity Strata (NWFSC 2015b). All six of the spring-run populations were classified as functionally independent, but are considered extinct (NMFS 2016c). NMFS (2016c) cited continued evidence of low population sizes relative to historical abundance, mixed trends in the few available time series of abundance indices available, and low abundance and extirpation of populations in the southern part of the ESU. In addition, the apparent loss of the spring-run life history type throughout the entire ESU as a significant diversity concern. The 2016 recovery plan determined that the four threats of greatest concern to the ESU are channel modification, roads and railroads, logging and wood harvesting, and both water diversion and impoundments and severe weather patterns.

Critical Habitat

NMFS designated critical habitat for the California Coastal Chinook salmon on September 2, 2005 (70 FR 52488). It includes multiple CALWATER hydrological units north from Redwood Creek and south to Russian River. Designated critical habitat for the California Coastal Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The total area of critical habitat includes 1,500 mi of stream habitat and about 25 mi^2 of estuarine habitat, mostly within Humboldt Bay. The PBFs that characterize these areas include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. There are 45 occupied CALWATER Hydrologic Subarea (HSA) watersheds within the freshwater and estuarine range of this ESU. Eight watersheds received a low rating, 10 received a medium rating, and 27 received a high rating of conservation value to the ESU (70 FR 52488). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating. Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing

habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The current condition of PBFs of the California Coastal Chinook salmon critical habitat indicates that PBFs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU.

Recovery Goals

Recovery goals, objectives and criteria for the California Coastal chinook salmon are fully outlined in NMFS (2016g). Recovery plan objectives are to: 1) Reduce the present or threatened destruction, modification, or curtailment of habitat or range; 2) Ameliorate utilization for commercial, recreational, scientific, or educational purposes; 3) Abate disease and predation; 4) Establish the adequacy of existing regulatory mechanisms for protecting California Coastal Chinook salmon now and into the future (i.e., post-delisting); 5) Address other natural or manmade factors affecting the continued existence of California Coastal Chinook salmon; and 6) Ensure the status of California Coastal Chinook salmon is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

6.2.10 Chinook Salmon – Central Valley Spring-Run ESU

The Chinook salmon, Central Valley spring-run ESU includes naturally spawned spring-run Chinook salmon originating from the Sacramento River and its tributaries, and also spring-run Chinook salmon from the Feather River Hatchery Spring-run Chinook salmon Program (Figure 47).

The physical attributes of Chinook salmon are discussed in Section 6.2.9. On September 16, 1999, NMFS listed the Central Valley ESU of spring-run Chinook salmon as a "threatened" species (FR 64 50394). Historically, spring-run Chinook salmon occurred in the headwaters of all major river systems in the Central Valley where natural barriers to migration were absent. The only known streams that currently support self-sustaining populations of non-hybridized spring-run Chinook salmon in the Central Valley are Mill, Deer and Butte creeks. Each of these populations is small and isolated (NMFS 2014c).

Life History

Adult Central Valley spring-run Chinook salmon leave the ocean to begin their upstream migration in late January and early February, and enter the Sacramento River between March and September, primarily in May and June (Moyle 2002; Yoshiyama et al. 1998). Spring-run Chinook salmon generally enter rivers as sexually immature fish and must hold in freshwater for up to several months before spawning. While maturing, adults hold in deep pools with cold water. Spawning normally occurs between mid- August and early October, peaking in September (Moyle 2002).

The length of time required for embryo incubation and emergence from the gravel is dependent on water temperature. For maximum embryo survival, water temperatures reportedly must be

between 41°F and 55.4°F and oxygen saturation levels must be close to maximum. Under those conditions, embryos hatch in 40 to 60 days and remain in the gravel as alevins (the life stage between hatching and egg sack absorption) for another 4 to 6 weeks before emerging as fry. Spring-run fry emerge from the gravel from November to March (Moyle 2002). Juveniles may reside in freshwater for 12 to 16 months, but some migrate to the ocean as young-of-the-year in the winter or spring months within eight months of hatching.

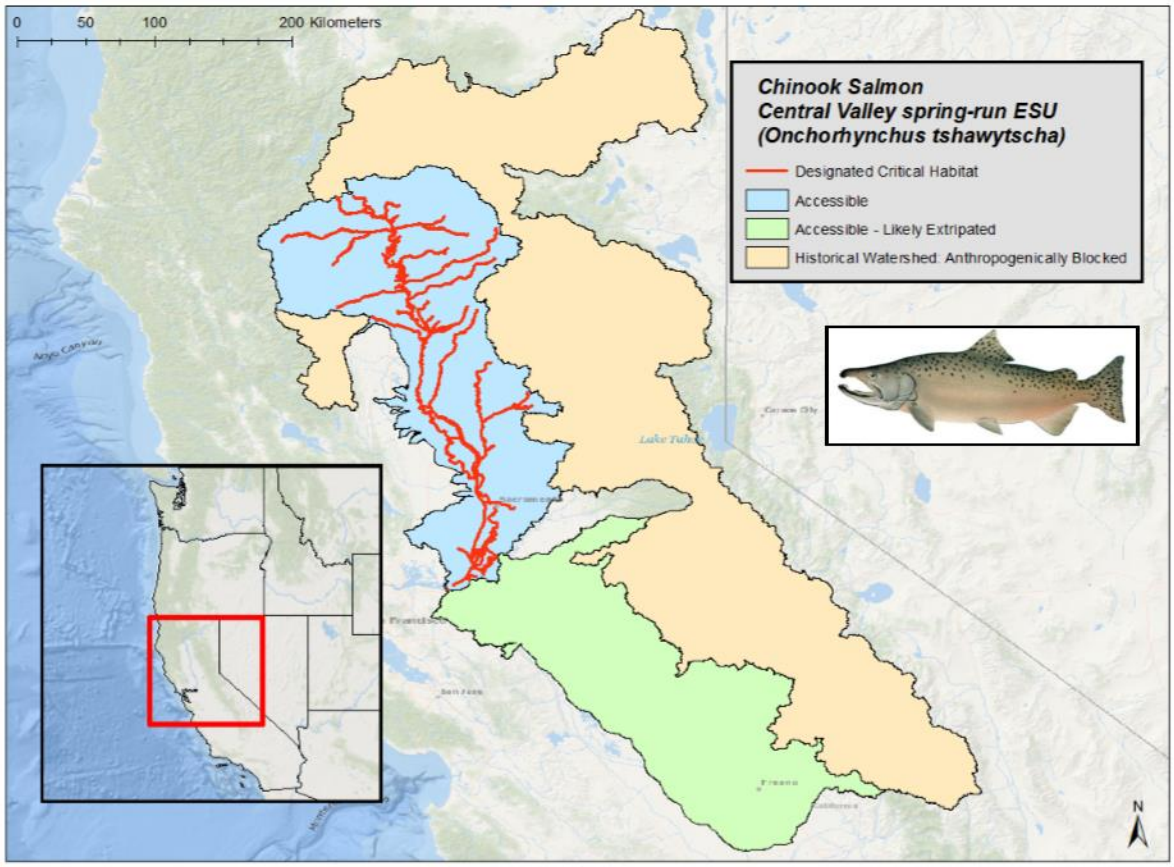


Figure 47. Geographic range and designated critical habitat of Central Valley spring-run ESU Chinook salmon.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Central Valley spring-run ESU of Chinook salmon.

The Central Valley as a whole is estimated to have supported spring-run Chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s. Current abundance estimates for the Central Valley spring-run ESU of Chinook salmon are presented in Table 68 below.

Table 68. Average abundance estimates for Central Valley Spring Run Chinook salmon natural- and hatchery-origin spawners from 2013-2017 (NMFS 2019d).

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[a]	% Hatchery Origin	Expected Number of Outmigrants[b]
Southern Cascades Stratum				
Battle Creek	191	0	0%	39,761
Mill Creek	302	0	0%	62,807
Deer Creek	409	0	0%	85,049
Butte Creek	2,750	0	0%	572,056
Big Chico Creek	0	0	0%	0
Antelope Creek	3	0	0%	598
Coastal Range Stratum				
Clear Creek	73	0	0%	15,143
Cottonwood / Beegum creeks	0.3	0	0%	60
Northern Sierra Stratum				
Feather River	0	2,273	100%	-
ESU Average	**3,727**	**2,273**	**37.9%**	**775,474**

[a] Geometric mean (2013-2017) of post-fishery spawners.

[b] Expected number of outmigrants=Total spawners*50% proportion of females*4,131 eggs per female*10% survival rate from egg to outmigrant.

Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation. The majority of Central Valley spring-run Chinook salmon are found to return as three-year olds, therefore looking at returns every three years is used as an estimate of the CRR. In the past, the CRR has fluctuated between just over 1.0 to just under 0.5, and in the recent years with high returns (2012 and 2013), CRR jumped to 3.84 and 8.68 respectively. CRR for 2014 was 1.85, and the CRR for 2015 with very low returns was a record low of 0.14. Low returns in 2015 were further decreased due to high temperatures and most of the Central Valley spring-run Chinook salmon tributaries experienced some pre-spawn mortality. Butte Creek experienced the highest prespawn mortality in 2015, resulting in a carcass survey CRR of only 0.02 (NMFS 2016a).

Threats to the genetic integrity of spring-run Chinook salmon was identified as a serious concern to the species when it was listed in 1999 (FR 64 50394; Myers et al. 1998a). Three main factors compromised the genetic integrity of spring-run Chinook salmon: (1) the lack of reproductive isolation following dam construction throughout the Central Valley resulting in introgression with fall-run Chinook salmon in the wild; (2) within basin and inter-basin mixing between spring and fall broodstock for artificial propagation, resulting in introgression in hatcheries; and (3) releasing hatchery-produced juvenile Chinook salmon in the San Francisco estuary, which contributes to the straying of returning adults throughout the Central Valley (NMFS 2014c).

The Central Valley Technical Recovery Team delineated 18 or 19 historic independent populations of Central Valley spring-run Chinook salmon, and a number of smaller dependent populations, that are distributed among four diversity groups (southern Cascades, northern Sierra, southern Sierra, and Coast Range) (Lindley et al. 2004). Of these independent populations, only three are extant (Mill, Deer, and Butte creeks) and they represent only the northern Sierra Nevada diversity group. Of the dependent populations, Central Valley spring-run Chinook salmon are found in Battle, Clear, Cottonwood, Antelope, Big Chico, and Yuba creeks, as well as the Sacramento and Feather rivers and a number of tributaries of the San Joaquin River including Mokelumne, Stanislaus, and Tuolumne rivers. The 2005 listing determination concluded that the Feather River Fish Hatchery spring-run Chinook salmon production should be included in the Central Valley spring-run Chinook salmon ESU (NMFS 2016a).

Status

Although spring-run Chinook salmon were probably the most abundant salmonid in the Central Valley, this ESU has suffered the most severe declines of any of the four Chinook salmon runs in the Sacramento River Basin (Fisher 1994). The ESU is currently limited to independent populations in Mill, Deer, and Butte creeks, persistent and presumably dependent populations in the Feather and Yuba rivers and in Big Chico, Antelope, and Battle creeks, and a few ephemeral or dependent populations in the Northwestern California region (e.g., Beegum, Clear, and Thomes creeks) (NMFS 2016a). The Central Valley spring-run Chinook salmon ESU is currently faced with three primary threats: (1) loss of most historic spawning habitat; (2) degradation of the remaining habitat; and (3) genetic introgression with the Feather River fish hatchery spring-run Chinook salmon strays. The potential effects of climate change are likely to adversely affect spring-run Chinook salmon and their recovery (NMFS 2014c).

Critical Habitat

NMFS published a final rule designating critical habitat for Central Valley spring-run Chinook salmon on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. Designated critical habitat for the Central Valley spring-run Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. The current condition of PBFs of the Central Valley Spring-run Chinook salmon critical habitat indicates that PBFs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. Spawning and rearing PBFs are degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds which maintained cool and clean water throughout the summer. The rearing PBF is degraded by floodplain habitat being disconnected from the mainstem of larger rivers throughout the Sacramento River watershed, thereby reducing effective foraging. Migration PBF is degraded by lack of natural cover along the migration corridors. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Recovery Goals

Recovery goals, objectives and criteria for the Central Valley spring-run Chinook salmon are fully outlined in the 2014 Recovery Plan (NMFS 2014c). The ESU delisting criteria for the spring-run Chinook salmon are: 1) One population in the Northwestern California Diversity Group at low risk of extinction; 2) Two populations in the Basalt and Porous Lava Diversity Group at low risk of extinction; 3) Four populations in the Northern Sierra Diversity Group at low risk of extinction; 4) Two populations in the Southern Sierra Diversity Group at low risk of extinction; and 5) Maintain multiple populations at moderate risk of extinction.

6.2.11 Chinook Salmon – Lower Columbia River ESU

Chinook salmon, Lower Columbia River ESU includes naturally spawned Chinook salmon originating from the Columbia River and its tributaries downstream of a transitional point east of the Hood and White Salmon Rivers, and any such fish originating from the Willamette River and its tributaries below Willamette Falls (Figure 48).

The physical attributes of Chinook salmon are discussed in Section 6.2.9. On March 24, 1999, NMFS listed the Lower Columbia River ESU of Chinook salmon as a "threatened" species (64 FR 14308). The listing was revisited and confirmed as "threatened" in 2005 (70 FR 37160).

Life History

Lower Columbia River Chinook salmon display three run types including early fall-runs, late fall-runs, and spring-runs. Presently, the fall-run is the predominant life history type. Spring-run Chinook salmon were numerous historically. Fall-run Chinook salmon enter fresh water typically in August through October. Early fall-run spawn within a few weeks in large river mainstems. The late fall-run enters in immature conditions, has a delayed entry to spawning grounds, and resides in the river for a longer time between river entry and spawning. Spring-run Chinook

salmon enter fresh water in March through June to spawn in upstream tributaries in August and September.

Offspring of fall-run spawning may migrate as fry to the ocean soon after yolk absorption (*i.e.*, ocean-type), at 30–45 mm in length (Healey 1991). In the Lower Columbia River system, however, the majority of fall-run Chinook salmon fry migrate either at 60-150 days post-hatching in the late summer or autumn of their first year. Offspring of fall-run spawning may also include a third group of yearling juveniles that remain in fresh water for their entire first year before emigrating. The spring-run Chinook salmon migrates to the sea as yearlings (stream-

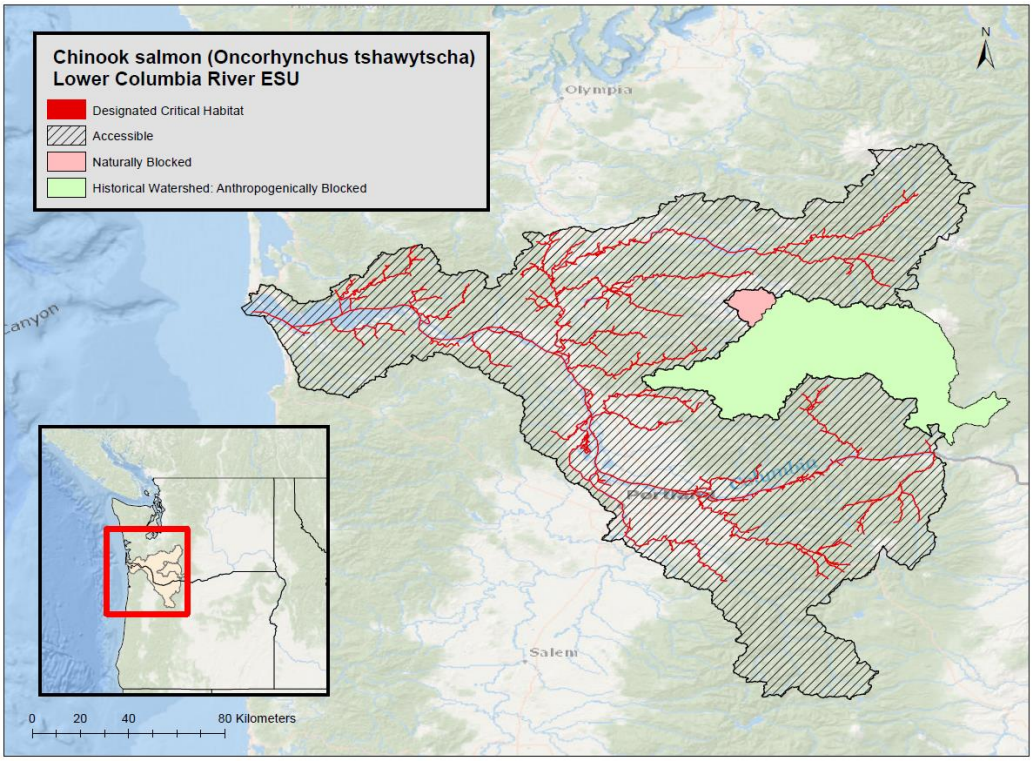


Figure 48. Geographic range and designated critical habitat of Lower Columbia River ESU Chinook salmon.

type) typically in spring. However, the natural timing of Lower Columbia River spring-run Chinook salmon emigration is obscured by hatchery releases (Myers et al. 2006). Once at sea, the ocean-type Columbia River Chinook salmon tend to migrate along the coast, while stream-type LCR Chinook salmon appear to move far off the coast into the central North Pacific Ocean (Healey 1991; Myers et al. 2006). Adults return to tributaries in the Lower Columbia River predominately as three- and four-year-olds for fall-run fish and four- and five-year-olds for spring-run fish.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items

(Kjelson et al. 1981; MacFarlane and Norton 2002; Sommer et al. 2001a). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the lower Columbia River ESU of Chinook salmon.

Populations of Lower Columbia River Chinook salmon have declined substantially from historical levels. Many of the ESU's populations are believed to have very low abundance of natural-origin spawners (100 fish or fewer), which increases genetic and demographic risks. Other populations have higher total abundance, but several of these also have high proportions of hatchery-origin spawners (NMFS 2016d). Current abundance estimates for the Lower Columbia River ESU of Chinook salmon are presented in Table 69 below.

Table 69. Abundance Estimates for the Lower Columbia River ESU of Chinook salmon (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	29,469
Natural	Juvenile	11,745,027
Listed Hatchery Intact Adipose	Juvenile	962,458
Listed Hatchery Clipped and Intact Adipose	Adult	38,594
Listed Hatchery Adipose Clip	Juvenile	31,353,395

The genetic diversity of all populations (except the late fall-run Chinook salmon) has been eroded by large hatchery influences and periodically by low effective population sizes. The near loss of the spring-run life history type remains an important concern for maintaining diversity within the ESU (NMFS 2016d).

The ESU spans three distinct ecological regions: Coastal, Cascade, and Gorge. Distinct life-histories (run and spawn timing) within ecological regions in this ESU were identified as major population groups (MPGs). In total, 32 historical demographically independent populations (DIPs) were identified in this ESU, 9 spring-run, 21 fall-run, and 2 late-fall run, organized in 6 MPGs (based on run timing and ecological region). The basin wide spatial structure has remained generally intact. However, the loss of about 35% of historic habitat has affected distribution within several Columbia River subbasins (NMFS 2016d).

Status

Populations of Lower Columbia River Chinook salmon have declined substantially from historical levels. Out of the 32 populations that make up this ESU, only the two late-fall runs (the North Fork Lewis and Sandy) are considered viable. Most populations (26 out of 32) have a very low probability of persistence over the next 100 years and some are extirpated or nearly so. Five of the six strata fall significantly short of the recovery plan criteria for viability. Low abundance, poor productivity, losses of spatial structure, and reduced diversity all contribute to the very low persistence probability for most Lower Columbia River Chinook salmon populations. Hatchery contribution to naturally-spawning fish remains high for a number of populations, and it is likely that many returning unmarked adults are the progeny of hatchery origin parents, especially where large hatchery programs operate. Continued land development and habitat degradation in combination with the potential effects of climate change will present a continuing strong negative influence into the foreseeable future (NMFS 2016d).

Critical Habitat

NMFS designated critical habitat for Lower Columbia River Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins. Designated critical habitat for the Lower Columbia River Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival.

Timber harvest, agriculture, and urbanization have degraded spawning and rearing PBFs by reducing floodplain connectivity and water quality, and by removing natural cover in several rivers. Hydropower development projects have reduced timing and magnitude of water flows, thereby altering the water quantity needed to form and maintain physical habitat conditions and support juvenile growth and mobility. Adult and juvenile migration PBFs are affected by several dams along the migration route.

Recovery Goals

Recovery plan targets for this species are tailored for each life history type, and within each type, specific population targets are identified (NMFS 2013a). For spring Chinook salmon, all populations are affected by aspects of habitat loss and degradation. Four of the nine populations require significant reductions in every threat category. Protection and improvement of tributary and estuarine habitat are specifically noted.

For fall Chinook salmon, recovery requires restoration of the Coast and Cascade strata to high probability of persistence, to be achieved primarily by ensuring habitat protection and restoration. Very large improvements are needed for most fall Chinook salmon populations to improve their probability of persistence.

For late fall Chinook salmon, recovery requires maintenance of the North Fork Lewis and Sandy populations which are comparatively healthy, together with improving the probability of persistence of the Sandy population from its current status of "high" to "very high." Improving the status of the Sandy population depends largely on harvest and hatchery changes. Habitat improvements to the Columbia River estuary and tributary spawning areas are also necessary. Of the 32 DIPs in this ESU, only the 2 late-fall run populations (Lewis River and Sandy River) could be considered viable or nearly so (NMFS 2016d).

6.2.12 Chinook Salmon – Puget Sound ESU

The Puget Sound ESU includes naturally spawned Chinook salmon originating from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Georgia Strait. Twenty-six artificial propagation programs are included as part of the Puget Sound ESU (Figure 49). The physical attributes of Chinook salmon are discussed in Section 6.2.9. On March 24, 1999, NMFS listed the Puget Sound ESU of Chinook salmon as a "threatened" species (64 FR 14308). The listing was revisited and confirmed as "threatened" in 2005 (70 FR 37160).

Life History

Puget Sound Chinook salmon populations are both early-returning (August) and late-returning (mid-September and October) spawners (Healey 1991). Juvenile Chinook salmon within the Puget Sound generally exhibit an "ocean-type" life history. However, substantial variation occurs with regard to juvenile residence time in freshwater versus estuarine environments. Hayman (Hayman et al. 1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20 percent to 60 percent of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer et al. 2005). Juveniles in tributaries to Lake Washington exhibit both a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (<1 meters) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor et al. 2006). Puget Sound Chinook salmon also have several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing (Beamer et al. 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature Chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan et al. 2004).

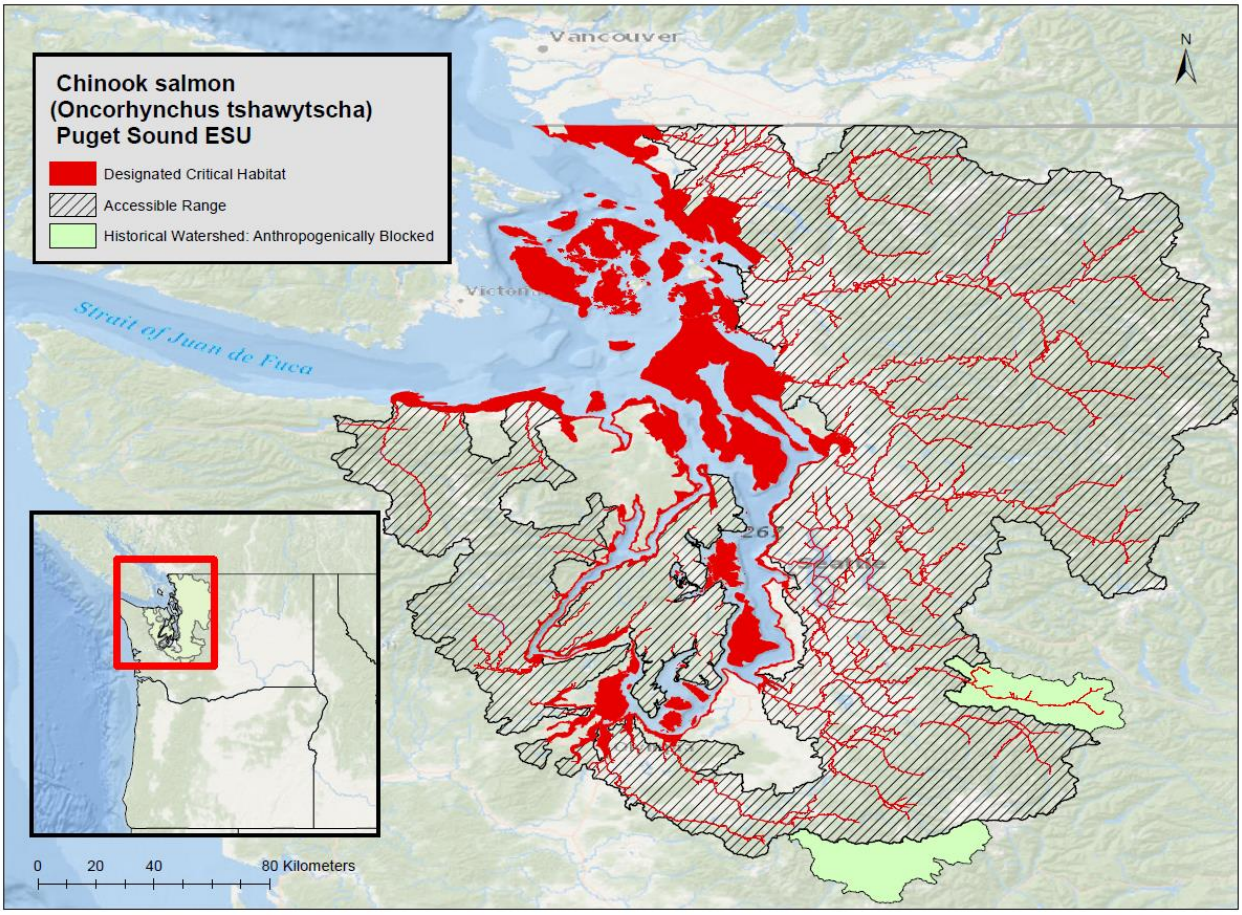


Figure 49. Geographic range and designated critical habitat of Puget Sound ESU Chinook salmon.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Puget Sound ESU of Chinook salmon.

Estimates of the historic abundance range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. During the period from 1996 to 2001, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over

9,489 fish. Thus, the historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005). Current abundance estimates for the Puget Sound ESU of Chinook salmon are found in Table 70 and Table 71 below.

Table 70. Average abundance estimates for Puget Sound Chinook salmon natural- and hatchery-origin spawners 2012-2016 (NMFS 2020c).

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[a]	% Hatchery Origin	Minimum Viability Abundance[b]	Expected Number of Outmigrants[c]
Georgia Strait MPG					
NF Nooksack River[d]	181	945	83.95%	16,000	90,009
SF Nooksack River[d]	18	15	45.04%	9,100	2,597
Strait of Juan de Fuca MPG					
Elwha River	130	2,156	94.30%	15,100	182,895
Dungeness River	189	213	52.91%	4,700	32,163
Hood Canal MPG					
Skokomish River	224	1,158	83.82%	12,800	110,505
Mid-Hood Canal	165	117	41.55%	11,000	22,589
Whidbey Basin MPG					
Skykomish River	2,001	1,466	42.29%	17,000	277,348
Snoqualmie River	881	219	19.93%	17,000	87,978
NF Stillaguamish River	385	291	43.04%	17,000	54,137
SF Stillaguamish River	42	29	40.57%	15,000	5,676
Upper Skagit River	9,505	120	1.25%	17,000	770,047
Lower Skagit River	2,207	13	0.60%	16,000	177,643
Upper Sauk River	1,106	5	0.46%	3,000	88,899
Lower Sauk River	559	3	0.59%	5,600	44,984
Suiattle River	590	5	0.77%	600	47,582
Cascade River	205	7	3.12%	1,200	16,937
Central / South Sound MPG					
Sammamish River	125	885	87.64%	10,500	80,823
Cedar River	883	440	33.26%	11,500	105,864
Duwamish/Green River	1,120	4,171	78.83%	17,000	423,326
Puyallup River	565	1,240	68.72%	17,000	144,384
White River	569	1,438	71.64%	14,200	160,622
Nisqually River	747	606	44.81%	13,000	108,281
ESU Average	**22,398**	**15,543**	**40.97%**		**3,035,288**

[a] Five-year geometric mean of post-fishery spawners (2013-2017).

[b] Ford (2011a)

[c] Expected number of outmigrants = total spawners*40% proportion of females*2,000 eggs per female*10% survival rate from egg to outmigrant

[d] 2012-2016 five year geometric mean (2017 data not available).

Table 71. Expected 2019 Puget Sound Chinook salmon hatchery releases (NMFS 2020c).

Subbasin	Artificial propagation program	Brood year	Run Timing	Clipped Adipose Fin	Intact Adipose Fin
Deschutes	Tumwater Falls	2018	Fall	3,800,000	-
Dungeness-Elwha	Dungeness	2018	Spring	-	50,000
	Elwha	2017	Fall	-	200,000
		2018	Fall	250,000	2,250,000
	Gray Wolf River	2018	Spring	-	50,000
	Hurd Creek	2018	Spring	-	50,000
	Upper Dungeness Pond	2018	Spring	-	50,000
Duwamish	Icy Creek	2017	Fall	300,000	-
	Palmer	2018	Fall	-	1,000,000
	Soos Creek	2018	Fall	3,000,000	200,000
Hood Canal	Hood Canal Schools	2018	Fall	-	500
	Hoodsport	2017	Fall	120,000	-
		2018	Fall	3,000,000	-
Kitsap	Bernie Gobin	2017	Spring	40,000	-
		2018	Fall	-	200,000
			Summer	2,300,000	100,000
	Garrison	2018	Fall	850,000	-
	George Adams	2018	Fall	3,375,000	425,000
	Gorst Creek	2018	Fall	730,000	-
	Grovers Creek	2018	Fall	1,250,000	-
	Hupp Springs	2018	Spring	-	400,000
	Lummi Sea Ponds	2018	Fall	500,000	-
	Minter Creek	2018	Fall	1,250,000	-
Lake Washington	Salmon in the Schools	2018	Fall	-	540
	Issaquah	2018	Fall	2,000,000	-
Nisqually	Clear Creek	2018	Fall	3,300,000	200,000
	Kalama Creek	2018	Fall	600,000	-
	Nisqually MS	2018	Fall	-	90
Nooksack	Kendall Creek	2018	Spring	800,000	-
	Skookum Creek	2018	Spring	-	1,000,000
Puyallup	Clarks Creek	2018	Fall	400,000	-
	Voights Creek	2018	Fall	1,600,000	-
	White River	2017	Spring	-	55,000
		2018	Spring	-	340,000
San Juan Islands	Glenwood Springs	2018	Fall	725,000	-
Skokomish	McKernan	2018	Fall	-	100,000
Skykomish	Wallace River	2017	Summer	500,000	-
		2018	Summer	800,000	200,000
Stillaguamish	Brenner	2018	Fall	-	200,000
	Whitehorse Pond	2018	Summer	220,000	-
Georgia Strait	Samish	2018	Fall	3,800,000	200,000
Upper Skagit	Marblemount	2018	Spring	387,500	200,000
			Summer	200,000	-
Total Annual Release Number				**36,297,500**	**7,271,130**

Available data on total abundance since 1980 indicate that although abundance trends have fluctuated between positive and negative for individual populations, there are widespread negative trends in natural-origin Chinook salmon spawner abundance across the ESU (Ford 2011a). Productivity remains low in most populations, and hatchery-origin spawners are present in high fractions in most populations outside of the Skagit watershed. Available data now shows that most populations have declined in abundance over the past 7 to 10 years. Further, escapement levels for all populations remain well below the Technical Recovery Team planning ranges for recovery, and most populations are consistently below the spawner-recruit levels identified by the Technical Recovery Team as consistent with recovery (Ford 2011a).

Current estimates of diversity show a decline over the past 25 years, indicating a decline of salmon in some areas and increases in others. Salmon returns to the Whidbey Region increased in abundance while returns to other regions declined. In aggregate, the diversity of the ESU as a whole has been declining over the last 25 years.

The Puget Sound technical recovery team identified 22 extant populations, grouped into five major geographic regions, based on consideration of historical distribution, geographic isolation, dispersal rates, genetic data, life history information, population dynamics, and environmental and ecological diversity.

Status

All Puget Sound Chinook salmon populations are well below escapement abundance levels identified as required for recovery to low extinction risk in the recovery plan. In addition, most populations are consistently below the productivity goals identified in the recovery plan as necessary for recovery. Although trends vary for individual populations across the ESU, most populations have declined in total natural origin recruit abundance since the last status review; and natural origin recruit escapement trends since 1995 are mostly stable. Several of the risk factors identified in the previous status review (Good et al. 2005) are still present, including high fractions of hatchery fish in many populations and widespread loss and degradation of habitat. Although this ESU's total abundance is a greatly reduced from historic levels, recent abundance levels do not indicate that the ESU is at immediate risk of extinction. This ESU remains relatively well distributed over 22 populations in 5 geographic areas across the Puget Sound. Although current trends are concerning, the available information indicates that this ESU remains at moderate risk of extinction (NMFS 2011a).

Critical Habitat

Critical habitat was designated for the Puget Sound ESU of Chinook salmon on September 2, 2005 (70 FR 52630) and includes 1,683 miles of stream channels, 41 mi^2 of lakes, and 2,182 mi of nearshore marine habitat (Figure 49). The specific PBFs (formerly referred to as PCEs) that characterize these sites are as follows:

1. Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development.
2. Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
3. Freshwater migration corridors free of obstruction with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.
4. Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh-and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.
5. Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.
6. Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

PBF numbers 1, 2, and 3 (above) are entirely within freshwater areas that do not overlap spatially with the NWTT action area. PBFs 4 and 5 are located in estuarine and nearshore marine areas that do overlap spatially with the action area. The features of estuarine areas (#4 above) are essential to conservation because without them juveniles cannot reach the ocean in a timely manner and use the variety of habitats that allow them to avoid predators, compete successfully, and complete the behavioral and physiological changes needed for life in the ocean. Similarly, these features are essential to the conservation of adults because they provide a final source of abundant forage that will provide the energy stores needed to make the physiological transition to fresh water, migrate upstream, avoid predators, and develop to maturity upon reaching spawning areas. The features of nearshore marine areas (#5 above) in Puget Sound are also essential to conservation because without them juveniles cannot successfully transition from natal streams to offshore marine areas. Number 6 (above) describes PBFs in offshore marine areas that are essential for Puget Sound Chinook salmon conservation. However, NMFS has not designated any specific areas based on this PBF.

NMFS excluded some particular DOD sites from Puget Sound Chinook salmon critical habitat designation because the benefits of exclusion outweigh the benefits of inclusion, and exclusion of those areas will not result in the extinction of the species. Most of the affected DOD sites

overlap habitat areas in nearshore zones occupied by Puget Sound Chinook salmon or Hood Canal summer-run chum salmon. The overlap consists of approximately 64 mi (103 km) of shoreline out of the 2,376 mi (3,824 km) of total occupied shoreline for these two ESUs. Freshwater and estuarine overlap areas also include approximately 20 mi (32 km) of stream used by Puget Sound Chinook salmon, outside of the action area. The following Department of Defense areas were not included as critical habitat:

- Naval Restricted Areas: Strait of Juan de Fuca naval air-to-surface weapon range; Strait of Juan de Fuca and Whidbey Island; Admiralty Inlet; Port Gardner Naval Base; Hood Canal; Port Orchard Passage; Sinclair Inlet; Carr Inlet;) Dabob Bay/Whitney Point; Port Townsend/Indian Island/Walan Point.
- Hood Canal and Dabob Bay naval non-explosive torpedo testing area
- Naval Submarine Base, Bangor
- Naval Undersea Warfare Center, Keyport
- Naval Ordnance Center, Port Hadlock (Indian Island)
- Naval Radio Station, Jim Creek
- Naval Fuel Depot, Manchester
- Naval Air Stations: Whidbey Island and Everett
- Bremerton Naval Hospital
- Puget Sound Naval Shipyard
- Naval Submarine Base Bangor security zone
- Army Sites: Fort Lewis; Pier 23; Yakima Training Center

Forestry practices have heavily impacted migration, spawning, and rearing PBFs in the upper watersheds of most rivers systems within critical habitat designated for the Puget Sound Chinook salmon. Degraded PBFs include reduced conditions of substrate supporting spawning, incubation and larval development caused by siltation of gravel; and degraded rearing habitat by removal of cover and reduction in channel complexity. Urbanization and agriculture in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have reduced channel function and connectivity, reduced available floodplain habitat, and affected water quality. Thus, these areas have degraded spawning, rearing, and migration PBFs. Hydroelectric development and flood control also obstruct Puget Sound Chinook salmon migration in several basins. The most functional PBFs are found in northwest Puget Sound: the Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlap with critical habitat designated for the Puget Sound Chinook salmon. However, estuary PBFs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels.

Recovery Goals

The recovery plan consists of two documents: the Puget Sound salmon recovery plan (Shared Strategy for Puget Sound 2007) and a supplement by NMFS (2006c). The recovery plan adopts ESU and population level viability criteria recommended by the Puget Sound Technical Recovery Team (PSTRT) (Ruckelshaus et al. 2002). The PSTRT's biological recovery criteria will be met when all of the following conditions are achieved:

- The viability status of all populations in the ESU is improved from current conditions, and when considered in the aggregate, persistence of the ESU is assured;
- Two to four Chinook salmon populations in each of the five biogeographical regions of the ESU achieve viability, depending on the historical biological characteristics and acceptable risk levels for populations within each region;
- At least one population from each major genetic and life history group historically present within each of the five biogeographical regions is viable;
- Tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations are functioning in a manner that is sufficient to support an ESU-wide recovery scenario; Production of Chinook salmon from tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations occurs in a manner consistent with ESU recovery; and
- Populations that do not meet the viability criteria for all VSP parameters are sustained to provide ecological functions and preserve options for ESU recovery.

6.2.13 Chinook Salmon – Sacramento River Winter-Run ESU

The Sacramento River winter-run Chinook salmon ESU includes winter-run Chinook salmon spawning naturally in the Sacramento River and its tributaries, as well as winter-run Chinook salmon that are part of the conservation hatchery program at the Livingston Stone National Fish Hatchery (Figure 50). The physical attributes of Chinook salmon are discussed in Section 6.2.9. On January 4, 1994, NMFS listed the Sacramento River winter-run ESU of Chinook salmon as Endangered (59 FR 440).

Life History

Winter-run Chinook salmon are unique because they spawn during summer months when air temperatures usually approach their yearly maximum. As a result, winter-run Chinook salmon require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in summer. Adult winter-run Chinook salmon immigration and holding (upstream spawning migration) through the Delta and into the lower Sacramento River occurs from December through July, with a peak during the period extending from January through April (Fish and Service 1995). Winter-run Chinook salmon are sexually immature when upstream migration begins, and they must hold for several months in suitable habitat prior to spawning. Spawning occurs between late-April and mid-August, with a peak in June and July as reported by CDFW annual escapement surveys (2000-2006).

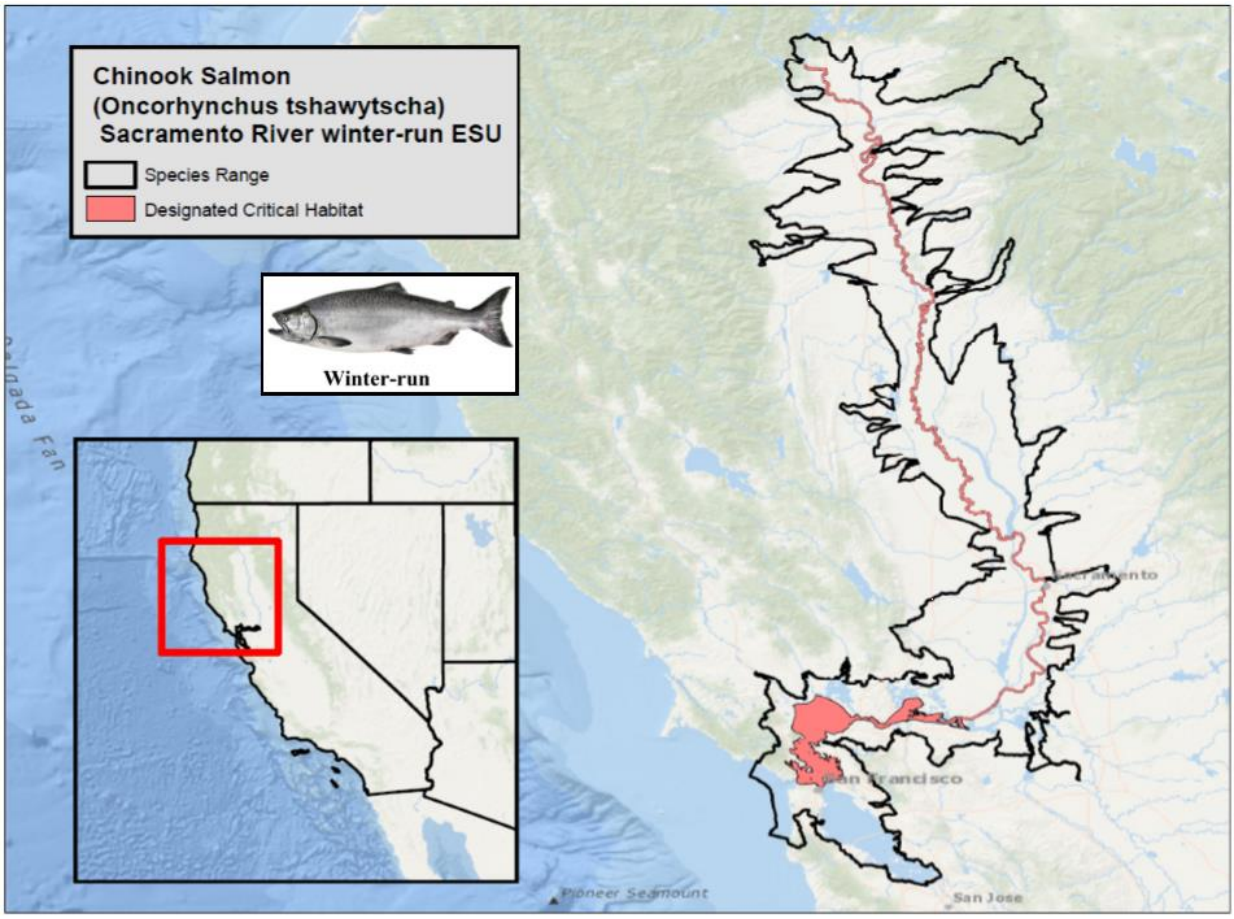


Figure 50. Geographic range and designated critical habitat of the Sacramento River winter-run ESU of Chinook salmon.

Winter-run Chinook salmon embryo incubation in the Sacramento River can extend into October (Vogel et al. 1988). Winter-run Chinook salmon fry rearing in the upper Sacramento River exhibit peak abundance during September, with fry and juvenile emigration past the Red Bluff Diversion Dam primarily occurring from July through November (Poytress and Carrillo 2010; 2011; 2012). Emigration of winter-run Chinook salmon juveniles past Knights Landing, located approximately 155.5 river miles downstream of the Red Bluff Diversion Dam, reportedly occurs between November and March, peaking in December, with some emigration continuing through May in some years (Snider and Titus 2000).

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et

al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Sacramento winter-run ESU of Chinook salmon.

Over the last 10 years of available data (2003-2013), the abundance of spawning winter-run Chinook salmon adults ranged from a low of 738 in 2011 to a high of 17,197 in 2007, with an average of 6,298 (NMFS 2016b). Current abundance estimates for the Sacramento winter-run ESU of Chinook salmon are found in Table 72 below.

Table 72. Average abundance estimates for Sacramento winter-run Chinook salmon natural- and hatchery-origin spawners 2013-2017 (NMFS 2019d).

Production	Life Stage	Abundance
Natural	Adult	210
Natural	Juvenile	195,354
Listed Hatchery Adipose Clip	Adult	2,232
Listed Hatchery Adipose Clip	Juvenile	200,000

The population declined from an escapement of near 100,000 in the late 1960s to fewer than 200 in the early 1990s (Good et al. 2005). More recent population estimates of 8,218 (2004), 15,730 (2005), and 17,153 (2006) show a three-year average of 13,700 returning winter-run Chinook salmon. However, the run size decreased to 2,542 in 2007 and 2,850 in 2008. Monitoring data indicated that approximately 5.6 percent of winter-run Chinook salmon eggs spawned in the Sacramento River in 2014 survived to the fry life stage (three to nearly 10 times lower than in previous years). The drought in 2015 made it another challenging year for winter-run Chinook salmon (NMFS 2016b).

The rising proportion of hatchery fish among returning adults threatens to increase the risk of extinction. Lindley et al. (2007) recommend that in order to maintain a low risk of genetic introgression with hatchery fish, no more than five percent of the naturally-spawning population should be composed of hatchery fish. Since 2001, hatchery origin winter-run Chinook salmon have made up more than five percent of the run, and in 2005 the contribution of hatchery fish exceeded 18 percent (Lindley et al. 2007).

The range of winter-run Chinook salmon has been greatly reduced by Keswick and Shasta dams on the Sacramento River and by hydroelectric development on Battle Creek. Currently, winter-run Chinook salmon spawning is limited to the main-stem Sacramento River between Keswick Dam (River Mile [RM] 302) and the Red Bluff Diversion Dam (RM 243) where the naturally-

spawning population is artificially maintained by cool water releases from the dams. Within the Sacramento River, the spatial distribution of spawners is largely governed by water year type and the ability of the Central Valley Project to manage water temperatures (NMFS 2016b).

Status

The Sacramento River winter-run Chinook salmon ESU is composed of just one small population that is currently under severe stress caused by California's 2011 to 2017 drought, one of California's worst droughts on record. 2013 to 2017 average abundance estimates of natural born adults are estimated to consist of 210 individuals. The population subsists in large part due to agency-managed cold water releases from Shasta Reservoir during the summer and artificial propagation from Livingston Stone National Fish Hatchery's winter-run Chinook salmon conservation program. Winter-run Chinook salmon are dependent on sufficient cold water storage in Shasta Reservoir, and it has long been recognized that a prolonged drought had devastating impacts, possibly leading to the species' extinction. The probability of extended droughts is increasing as the effects of climate change continue (NMFS 2016b). In addition to drought, another important threat to winter-run Chinook salmon is a lack of suitable rearing habitat in the Sacramento River and Delta to allow for sufficient juvenile growth and survival (NMFS 2016b).

Critical Habitat

NMFS designated critical habitat for the Sacramento winter-run Chinook salmon on June 16, 1993 (58 FR 33212). Designated critical habitat for the Sacramento winter-run Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

PBFs that are essential for the conservation of Sacramento winter-run Chinook salmon, based on the best available information, include (1) access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River; (2) the availability of clean gravel for spawning substrate; (3) adequate river flows for successful spawning, incubation of eggs, fry development and emergence, and downstream transport of juveniles; (4) water temperatures between 42.5 and 57.5°F (between 5.8 and 14.1°C) for successful spawning, egg incubation, and fry development; (5) habitat and adequate prey free of contaminants; (6) riparian habitat that provides for successful juvenile development and survival; and (7) access of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean (58 FR 33212).

The current condition of PBFs for the Sacramento River Winter-run Chinook salmon indicates that they are not currently functioning or are degraded. Their conditions are likely to maintain low population abundances across the ESU. Spawning and rearing PBFs are especially degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds where water maintain lower temperatures. The rearing PBF is further degraded by floodplain habitat disconnected from the mainstems of larger rivers throughout the Sacramento River watershed. The migration PBF is also degraded by the lack of natural cover along the

migration corridors. Rearing and migration PBFs are further affected by pollutants entering the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Recovery Goals

Recovery goals, objectives and criteria for the Sacramento River winter-run Chinook salmon are fully outlined in the 2014 Recovery Plan (NMFS 2014c). In order to achieve the downlisting criteria, the species would need to be composed of two populations – one viable and one at moderate extinction risk. Having a second population would improve the species' viability, particularly through increased spatial structure and abundance, but further improvement would be needed to reach the goal of recovery. To delist winter-run Chinook salmon, three viable populations are needed. Thus, the downlisting criteria represent an initial key step along the path to recovering winter-run Chinook salmon.

6.2.14 Chinook Salmon – Snake River Fall-Run ESU

The listed ESU currently includes all natural-origin fall-run Chinook salmon originating from the mainstem Snake River below Hells Canyon Dam (the lowest of three impassable dams that form the Hells Canyon Complex) and from the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins. The listed ESU also includes fall-run Chinook salmon from four artificial propagation programs (NMFS 2011b; NMFS 2015d) (Figure 51). The physical attributes of Chinook salmon are discussed in Section 6.2.9. NMFS first listed Snake River fall Chinook salmon as a threatened species under the ESA on April 22, 1992 (57 FR 14658). NMFS reaffirmed the listing status in June 28, 2005 (70 FR 37160), and reaffirmed the status again in its 2014 (79 FR 20802).

Life History

Snake River fall-run Chinook salmon return to the Columbia River in August and September, pass the Bonneville Dam from mid-August to the end of September, and enter the Snake River between early September and mid-October (DART 2013). Once they reach the Snake River, fall Chinook salmon generally travel to one of five major spawning areas and spawn from late October through early December (Connor et al. 2014).

Upon emergence from the gravel, most young fall Chinook salmon move to shoreline riverine habitat (NMFS 2015d). Some fall Chinook salmon smolts sustain active migration after passing Lower Granite Dam and enter the ocean as subyearlings, whereas some delay seaward migration and enter the ocean as yearlings (Connor et al. 2005; McMichael et al. 2008; NMFS 2015d). Snake River fall Chinook salmon can be present in the estuary as juveniles in winter, as fry from March to May, and as fingerlings throughout the summer and fall (Fresh et al. 2005; Roegner et al. 2012; Teel et al. 2014).

Once in the Northern California Current, dispersal patterns differ for yearlings and subyearlings. Subyearlings migrate more slowly, are found closer to shore in shallower water, and do not disperse as far north as yearlings (Fisher et al. 2014; Sharma and Quinn 2012; Trudel et al. 2009a; Tucker et al. 2011). Snake River basin fall Chinook salmon spend one to four years in the Pacific Ocean, depending on gender and age at the time of ocean entry (Connor et al. 2005).

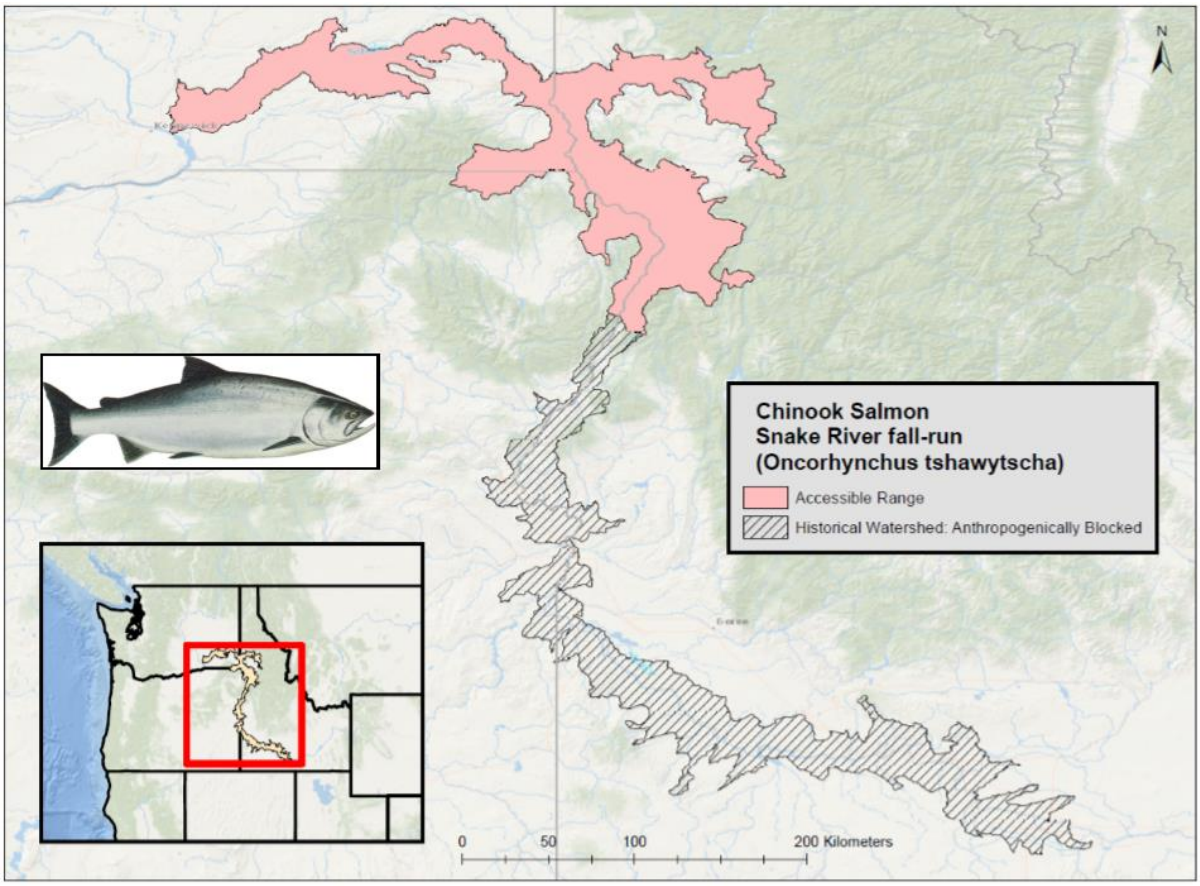


Figure 51. Geographic range of Snake River fall-run ESU Chinook salmon.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Snake River fall-run ESU of Chinook salmon.

The naturally spawning fall Chinook salmon in the lower Snake River have included both returns originating from naturally spawning parents and from returning hatchery releases. The geometric mean natural-origin adult abundance from 2005 to 2014 of annual spawner escapement estimates was 6,418, with a standard error of 0.19 (NMFS 2015d). Current abundance estimates for the Snake River fall-run ESU of Chinook salmon are presented in Table 73 below.

Table 73. Average Abundance Estimates for the Snake River Fall-Run ESU of Chinook salmon from 2015 to 2019 (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	10,337
Natural	Juvenile	692,819
Listed Hatchery Adipose Clip	Adult	12,508
Listed Hatchery Adipose Clip	Juvenile	2,483,713
Listed Hatchery Intact Adipose	Adult	13,551
Listed Hatchery Intact Adipose	Juvenile	2,862,418

Past estimates of productivity for this population (1990-2009 brood years) was 1.53 with a standard error of 0.18. This estimate of productivity, however, may be problematic for two reasons: (1) the increasingly small number of years that actually contribute to the productivity estimate means that there is increasing statistical uncertainty surrounding that estimate, and (2) the years contributing to the estimate are now far in the past and may not accurately reflect the true productivity of the current population NMFS (2015d).

Genetic samples from the aggregate population in recent years indicate that composite genetic diversity is being maintained and that the Snake River Fall Chinook salmon hatchery stock is similar to the natural component of the population, an indication that the actions taken to reduce the potential introgression of out-of-basin hatchery strays has been effective. Overall, the current genetic diversity of the population represents a change from historical conditions and, applying the Interior Columbia Technical Recovery Team (ICTRT) McClure et al. (2005) guidelines, the rating for this metric is moderate risk (NMFS 2015d).

The ICTRT identified three populations of this species, although only the lower mainstem population exists at present, and it spawns in the lower main stem of the Clearwater, Imnaha, Grande Ronde, Salmon, and Tucannon rivers. The extant population of Snake River fall-run

Chinook salmon is the only remaining population from an historical ESU that also included large mainstem populations upstream of the current location of the Hells Canyon Dam complex (ICTRT 2003; McClure et al. 2005). The population is at moderate risk for diversity and spatial structure (Ford 2011a).

Status

As late as the late 1800s, approximately 408,500 to 536,180 fall Chinook salmon are believed to have returned annually to the Snake River. The run began to decline in the late 1800s and then continued to decline through the early and mid-1900s as a result of overfishing and other human activities, including the construction of major dams. This ESU has one extant population. The extant population is at moderate risk for both diversity and spatial structure and abundance and productivity (NMFS 2016j). The overall viability rating for this population is 'viable.' Overall, the status of Snake River fall Chinook salmon has clearly improved compared to the time of listing and compared to prior status reviews. The single extant population in the ESU is currently meeting the criteria for a rating of 'viable' developed by the ICTRT, but the ESU as a whole is not meeting the recovery goals described in the recovery plan for the species, which require the single population to be "highly viable with high certainty" and/or will require reintroduction of a viable population above the Hells Canyon Dam complex (NMFS 2016j).

Critical Habitat

NMFS designated critical habitat for Snake River fall-run Chinook salmon on December 28, 1993 (58 FR 68543). Designated critical habitat for the Snake River fall-run Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

PBFs considered essential for the conservation of Chinook salmon, Snake River fall-run ESU are: 1) juvenile rearing areas which include adequate spawning gravel, water quality, water quantity, water temperature, cover/shelter, food, riparian vegetation, space; and 2) juvenile and adult migration corridors with adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food (juveniles only), riparian vegetation, space, safe passage conditions.

Water quality impairments in the designated critical habitat are common within the range of this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine sediments from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary; traveling along with contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue.

Recovery Goals

Recovery goals, objectives and criteria for the Snake River fall-run Chinook salmon are fully outlined in the 2015 Recovery Plan (NMFS 2015d). ESA recovery goals should support conservation of natural fish and the ecosystems upon which they depend. Thus, the ESA recovery goal for Snake River fall Chinook salmon is that: the ecosystems upon which Snake River fall Chinook salmon depend are conserved such that the ESU is self-sustaining in the wild and no longer needs ESA protection.

6.2.15 Chinook Salmon – Snake River Spring/Summer-Run ESU

The Snake River spring/summer Chinook salmon ESU includes all naturally spawned populations of spring/summer Chinook salmon in the mainstem Snake River and the Tucannon River, Grand Ronde River, Imnaha River, and Salmon River subbasins (Figure 52). The ESU is broken into five major population groups (MPG). Together, the MPGs contain 28 extant independent naturally spawning populations, three functionally extirpated populations, and one extirpated population. The Upper Salmon River MPG contains eight extant populations and one extirpated population. The Middle Fork Salmon River MPG contains nine extant populations. The South Fork Salmon River MPG contains four extant populations. The Grande Ronde/Imnaha Rivers MPG contains six extant populations, with two functionally extirpated populations. The Lower Snake River MPG contains one extant population and one functionally extirpated population. The South Fork and Middle Fork Salmon Rivers currently support most of the natural spring/summer Chinook salmon production in the Snake River drainage (NMFS 2016j). The physical attributes of Chinook salmon are discussed in Section 6.2.9. Snake River spring/summer-run Chinook salmon, an ESU was listed as a threatened species under the ESA on April 22, 1992 (57 FR 14658). NMFS reaffirmed the listing on June 28, 2005 (70 FR 37160) and made minor technical corrections to the listing on April 14, 2014 (79 FR 20802).

Life History

Adult spring-run Chinook salmon destined for the Snake River return to the Columbia River from the ocean in early spring and pass Bonneville Dam beginning in early March and ending May 31st. Snake River summer-run Chinook salmon return to the Columbia River from June through July. Adults from both runs hold in deep pools in the mainstem Columbia and Snake Rivers and the lower ends of the spawning tributaries until late summer, when they migrate into the higher elevation spawning reaches. Generally, Snake River spring-run Chinook salmon spawn in mid- through late August. Snake River summer-run Chinook salmon spawn approximately one month later than spring-run fish and tend to spawn lower in the tributary drainages, although their spawning areas often overlap with those of spring-run spawners.

The eggs that Snake River spring and summer Chinook salmon deposit in late summer and early fall incubate over the following winter, and hatch in late winter and early spring. Juveniles rear through the summer, overwinter, and typically migrate to sea in the spring of their second year of

life, although some juveniles may spend an additional year in fresh water. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Most yearling fish are thought to spend relatively little time in the estuary compared to sub-yearling ocean-type fish however there is considerable variation in residence times in different habitats and in the timing of estuarine and ocean entry among individual fish (Holsman et al. 2012; McElhany et al. 2000).

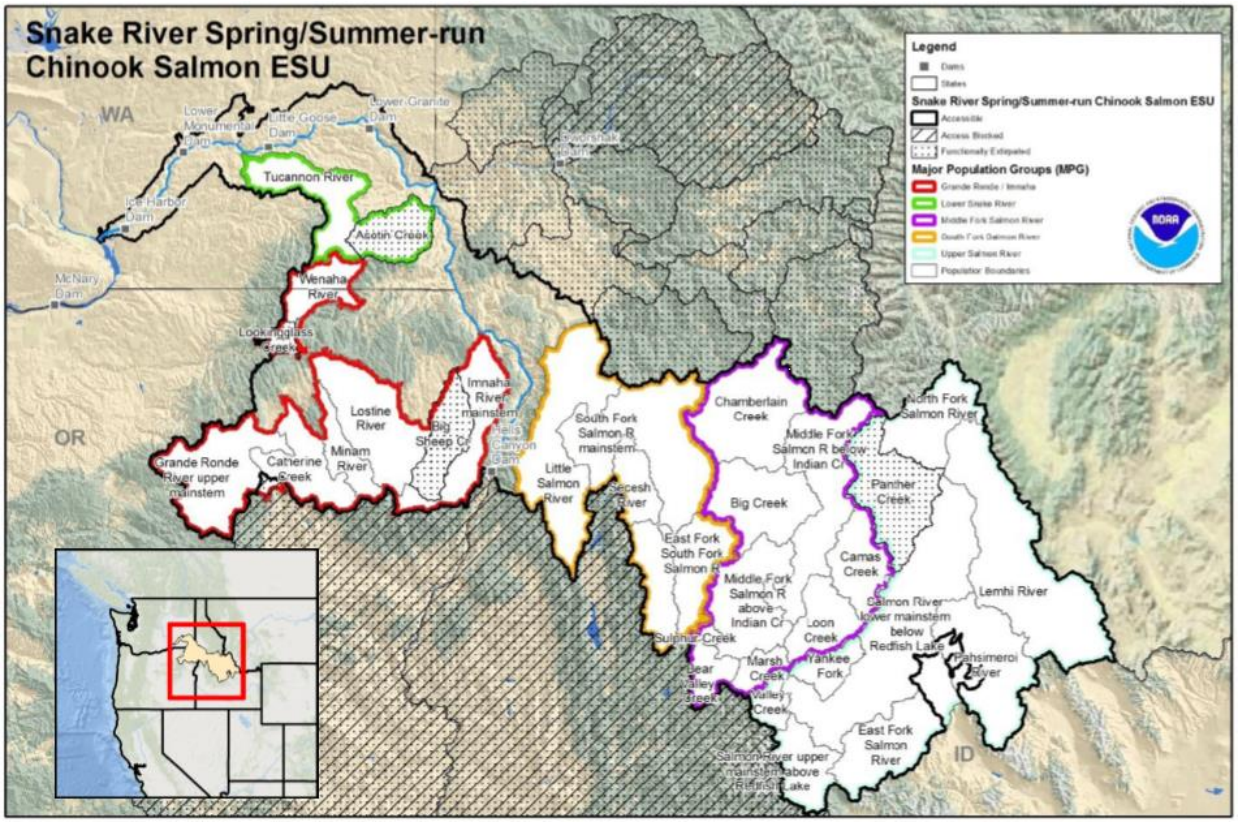


Figure 52. Geographic range and major population groups of Snake River spring/summer-run ESU Chinook salmon.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, and genetic diversity as it relates major population groups (MPGs) within the Snake River spring/summer-run ESU of Chinook salmon. Current

abundance estimates of the Snake River spring/summer-run ESU of Chinook salmon are presented in Table 74 below.

Table 74. Average Abundance Estimates for the Snake River Spring/Summer-Run ESU of Chinook salmon for 2014-2018 (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	12,798
Natural	Juvenile	1,296,641
Listed Hatchery Adipose Clip	Adult	2,387
Listed Hatchery Adipose Clip	Juvenile	4,760,250
Listed Hatchery Intact Adipose	Adult	421
Listed Hatchery Intact Adipose	Juvenile	868,679

Lower Snake River MPG: Abundance and productivity remain the major concern for the Tucannon River population. Natural spawning abundance (10-year geometric mean) has increased but remains well below the minimum abundance threshold for the single extant population in this MPG. Poor natural productivity continues to be a major concern. The integrated spatial structure/diversity risk rating for the Lower Snake River MPG is moderate (NMFS 2016j).

Grande Ronde/Imnaha MPG: The Wenaha River, Lostine/Wallowa River and Minam River populations showed substantial increases in natural abundance relative to the previous ICTRT review, although each remains below their respective minimum abundance thresholds. The Catherine Creek and Upper Grande Ronde populations each remain in a critically depressed state. Geometric mean productivity estimates remain relatively low for all populations in the MPG. The Upper Grande Ronde population is rated at high risk for spatial structure and diversity while the remaining populations are rated at moderate (NMFS 2016j).

South Fork Salmon River MPG: Natural spawning abundance (10-year geometric mean) estimates increased for the three populations with available data series. Productivity estimates for these populations are generally higher than estimates for populations in other MPGs within the ESU. Viability ratings based on the combined estimates of abundance and productivity remain at high risk, although the survival/capacity gaps relative to moderate and low risk viability curves are smaller than for other ESU populations. Spatial structure/diversity risks are currently rated moderate for the South Fork Mainstem population (relatively high proportion of hatchery spawners) and low for the Secesh River and East Fork South Fork populations (NMFS 2016j).

Middle Fork Salmon River MPG: Natural-origin abundance and productivity remains extremely low for populations within this MPG. As in the previous ICTRT assessment, abundance and productivity estimates for Bear Valley Creek and Chamberlain Creek (limited data series) are the

closest to meeting viability minimums among populations in the MPG. Spatial structure/diversity risk ratings for Middle Fork Salmon River MPG populations are generally moderate. This primarily is driven by moderate ratings for genetic structure assigned by the ICTRT because of uncertainty arising from the lack of direct genetic samples from within the component populations (NMFS 2016j).

Upper Salmon River MPG: Abundance and productivity estimates for most populations within this MPG remain at very low levels relative to viability objectives. The Upper Salmon Mainstem has the highest relative abundance and productivity combination of populations within the MPG. Spatial structure/diversity risk ratings vary considerably across the Upper Salmon River MPG. Four of the eight populations are rated at low or moderate risk for overall spatial structure and diversity and could achieve viable status with improvements in average abundance/productivity. The high spatial structure/diversity risk rating for the Lemhi population is driven by a substantial loss of access to tributary spawning/rearing habitats and the associated reduction in life-history diversity. High risk ratings for Pahsimeroi River, East Fork Salmon River, and Yankee Fork Salmon River are driven by a combination of habitat loss and diversity concerns related to low natural abundance combined with chronically high proportions of hatchery spawners in natural areas (NMFS 2016j).

Status

The historical run of Chinook salmon in the Snake River likely exceeded one million fish annually in the late 1800s, by the 1950s the run had declined to nearly 100,000 adults per year. The adult counts fluctuated throughout the 1980s but then declined further, reaching a low of 2,200 fish in 1995. Currently, the majority of extant spring/summer Chinook salmon populations in the Snake River spring/summer Chinook salmon ESU remain at high overall risk of extinction, with a low probability of persistence within 100 years. Factors cited in the 1991 status review as contributing to the species' decline since the late 1800s include overfishing, irrigation diversions, logging, mining, grazing, obstacles to migration, hydropower development, and questionable management practices and decisions (Matthews and Waples 1991). In addition, new threats such as those posed by toxic contamination, increased predation by non-native species, and effects due to climate change are emerging (NMFS 2016j). Hinrichsen and Paulsen (2020) estimated carrying capacity and 24-year extinction probabilities for 26 populations in the Snake River spring/summer Chinook salmon ESU using alternative quasi-extinction thresholds. They found that carrying capacities estimates were low in several of the populations and that extinction probability increases sharply with decreasing carrying capacity.

Critical Habitat

Critical habitat for Snake River spring/summer Chinook salmon was designated on December 28, 1993 (58 FR 68543) and revised slightly on October 25, 1999 (64 FR 57399). Designated critical habitat for the Snake River spring/summer Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

PBFs considered essential for the conservation of Chinook salmon, Snake River spring/summer ESU are: 1) juvenile rearing areas which include adequate spawning gravel, water quality, water quantity, water temperature, cover/shelter, food, riparian vegetation, space; and 2) juvenile and adult migration corridors with adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food (juveniles only), riparian vegetation, space, safe passage conditions. Spawning and juvenile rearing PBFs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access that has resulted from altered flow regimes from hydroelectric dams. According to the ICTRT, the Panther Creek population was extirpated because of legacy and modern mining-related pollutants creating a chemical barrier to fish passage (Chapman and Julius 2005).

Recovery Goals

Recovery goals, scenarios and criteria for the Snake River spring and summer-run Chinook salmon are fully outlined in the 2016 proposed recovery plan (NMFS 2016j). The status levels targeted for populations within an ESU or DPS are referred to collectively as the "recovery scenario" for the ESU or DPS. NMFS has incorporated the viability criteria into viable recovery scenarios for each Snake River spring/summer Chinook salmon and steelhead MPG. The criteria should be met for an MPG to be considered Viable, or low (5 percent or less) risk of extinction, and thus contribute to the larger objective of ESU or DPS viability.

6.2.16 Chinook Salmon – Upper Columbia River Spring-Run ESU

The Snake River spring/summer Chinook salmon ESU includes all naturally spawned populations of spring/summer Chinook salmon in the mainstem Snake River and the Tucannon River, Grand Ronde River, Imnaha River, and Salmon River subbasins as well as spring/summer Chinook salmon from 11 artificial propagation programs (NMFS 2016j) (Figure 53). The physical attributes of Chinook salmon are discussed in Section 6.2.9. Upper Columbia River spring-run Chinook salmon, an ESU was listed as an endangered species under the ESA on March 24, 1999 (64 FR 14308). NMFS reaffirmed the listing on June 28, 2005 (70 FR 37160).

Life History

Adult Spring Chinook salmon in the Upper Columbia Basin begin returning from the ocean in the early spring, with the run into the Columbia River peaking in mid-May. Spring Chinook salmon enter the Upper Columbia tributaries from April through July. After migration, they hold in freshwater tributaries until spawning occurs in the late summer, peaking in mid to late August. Juvenile spring Chinook salmon spend a year in freshwater before migrating to salt water in the spring of their second year of life. Most Upper Columbia spring Chinook salmon return as adults after two or three years in the ocean. Some precocious males, or jacks, return after one winter at sea. A few other males mature sexually in freshwater without migrating to the sea. However, four and five year old fish that have spent two and three years at sea, respectively, dominate the run. Fecundity ranges from 4,200 to 5,900 eggs, depending on the age and size of the female.

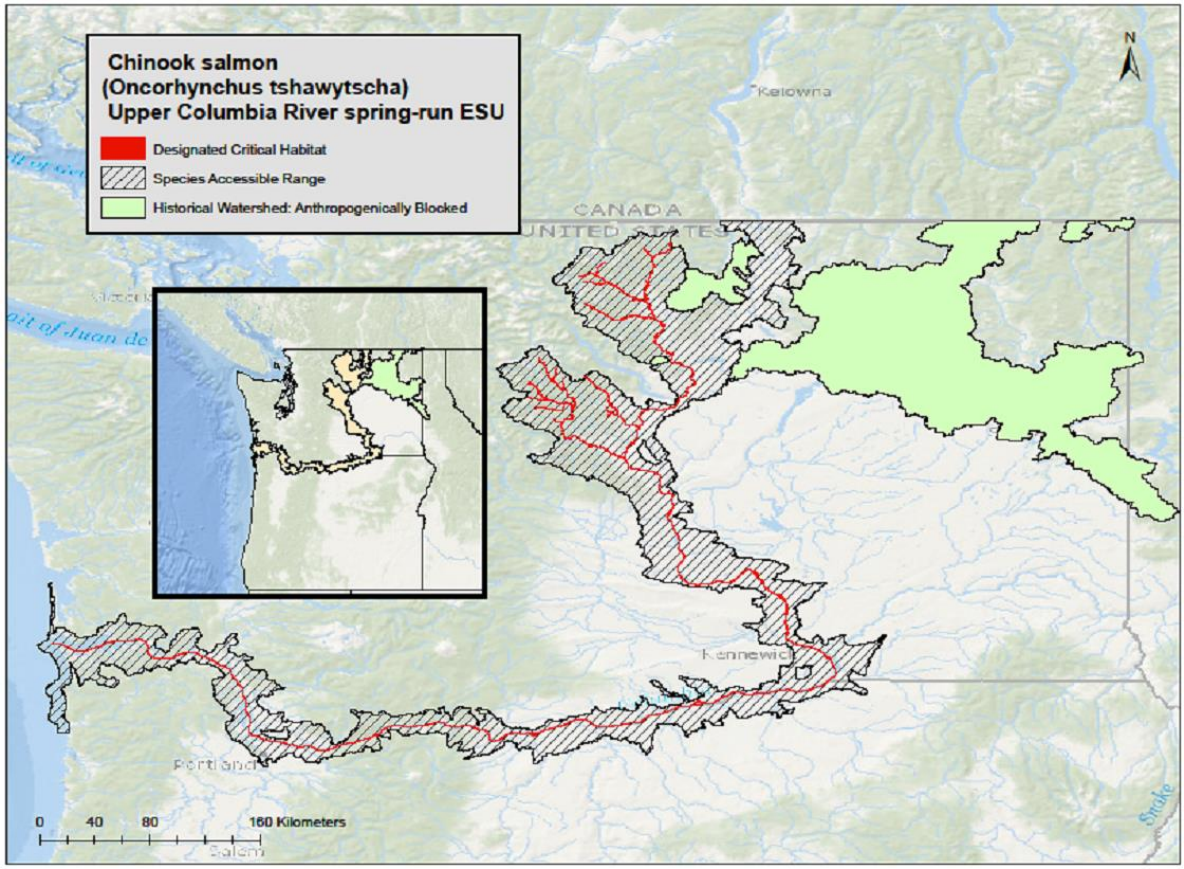


Figure 53. Geographic range and designated critical habitat of Chinook salmon, upper Columbia River ESU.

Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the upper Columbia River spring-run ESU of Chinook salmon.

For all populations, average abundance over the recent 10-year period is below the average abundance thresholds that the ICTRT identifies as a minimum for low risk (ICTRT 2008a; 2008b; 2008c). The geometric mean spawning escapements from 1997 to 2001 were 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population. These

numbers represent only 8 percent to 15 percent of the minimum abundance thresholds. The 10-year geometric mean abundance of adult natural-origin spawners has increased for each population relative to the levels reported in the 2011 status review, but natural origin escapements remain below the corresponding ICTRT thresholds. Current abundance estimates of the upper Columbia River spring-run ESU of Chinook salmon are presented in Table 75 below.

Table 75. Five Year Average (2015 to 2020) Abundance Estimates for the Upper Columbia River Spring-Run ESU of Chinook salmon (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	2,872
Natural	Juvenile	468,820
Listed Hatchery Adipose Clip	Adult	6,226
Listed Hatchery Adipose Clip	Juvenile	621,759
Listed Hatchery Intact Adipose	Adult	3,364
Listed Hatchery Intact Adipose	Juvenile	368,642

Overall abundance and productivity remains rated at high risk for each of the three extant populations in this MPG/ESU (NWFSC 2015b). The Short term lambda estimate for the Wenatchee River is 0.60; the Entiat River is 0.94; and the Methow River is 0.46.

The ICTRT characterizes the diversity risk to all UCR Spring-run Chinook salmon populations as "high". The high risk is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project in 1939-1943.

Spring Chinook salmon currently spawn and rear in the upper main Wenatchee River upstream from the mouth of the Chiwawa River, overlapping with summer Chinook salmon in that area (Peven et al. 1994). The primary spawning areas of spring Chinook salmon in the Wenatchee subbasin include Nason Creek and the Chiwawa, Little Wenatchee, and White rivers. The current spawning distribution for spring Chinook salmon in the Entiat subbasin has been described as the Entiat River (river mile 16.2 to 28.9) and the Mad River (river mile 32 1.5-5.0) (NMFS 2007b). Spring Chinook salmon of the Methow population currently spawn in the mainstem Methow River and the Twisp, Chewuch, and Lost drainages (NMFS 2007b). A few also spawn in Gold, Wolf, and Early Winters creeks.

Status

This ESU comprises four independent populations. Three are at high risk and one is functionally extirpated. Current estimates of natural origin spawner abundance increased relative to the levels observed in the prior review for all three extant populations, and productivities were higher for the Wenatchee and Entiat populations and unchanged for the Methow population. However,

abundance and productivity remained well below the viable thresholds called for in the Upper Columbia Recovery Plan for all three populations. Although the status of the ESU is improved relative to measures available at the time of listing, all three populations remain at high risk (NWFSC 2015b).

Critical Habitat

NMFS designated critical habitat for Upper Columbia River Spring-run Chinook salmon on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. Designated critical habitat for the Upper Columbia River Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Spawning and rearing PBFs are somewhat degraded in tributary systems by urbanization in lower reaches, grazing in the middle reaches, and irrigation and diversion in the major upper drainages. These activities have resulted in excess erosion of fine sediment and silt that smother spawning gravel; reduction in flow quantity necessary for successful incubation, formation of physical rearing conditions, and juvenile mobility. Moreover siltation further affects critical habitat by reducing water quality through contaminated agricultural runoff; and removing natural cover. Adult and juvenile migration PBFs are heavily degraded by Columbia River Federal dam projects and a number of mid-Columbia River Public Utility District dam projects also obstruct the migration corridor.

Recovery Goals

Recovery goals, objectives and detailed criteria for the Central Valley spring-run Chinook salmon are fully outlined in the 2016 Recovery Plan. The general recovery objectives are:

- Increase the abundance of naturally produced spring Chinook salmon spawners within each population in the Upper Columbia ESU to levels considered viable.
- Increase the productivity (spawner ratios and smolts/redds[18]) of naturally produced spring Chinook salmon within each population to levels that result in low risk of extinction.
- Restore the distribution of naturally produced spring Chinook salmon to previously occupied areas (where practical) and allow natural patterns of genetic and phenotypic diversity to be expressed.

6.2.17 Chinook Salmon – Upper Willamette River ESU

This evolutionarily significant unit, or ESU, includes naturally spawned spring-run Chinook salmon originating from the Clackamas River and from the Willamette River and its tributaries

[18] gravel nests excavated by spawning females.

above Willamette Falls (Figure 54). Also, the Upper Willamette River spring-run ESU of Chinook salmon originate from six artificial propagation programs.

The physical attributes of Chinook salmon are discussed in Section 6.2.9. The upper Willamette River spring-run Chinook salmon ESU was listed as an endangered species under the ESA on March 24, 1999 (64 FR 14308). NMFS reaffirmed the listing on June 28, 2005 (70 FR 37160).

Life History

Upper Willamette River Chinook salmon exhibit an earlier time of entry into the Columbia River than other spring-run Chinook salmon ESUs (Myers et al. 1998b). Adults appear in the lower Willamette River in February, but the majority of the run ascends Willamette Falls in April and May, with a peak in mid- to late May. However, present-day salmon ascend the Willamette Falls via a fish ladder. Consequently, the migration of spring Chinook salmon over Willamette Falls extends into July and August (overlapping with the beginning of the introduced fall-run of Chinook salmon).

The adults hold in deep pools over summer and spawn in late fall or early winter when winter storms augments river flows. Fry may emerge from February to March and sometimes as late as June (Myers et al. 2006). Juvenile migration varies with three distinct juvenile emigration "runs": fry migration in late winter and early spring; sub-yearling (0 yr +) migration in fall to early winter; and yearlings (1 yr +) migrating in late winter to spring. Sub-yearlings and yearlings rear in the mainstem Willamette River where they also use floodplain wetlands in the lower Willamette River during the winter-spring floodplain inundation period. Juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002). Upon reaching the ocean, juvenile Chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

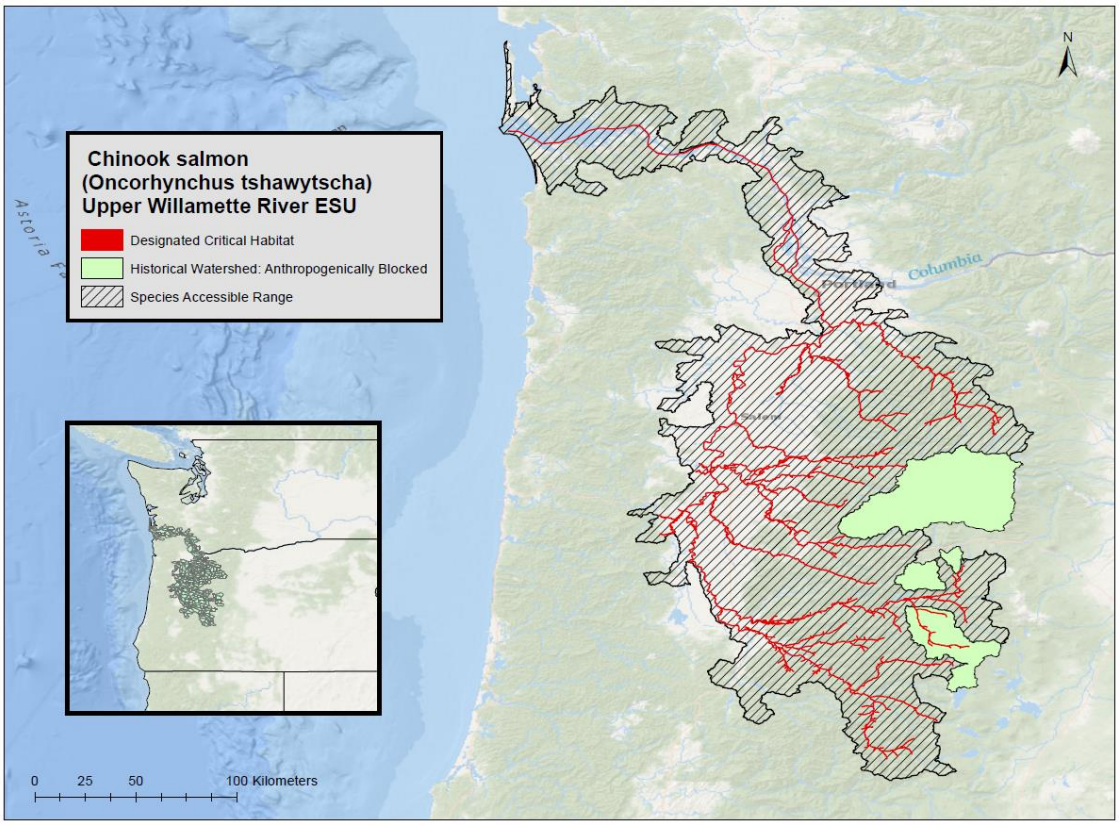


Figure 54.Geographic range and designated critical habitat of Chinook salmon, upper Willamette River ESU.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the upper Willamette River ESU of Chinook salmon.

Abundance levels for five of the seven DIPs in this ESU remain well below their recovery goals. Of these, the Calapooia River may be functionally extinct and the Molalla River remains critically low (although perhaps only marginally better than the 0 VSP score estimated in the Recovery Plan; ODFW and NMFS (2011)). Abundances in the North and South Santiam rivers have risen since the 2010 review, but still range only in the high hundreds of fish. The proportion of natural origin spawners improved in the North and South Santiam basins, but was still well below identified recovery goals. Improvement in the status of the Middle Fork Willamette River relates solely to the return of natural adults to Fall Creek, however the capacity of the Fall Creek basin alone is insufficient to achieve the recovery goals for this DIP. The Clackamas and McKenzie Rivers have previously been viewed as natural population strongholds, but have both experienced declines in abundance despite having access to much of their historical spawning habitat. Overall, populations appear to be at either moderate or high risk, there has been likely little net change in the VSP score for the ESU since the last review, so the ESU remains at

moderate risk (NWFSC 2015b). Current abundance estimates of the Upper Willamette River spring-run ESU of Chinook salmon are presented in Table 76 below.

Table 76. Average Abundance Estimates for the Upper Willamette River Spring-Run ESU of Chinook salmon from 2014 to 2018 for Adults and 2015 to 2020 for Juveniles (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	10,203
Natural	Juvenile	1,211,863
Listed Hatchery Clipped and Intact Adipose	Adult	31,476
Listed Hatchery Adipose Clip	Juvenile	4,709,045
Listed Hatchery Intact Adipose	Juvenile	157

Access of fall-run Chinook salmon to the upper Willamette River and the mixing of hatchery stocks within the ESU have threatened the genetic integrity and diversity of the species. Much of the genetic diversity that existed between populations has been homogenized (Myers et al. 2006).

Radio-tagging results from 2014 suggest that few fish strayed into west-side tributaries (no detections) and relatively fewer fish were unaccounted for between Willamette Falls and the tributaries, 12.9 percent of clipped fish and 5.3 percent of unclipped fish (NWFSC 2015b). In contrast to most of the other populations in this ESU, McKenzie River Chinook salmon have access to much of their historical spawning habitat, although access to historically high quality habitat above Cougar Dam (South Fork McKenzie River) is still limited by poor downstream juvenile passage. Similarly, natural-origin returns to the Clackamas River have remained flat, despite adults having access to much of their historical spawning habitat.

Status

The Upper Willamette River Chinook salmon ESU is considered to be extremely depressed, likely numbering less than 10,000 fish compared to a historical abundance estimate of 300,000 (NMFS 2011f). There are seven demographically independent populations of spring-run Chinook salmon in the UWR Chinook salmon ESU: Clackamas, Molalla, North Santiam, South Santiam, Calapooia, McKenzie, and the Middle Fork Willamette (NMFS 2011f). The Clackamas and McKenzie Rivers have previously been viewed as natural population strongholds, but have both experienced declines in abundance despite having access to much of their historical spawning habitat. Juvenile spring Chinook salmon produced by hatchery programs are released throughout many of the subbasins and adult Chinook salmon returns to the ESU are typically 80-90 percent hatchery origin fish. Access to historical spawning and rearing areas is restricted by

large dams in the four historically most productive tributaries, and in the absence of effective passage programs will continue to be confined to more lowland reaches where land development, water temperatures, and water quality may be limiting. Pre-spawning mortality levels are generally high in the lower tributary reaches where water temperatures and fish densities are generally the highest.

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors and estuarine areas. Designated critical habitat for the Upper Willamette River Chinook salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival.

The current condition of PBFs of the UWR Chinook salmon critical habitat indicates that migration and rearing PBFs are not currently functioning or are degraded. The migration PBF is degraded by dams altering migration timing and water management altering the water quantity necessary for mobility and survival. Migration, rearing, and estuary PBFs are also degraded by loss of riparian vegetation and instream cover. Degraded water quality in the lower Willamette River where important floodplain rearing habitat is present affects the ability of this habitat to sustain its role to conserve the species.

Recovery Goals

Recovery goals, objectives and detailed criteria for the Upper Willamette River Chinook salmon are fully outlined in the 2011 Recovery Plan (NMFS 2011f). The 2011 recovery plan outlines five potential scenario options for meeting the viability criteria for recovery. Of the five scenarios, "scenario one" reportedly represented the most balanced approach given limitations in some populations. The approach in this scenario is to recover the McKenzie (core and genetic legacy population) and the Clackamas populations to an extinction risk status of very low risk (beyond minimal viability thresholds), to recover the North Santiam and Middle Fork Willamette populations (core populations) to an extinction risk status of low risk, to recover the South Santiam population to moderate risk, and improve the status of the remaining populations from very high risk to high risk.

6.2.18 Chum Salmon – Columbia River ESU

The Columbia River ESU of chum salmon includes naturally spawned chum salmon originating from the Columbia River and its tributaries in Washington and Oregon (Figure 55), and also chum salmon from two artificial propagation programs.

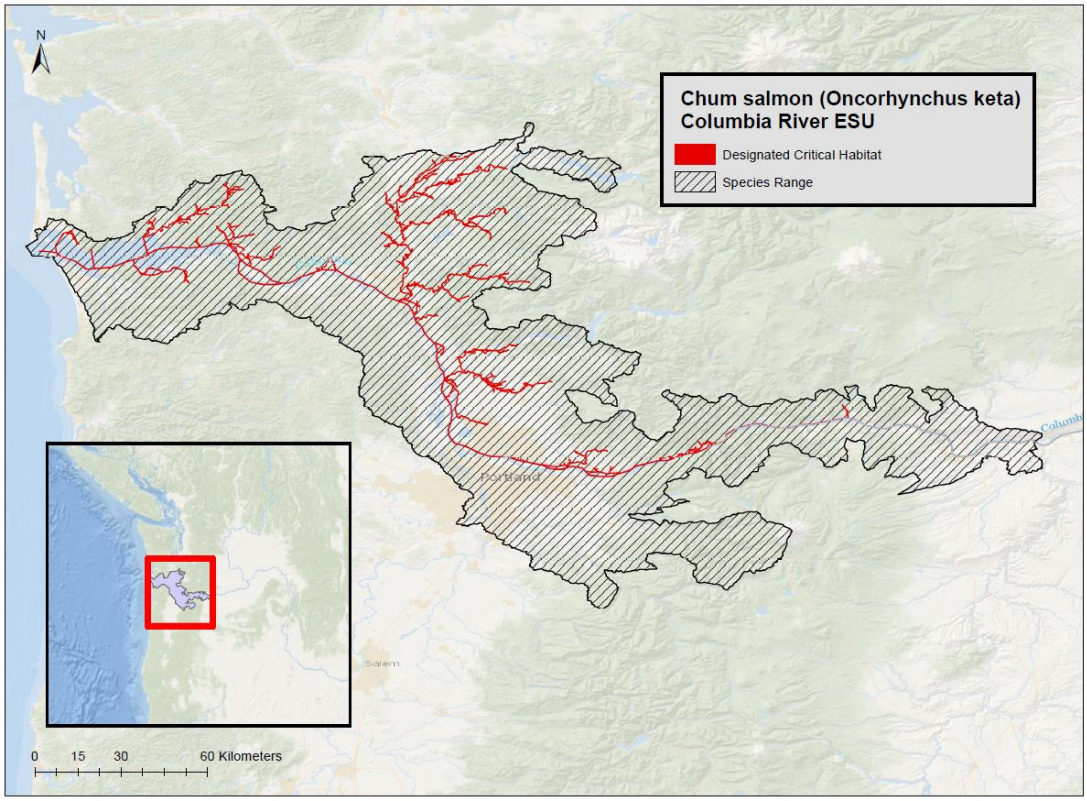


Figure 55. Geographic range and designated critical habitat of chum salmon, Columbia River ESU.

Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between eight and fifteen pounds, but they can get as large as 45 pounds and 3.6 feet long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning. Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with black speckles. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids. On March 25, 1999, NMFS listed the Hood Canal summer-run ESU and the Columbia River ESU of chum salmon as threatened (64 FR 14508). NMFS reaffirmed the status of these two ESUs as threatened on June 28, 2005 (70 FR 37160).

Life History

Most chum salmon mature and return to their birth stream to spawn between three and five years of age, with 60 to 90 percent of the fish maturing at four years of age. Age at maturity appears to follow a latitudinal trend (i.e., greater in the northern portion of the species' range). Chum salmon typically spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to 100 km from the sea. Juveniles out-

migrate to seawater almost immediately after emerging from the gravel covered redds (Salo 1991). The survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history compared to other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175 E longitude (Johnson et al. 1997). North American chum salmon migrate north along the coast in a narrow band that broadens in southeastern Alaska, although some data suggests that chum salmon may travel directly offshore into the north Pacific Ocean (Johnson et al. 1997).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the chum salmon Columbia River ESU.

Chum salmon populations in the Columbia River historically reached hundreds of thousands to a million adults each year (NMFS 2017a). In the past 50 years, the average has been a few thousand a year. The majority of populations in the Columbia River chum salmon ESU remain at high to very high risk, with very low abundances (NWFSC 2015b). Ford (2011b) concluded that 14 out of 17 of chum salmon populations in this ESU were either extirpated or nearly extirpated. Current abundance estimates of the Columbia River ESU of chum salmon are presented in Table 77 below. To estimate abundance of juvenile CR chum salmon, we calculate the geometric mean for outmigrating smolts over the past five years (2015-2019) by using annual abundance estimates provided by NMFS' Northwest Fisheries Science Center (Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020). For juvenile natural-origin CR chum salmon is juvenile salmon, an estimated average of 6,626,218 outmigrated over the last five years.

Table 77. Abundance Estimates for the Columbia River ESU of Chum salmon (NMFS 2019c; Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020).

Production	Life Stage	Abundance
Natural	Adult	10,644
Natural	Juvenile	6,626,218
Listed Hatchery Intact Adipose	Adult	426
Listed Hatchery Intact Adipose	Juvenile	601,503

Only one population (Grays River) is at low risk, with spawner abundances in the thousands, and demonstrating a recent positive trend. Two other populations (Washougal River and Lower Gorge) maintain moderate numbers of spawners and appear to be relatively stable (NWFSC 2015b). The overall trend since 2000 is negative, with the recent peak in abundance (2010-2011) being considerably lower than the previous peak in 2002.

There are currently four hatchery programs in the Lower Columbia River releasing juvenile chum salmon: Grays River Hatchery, Big Creek Hatchery, Lewis River Hatchery, and Washougal Hatchery (NMFS 2017a). Total annual production from these hatcheries has not exceeded 500,000 fish. All of the hatchery programs in this ESU use integrated stocks developed to supplement natural production. Other populations in this ESU persist at very low abundances and the genetic diversity available would be very low (NWFSC 2015b). Diversity has been greatly reduced at the ESU level because of presumed extirpations and low abundance in the remaining populations (fewer than 100 spawners per year for most populations) (LCFRB 2010; NMFS 2013b).

The Columbia River chum salmon ESU includes all natural-origin chum salmon in the Columbia River and its tributaries in Washington and Oregon. The ESU consists of three populations: Grays River, Hardy Creek and Hamilton Creek in Washington State. Chum salmon from four artificial propagation programs also contribute to this ESU.

Status

The majority of the populations within the Columbia River chum salmon ESU are at high to very high risk, with very low abundances (NWFSC 2015b). These populations are at risk of extirpation due to demographic stochasticity and Allee effects. One population, Grays River, is at low risk, with spawner abundances in the thousands and demonstrating a recent positive trend. The Washougal River and Lower Gorge populations maintain moderate numbers of spawners and appear to be relatively stable. The life history of chum salmon is such that ocean conditions have a strong influence on the survival of emigrating juveniles. The potential prospect of poor ocean conditions for the near future may put further pressure on the Columbia River chum salmon ESU (NWFSC 2015b). Freshwater habitat conditions may be negatively influencing spawning and early rearing success in some basins, and contributing to the overall low productivity of the ESU. Columbia River chum salmon were historically abundant and subject to substantial harvest until the 1950s (NWFSC 2015b). There is no directed harvest of this ESU and the incidental harvest rate has been below one percent for the last five years (NWFSC 2015b). Land development, especially in the low gradient reaches that chum salmon prefer, will continue to be a threat to most chum salmon populations due to projected increases in the population of the greater Vancouver-Portland area and the Lower Columbia River overall (Metro 2015). The Columbia River chum salmon ESU remains at a moderate to high risk of extinction (NWFSC 2015b).

Critical Habitat

NMFS designated critical habitat for the Columbia River chum salmon ESU in 2005 (70 FR 52630). This designation includes defined areas in the following subbasins: Middle Columbia/Hood, Lower Columbia/Sandy, Lewis, Lower Columbia/Clatskanie, Lower Cowlitz, and Lower Columbia sub-basin and river corridor (Figure 55). Columbia River chum salmon critical habitat includes freshwater spawning, freshwater rearing, and freshwater migration areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Designated critical habitat for the Columbia River chum salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The major factors limiting recovery for Columbia River chum salmon are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem. In addition, the PBFs related to water quality and cover for estuary and rearing areas have decreased in quality to the extent that these features are not likely to maintain their intended function to conserve the species.

Recovery Goals

The ESU recovery strategy for Columbia River chum salmon focuses on improving tributary and estuarine habitat conditions, reducing or mitigating hydropower impacts, and reestablishing chum salmon populations where they may have been extirpated (NMFS 2013a). The goal of the strategy is to increase the abundance, productivity, diversity, and spatial structure of chum salmon populations such that the Coast and Cascade chum salmon strata are restored to a high probability of persistence, and the persistence probability of the two Gorge populations improves. For details on Columbia River chum salmon ESU recovery goals, including complete down-listing/delisting criteria, see the NMFS 2013 recovery plan (NMFS 2013a).

6.2.19 Chum Salmon – Hood Canal Summer-Run ESU

Hood Canal summer-run ESU chum include naturally spawned summer-run chum salmon originating from Hood Canal and its tributaries as well as from Olympic Peninsula rivers between Hood Canal and Dungeness Bay (Figure 56). Also, summer-run chum salmon originate from four artificial propagation programs.

Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between eight and fifteen pounds, but they can get as large as 45 pounds and 3.6 feet long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning. Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with

black speckles. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids. On March 25, 1999, NMFS listed the Hood Canal Summer-run ESU and the Columbia River ESU of chum salmon as threatened (64 FR 14508). NMFS reaffirmed the status of these two ESUs as threatened on June 28, 2005 (70 FR 37160).

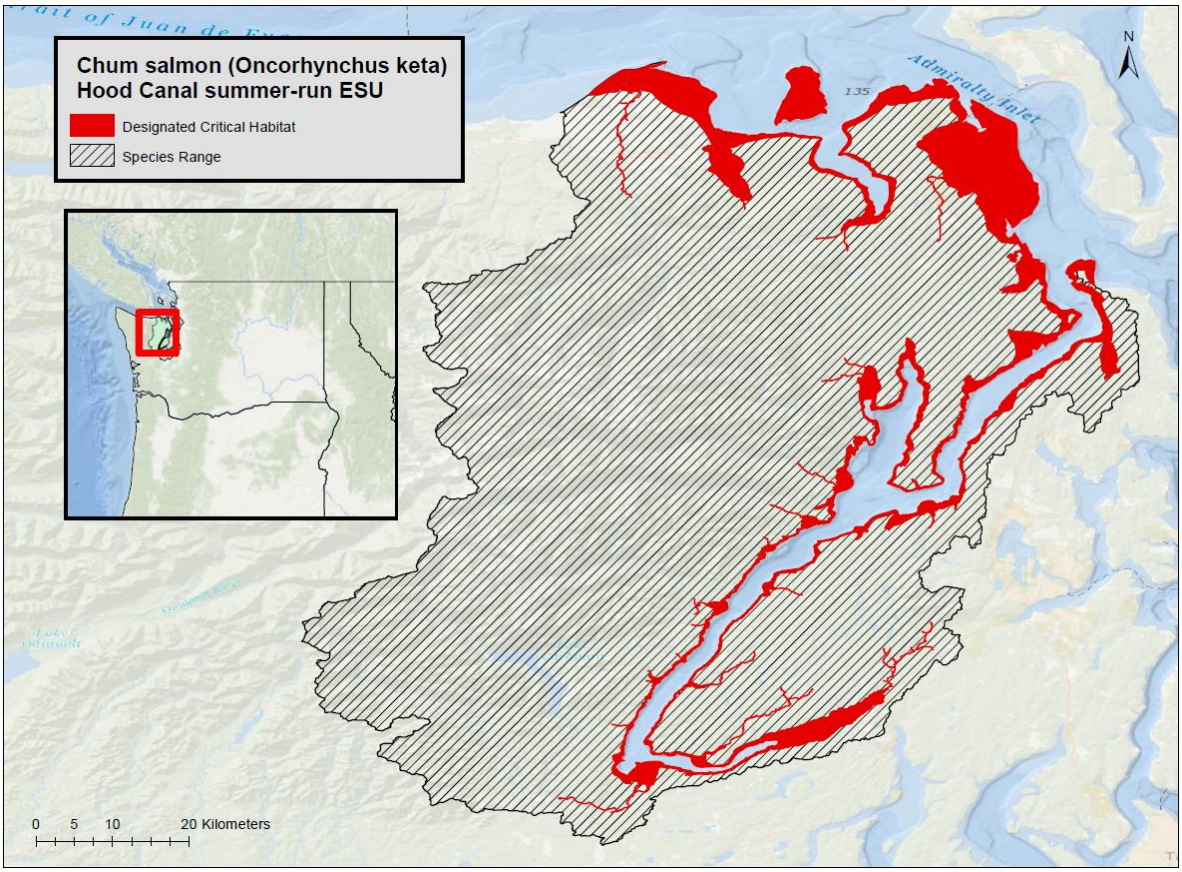


Figure 56. Geographic range and designated critical habitat of chum salmon, Hood Canal ESU.

Life History

Most chum salmon mature and return to their birth stream to spawn between three and five years of age, with 60 to 90 percent of the fish maturing at four years of age. Age at maturity appears to follow a latitudinal trend (i.e., greater in the northern portion of the species' range). Chum salmon typically spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to 100 km from the sea. Juveniles out-migrate to seawater almost immediately after emerging from the gravel covered redds (Salo 1991). The survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable

estuarine conditions. Chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history compared to other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175 E longitude (Johnson et al. 1997). North American chum salmon migrate north along the coast in a narrow band that broadens in southeastern Alaska, although some data suggests that chum salmon may travel directly offshore into the north Pacific Ocean (Johnson et al. 1997).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Hood Canal summer-run ESU of chum salmon.

Of the sixteen populations that comprise the Hood Canal Summer-run chum salmon ESU, seven are considered "functionally extinct" (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek and Chimicum). NMFS examined average escapements (geometric means) for five-year intervals and estimated trends over the intervals for all natural spawners and for natural-origin only spawners. For both populations, abundance was relatively high in the 1970s, lowest for the period 1985-1999, and high again from 2005 to 2015 (NWFSC 2015b). Current abundance estimates of the Hood Canal summer-run ESU of chum salmon are presented in Table 78 and Table 79 below.

Table 78. Hood Canal summer-run juvenile chum salmon hatchery releases (NMFS 2020c).

Subbasin	Artificial propagation program	Brood year	Run Timing	Clipped Adipose Fin	Intact Adipose Fin
Hood Canal	LLTK - Lilliwaup	2018	Summer	-	150,000
Total Annual Release Number				**-**	**150,000**

Table 79. Abundance of natural-origin and hatchery-origin HCS chum salmon spawners in escapements 2013-2017 (NMFS 2020c).

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[b]	% Hatchery Origin	Expected Number of Outmigrants[c]
Strait of Juan de Fuca Population				
Jimmycomelately Creek	1,288	0	0.00%	188,313
Salmon Creek	1,836	0	0.00%	268,531
Snow Creek	311	0	0.00%	45,541
Chimacum Creek	902	0	0.00%	131,971

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[b]	% Hatchery Origin	Expected Number of Outmigrants[c]
Population Average[d]	4,337	0	0.00%	634,355
Hood Canal Population				
Big Quilcene River	6,437	0	0.00%	941,450
Little Quilcene River	122	0	0.00%	17,795
Big Beef Creek	10	0	0.00%	1,532
Dosewallips River	2,021	0	0.00%	295,524
Duckabush River	3,172	0	0.00%	463,856
Hamma Hamma River	2,944	10	0.34%	432,056
Anderson Creek	3	0	0.00%	376
Dewatto River	95	0	0.00%	13,947
Lilliwaup Creek	857	1,141	57.10%	292,159
Tahuya River	205	299	59.36%	73,777
Union River	2,789	2	0.07%	408,166
Skokomish River	2,154	0	0.00%	314,960
Population Average[d]	20,809	1,452	6.52%	3,255,599
ESU Average	**25,146**	**1,452**	**5.46%**	**3,889,955**

[a] Five-year geometric mean of post fishery natural-origin spawners (2015-2019).

[b] Five-year geometric mean of post fishery hatchery-origin spawners (2015-2019).

[c] Expected number of outmigrants = total spawners*45% proportion of females*2,500 eggs per female*13% survival rate from egg to outmigrant.

[d] Averages are calculated as the geometric mean of the annual totals (2015-2019).

The overall trend in spawning abundance is generally stable for the Hood Canal population (all natural spawners and natural-origin only spawners) and for the Strait of Juan de Fuca population (all natural spawners). Productivity rates, which were quite low during the five-year period from 2005-2009 (Ford 2011b), increased from 2011-2015 and were greater than replacement rates from 2014-2015 for both major population groups (NWFSC 2015b).

There were likely at least two ecological diversity groups within the Strait of Juan de Fuca population and at least four ecological diversity groups within the Hood Canal population. With the possible exception of the Dungeness River aggregation within the Strait of Juan de Fuca population, Hood Canal ESU summer chum salmon spawning groups exist today that represent each of the ecological diversity groups within the two populations (NMFS 2017a). Diversity values (Shannon diversity index) were generally lower in the 1990s for both independent populations within the ESU, indicating that most of the abundance occurred at a few spawning sites (NWFSC 2015b). Although the overall linear trend in diversity appears to be negative, the last five-year interval shows the highest average value for both populations within the Hood Canal ESU.

The Hood Canal summer-run chum salmon ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic

Peninsula rivers between Hood Canal and Dungeness Bay, Washington. The nine populations are well distributed throughout the ESU range except for the eastern side of Hood Canal (Johnson et al. 1997). Two independent major population groups have been identified for this ESU: (1) spawning aggregations from rivers and creeks draining into the Strait of Juan de Fuca, and (2) spawning aggregations within Hood Canal proper (Sands 2009).

Status

The two most recent status reviews (2011 and 2015) indicate some positive signs for the Hood Canal summer-run chum salmon ESU. Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; considered a good sign for viability in terms of spatial structure and diversity (Ford 2011b). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NWFSC 2015b). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (Ford 2011b). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years (NWFSC 2015b). Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time (NWFSC 2015b). Overall, the Hood Canal Summer-run chum salmon ESU remains at a moderate risk of extinction.

Critical Habitat

NMFS designated critical habitat for Hood Canal summer-run chum salmon in 2005 (70 FR 52630) and includes 79 miles of stream channels and 377 miles of nearshore marine habitat (Figure 56). Identified PBFs essential for species conservation overlap with the action area in estuarine and nearshore marine areas. For details on the PBFs of Hood Canal Summer-run chum salmon critical habitat refer to Section 6.2.6 since these are the same PBFs as those identified for Puget Sound Chinook salmon.

NMFS excluded some particular DOD sites from the Hood Canal Summer-run chum salmon critical habitat designation because the benefits of exclusion outweigh the benefits of inclusion, and exclusion of those areas will not result in the extinction of the species. For details of the excluded DOD sites refer to Section 6.2.6 since the exclusions that apply to Puget Sound Chinook salmon critical habitat are the same as those that apply to Hood Canal Summer-run chum salmon critical habitat.

The spawning PBF is degraded by excessive fine sediment in the gravel, and the rearing PBF is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Low river flows in several rivers also adversely affect most PBFs. In the estuarine areas, both migration and rearing PBFs of juveniles are impaired by loss of functional floodplain areas necessary for growth and development of juvenile chum salmon.

Recovery Goals

The recovery strategy for Hood Canal Summer-run chum salmon focuses on habitat protection and restoration throughout the geographic range of the ESU, including both freshwater habitat and nearshore marine areas within a one-mile radius of the watersheds' estuaries (NMFS 2007a). The recovery plan includes an ongoing harvest management program to reduce exploitation rates, a hatchery supplementation program, and the reintroduction of naturally spawning summer chum salmon aggregations to several streams where they were historically present. The Hood Canal plan gives first priority to protecting the functioning habitat and major production areas of the ESU's eight extant stocks, keeping in mind the biological and habitat needs of different life-history stages, and second priority to restoration of degraded areas, where recovery of natural processes appears to be feasible (HCCC 2005). For details on Hood Canal Summer-run chum salmon ESU recovery goals, including complete down-listing/delisting criteria, see the Hood Canal Coordinating Council 2005 recovery plan (HCCC 2005) and the NMFS 2007 supplement to this recovery plan (NMFS 2007a).

6.2.20 Coho Salmon – Central California Coast ESU

This evolutionarily significant unit, or ESU, includes naturally spawned coho salmon originating from rivers south of Punta Gorda, California up to and including Aptos Creek, as well as such coho salmon originating from tributaries to San Francisco Bay. Also, coho salmon from three artificial propagation programs are included in this ESU (Figure 57).

Coho salmon are an anadromous species (i.e., adults migrate from marine to freshwater streams and rivers to spawn). Adult coho salmon are typically about two feet long and eight pounds. Coho salmon have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Central California Coast coho salmon, an ESU was listed as threatened under the ESA on October 31, 1996 (64 FR 56138). NMFS re-classified the ESU as endangered on June 28, 2005 (70 FR 37160).

Life History

Coho salmon is an anadromous fish species that generally exhibits a relatively simple three-year life cycle. Adults typically begin their freshwater spawning migration in the late summer and fall, spawn by mid-winter, and then die. The run and spawning times vary between and within populations. Depending on river temperatures, eggs incubate in redds for 1.5 to 4 months before hatching as ''alevins'' (a larval life stage dependent on food stored in a yolk sac). Once most of the yolk sac is absorbed, the 30 to 35 millimeter fish (then termed "fry") begin emerging from the gravel in search of shallow stream margins for foraging and safety (Council 2004). Coho salmon fry typically transition to the juvenile stage by about mid-June when they are about 50 to 60 mm, and both stages are collectively referred to as "young of the year." Juveniles develop vertical dark bands or "parr marks", and begin partitioning available instream habitat through aggressive agonistic interactions with other juvenile fish (Quinn 2005). Juveniles rear in fresh

water for up to 15 months, then migrate to the ocean as ‘‘smolts’’ in the spring. Coho salmon typically spend 2 growing seasons in the ocean before returning to their natal stream to spawn as 3 year-olds. Some precocious males, called ‘‘jacks,’’ return to spawn after only 6 months at sea (NMFS 2014b).

Central California Coast coho salmon typically enter freshwater from November through January, and spawn into February or early March (Moyle 2002). The upstream migration towards spawning areas coincides with large increases in stream flow (Hassler 1987). Coho salmon often are not able to enter freshwater until heavy rains have caused breaching of sand bars that form at the mouths of many coastal California streams. Spawning occurs in streams with direct flow to the ocean, or in large river tributaries (Moyle 2002). Female coho salmon choose a site to spawn at the head of a riffle, just downstream of a pool where water flow changes from slow to turbulent, and where medium to small size gravel is abundant (Moyle 2002).

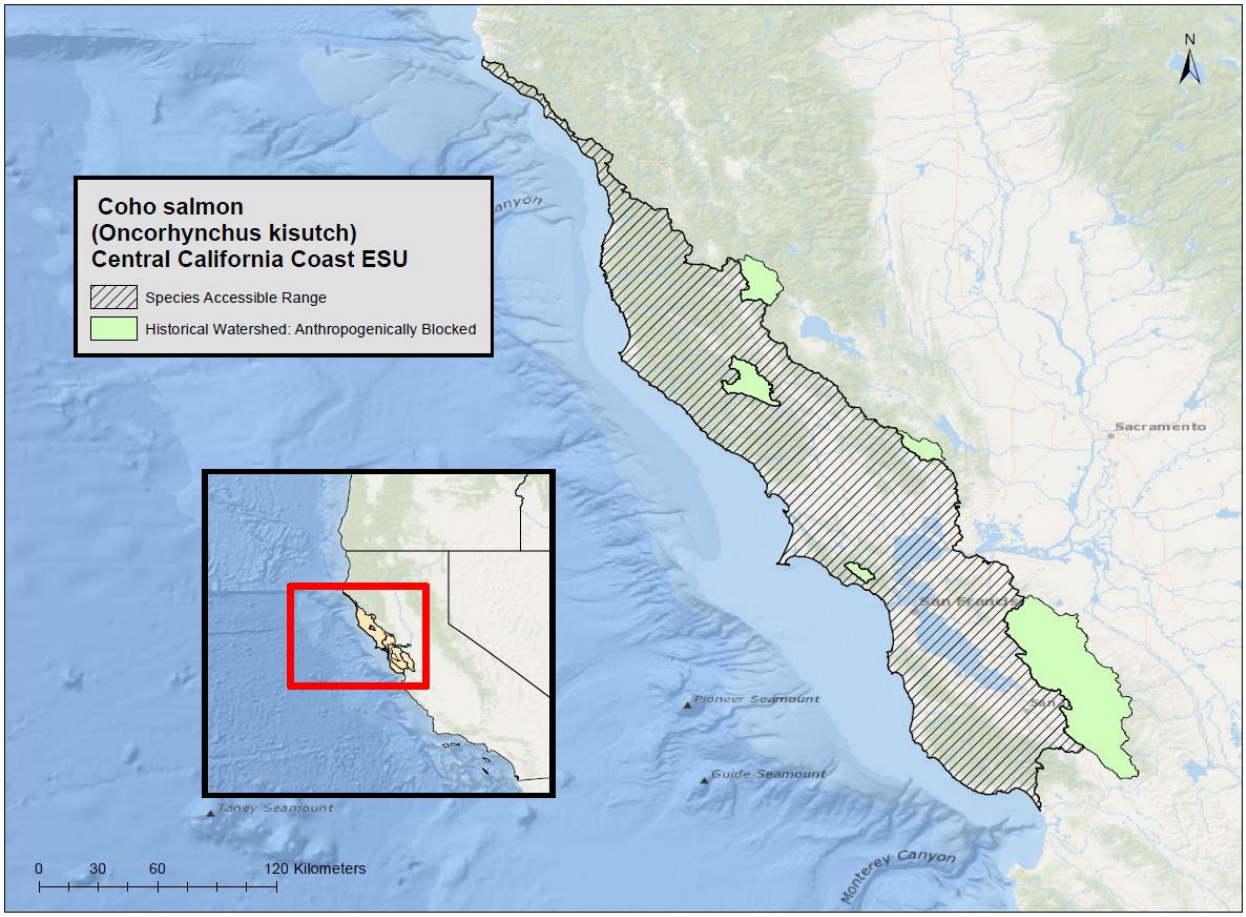


Figure 57. Geographic range of Coho salmon, Central California Coast ESU.

Eggs incubate in redds from November through April, and hatch into “alevins” after a period of 35-50 days (Shapovalov and Taft 1954). The period of incubation is inversely related to water temperature. Alevins remain in the gravel for two to ten weeks then emerge into the water

column as young juveniles, known as "fry". Juveniles, or fry, form schools in shallow water along the undercut banks of the stream to avoid predation. The juveniles feed heavily during this time, and as they grow they set up individual territories. Juveniles are voracious feeders, ingesting any organism that moves or drifts over their holding area. The juvenile's diet is mainly aquatic insect larvae and terrestrial insects, but small fish are taken when available (Moyle 2002).

After one year in freshwater juvenile coho salmon undergo physiological transformation into "smolts" for outmigration to the ocean. Smolts may spend time residing in the estuarine habitat prior to ocean entry, to allow for the transition to the saline environment. After entering the ocean, the immature salmon initially remain in the nearshore waters close to their natal stream. They gradually move northward, generally staying over the continental shelf (Brown et al. 1994). After approximately two years at sea, adult coho salmon move slowly homeward. Adults begin their freshwater migration upstream after heavy fall or winter rains breach the sandbars at the mouths of coastal streams (Sandercock 1991) and/or flows are sufficient to reach upstream spawning areas.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Central California Coast ESU of coho salmon.

Limited information exists on the abundance of coho salmon within the Central California Coast ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (Good et al. 2005). This escapement declined to about 99,000 by the 1960s with approximately 56,000 (56 percent) originating from streams within the Central California Coast ESU. The estimated number of coho salmon produced within the ESU in 2011 was between 2,000 and 3,000 wild adults (Gallagher et al. 2010). Current abundance estimates of the Central California Coast ESU of coho salmon are presented in Table 80 and Table 81 below.

Table 80. Average juvenile Central California Coast coho salmon coho salmon hatchery releases (NMFS 2019d).

Artificial propagation program	Watershed	Years	Clipped Adipose Fin
Don Clausen Fish Hatchery Captive Broodstock Program	Russian River tributaries	2014-2018	132,680
Scott Creek/King Fisher Flats Conservation Program	Gazos and San Vicente creeks	2018	12,000
Scott Creek Captive Broodstock Program	Scott Creek	2013-2017	21,200
Average Annual Release Number			**165,880**

Table 81. Geometric mean abundances of Central California Coast coho salmon spawner escapements by population. Populations in bold font are independent populations (NMFS 2019d).

Stratum	Population	Spawners		Expected Number of Outmigrants[b]
		Natural-origin	Hatchery-origin[a]	
Lost Coast – Navarro Point	**Ten Mile River**	69	-	4,830
	Usal Creek	4	-	280
	Noyo River	455	-	31,850
	Pudding Creek	184	-	12,880
	Caspar Creek	40	-	2,800
	Big River	183	-	12,810
	Little River	30		2,100
	Albion River	21	-	1,470
	Big Salmon Creek	3		210
Navarro Point – Gualala Point	**Navarro River**	102	-	7,140
	Greenwood Creek	3		210
	Garcia River	18	-	1,260
	Gualala River	-	-	-
Coastal	**Russian River**	364[c]	323	48,090
	Salmon Creek	-	-	-
	Walker Creek		-	-
	Lagunitas Creek	408	-	28,560
	Pine Gulch	2		140
	Redwood Creek	23	-	1,610
Santa Cruz Mountains	**Pescadero Creek**	1	-	70
	San Lorenzo River	1	-	70
	Waddell Creek	1	-	70
	Scott Creek	18	4	1,540
	San Vicente Creek	2	-	140
	Soquel Creek	-	-	-
ESU Total		**1,932**	**327**	**158,130**

[a] J. Jahn, pers. comm., July 2, 2013
[b] Expected number of outmigrants=Total spawners*50% proportion of females*2,000 eggs per female*7% survival rate from egg to outmigrant
[c] Arithmetic mean used due to unavailability of geometric mean

Within the Lost Coast – Navarro Point stratum and the Navarro Point – Gualala Point stratum, most independent populations show positive but non-significant population trends. Dependent populations within these stratum have declined significantly since 2011. In the Russian River and Lagunitas Creek watersheds, which are the two largest within the Central Coast strata, recent coho salmon population trends suggest limited improvement, although both populations remain well below recovery targets. Recent sampling within Pescadero Creek and San Lorenzo River,

the only two independent populations within the Santa Cruz Mountains strata, suggest coho salmon have likely been extirpated within both basins.

Genetic studies show little homogenization of populations, *i.e.*, transfer of stocks between basins have had little effect on the geographic genetic structure of central California Coast coho salmon (Sonoma County Water Agency (SCWA) 2002). This ESU likely has considerable diversity in local adaptations given that the ESU spans a large latitudinal diversity in geology and ecoregions, and include both coastal and inland river basins.

The TRT identified 11 "functionally independent", one "potentially independent" and 64 "dependent" populations in the Central California Coast ESU of coho salmon (Spence et al. 2008). The 75 populations were grouped into five Diversity Strata. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of the Central California Coast coho salmon ESU (Spence et al. 2008). The Russian River population, once the largest and most dominant source population in the ESU, is now at high risk of extinction because of low abundance and failed productivity (Spence et al. 2008). The Lost Coast and Navarro Point contain the majority of coho salmon remaining in the ESU.

Status

The low survival of juveniles in freshwater, in combination with poor ocean conditions, has led to the precipitous declines of Central California Coast ESU coho salmon populations. Most independent populations remain at critically low levels, with those in the southern Santa Cruz Mountains strata likely extirpated. Data suggests some populations show a slight positive trend in annual escapement, but the improvement is not statistically significant. Overall, all populations remain, at best, a slight fraction of their recovery target levels, and, aside from the Santa Cruz Mountains strata, the continued extirpation of dependent populations continues to threaten the ESU's future survival and recovery.

Critical Habitat

Critical habitat for the Central California Coast ESU of coho salmon was designated on May 5, 1999 (64 FR 24049). Critical habitat includes juvenile summer and winter rearing areas, juvenile migration corridors, areas for growth and development to adulthood, adult migration corridors, and spawning areas. The physical or biological features that characterize these sites include substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions. Designated critical habitat for the Central California Coast ESU of coho salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

NMFS (2008a) evaluated the condition of each habitat attribute in terms of its current condition relative to its role and function in the conservation of the species. The assessment of habitat for this species showed a distinct trend of increasing degradation in quality and quantity of all PBFs as the habitat progresses south through the species range, with the area from the Lost Coast to the

Navarro Point supporting most of the more favorable habitats and the Santa Cruz Mountains supporting the least. However, all populations are generally degraded regarding spawning and incubation substrate, and juvenile rearing habitat. Elevated water temperatures occur in many streams across the entire ESU.

Recovery Goals

See the 2012 Recovery Plan for complete down listing/delisting criteria for each of the following recovery goals (NMFS 2012c):

1. Prevent extinction by protecting existing populations and their habitats;
2. Maintain current distribution of coho salmon and restore their distribution to previously occupied areas essential to their recovery;
3. Increase abundance of coho salmon to viable population levels, including the expression of all life history forms and strategies;
4. Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta populations;
5. Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally;
6. Ensure all factors that led to the listing of the species have been ameliorated; and
7. Develop and maintain a program of monitoring, research, and evaluation that advances understanding of the complex array of factors associated with coho salmon survival and recovery and which allows for adaptively managing our approach to recovery over time.

6.2.21 Coho Salmon – Lower Columbia River ESU

This ESU includes naturally spawned coho salmon originating from the Columbia River and its tributaries downstream from the Big White Salmon and Hood Rivers (inclusive) and any such fish originating from the Willamette River and its tributaries below Willamette Falls. Also, coho salmon originate from 21 artificial propagation programs (Figure 58). A physical description of coho salmon is presented in Section 6.2.9. The Lower Columbia River ESU of coho salmon was listed as threatened under the ESA on June 28, 2005(*Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs*).

Life History

Lower Columbia River coho salmon are typically categorized into early- and late-returning stocks. Early-returning (Type S) adult coho salmon enter the Columbia River in mid-August and begin entering tributaries in early September, with peak spawning from mid-October to early November. Late-returning (Type N) coho salmon pass through the lower Columbia from late September through December and enter tributaries from October through January. Most spawning occurs from November to January, but some occurs as late as March (LCFRB 2010).

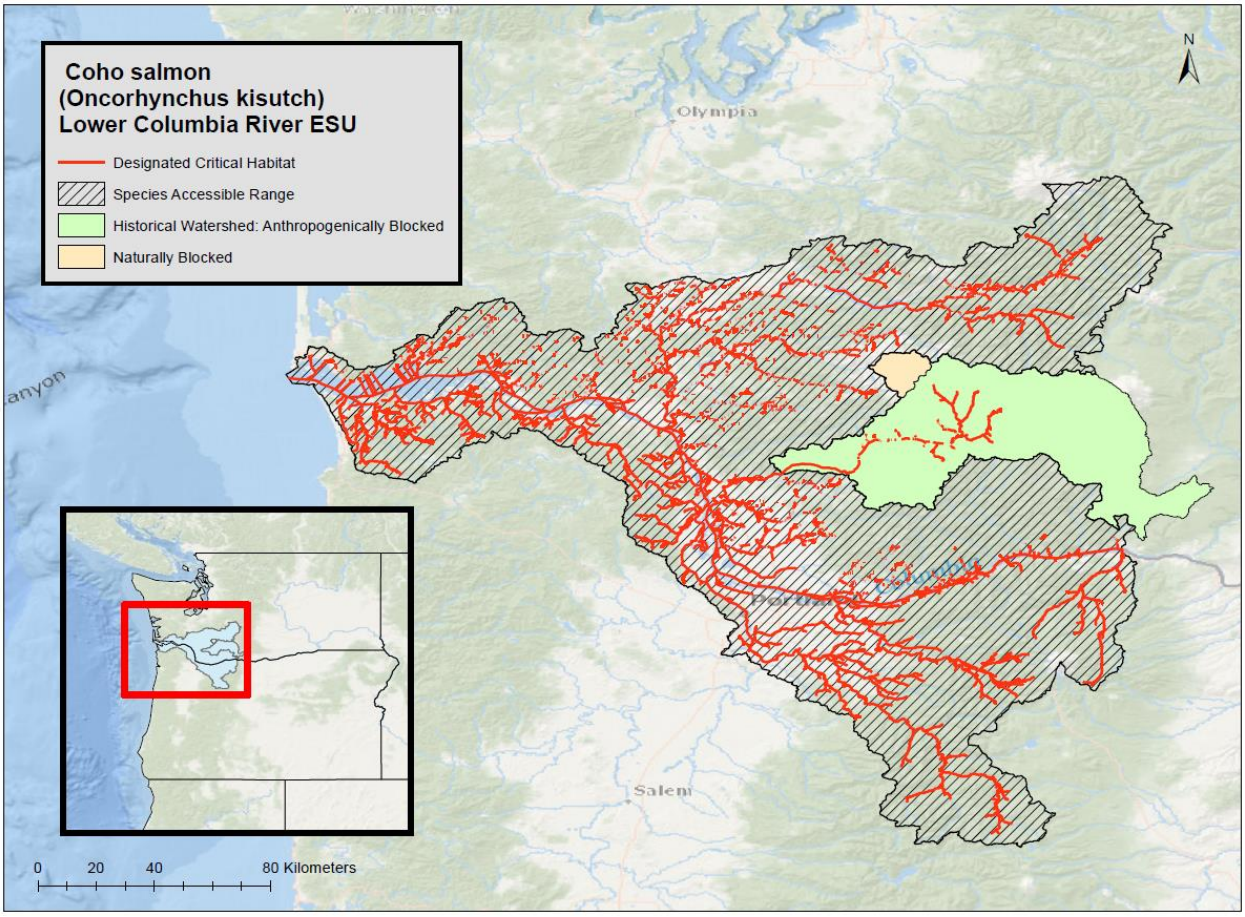


Figure 58. Geographic range and designated critical habitat of Coho salmon, Lower Columbia River ESU.

Coho salmon typically spawn in small to medium, low- to-moderate elevation streams from valley bottoms to stream headwaters. Coho salmon construct redds in gravel and small cobble substrate in pool tailouts, riffles, and glides, with sufficient flow depth for spawning activity (NMFS 2013c). Eggs incubate over late fall and winter for about 45 to 140 days, depending on water temperature, with longer incubation in colder water. Fry may thus emerge from early spring to early summer (ODFW 2010). Juveniles typically rear in freshwater for more than a year. After emergence, coho salmon fry move to shallow, low-velocity rearing areas, primarily along the stream edges and inside channels. Juvenile coho salmon favor pool habitat and often congregate in quiet backwaters, side channels, and small creeks with riparian cover and woody debris. Side-channel rearing areas are particularly critical for overwinter survival, which is a key regulator of freshwater productivity (LCFRB 2010).

Most juvenile coho salmon migrate seaward as smolts in April to June, typically during their second year. Salmon that have stream-type life histories, such as coho salmon, typically do not linger for extended periods in the Columbia River estuary, but the estuary is a critical habitat used for feeding during the physiological adjustment to salt water. Juvenile coho salmon are

present in the Columbia River estuary from March to August. Columbia River coho salmon typically range throughout the nearshore ocean over the continental shelf off of the Oregon and Washington coasts. Early-returning (Type S) coho salmon are typically found in ocean waters south of the Columbia River mouth. Late-returning (Type N) coho salmon are typically found in ocean waters north of the Columbia River mouth. Most coho salmon sexually mature at age three, except for a small percentage of males (called "jacks") who return to natal waters at age two, after only 5 to 7 months in the ocean (LCFRB 2010).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to Lower Columbia River ESU of coho salmon.

Washington tributaries indicate the presence of moderate numbers of coho salmon, with total abundances in the hundreds to low thousands of fish. Oregon tributaries have abundances in the hundreds of fish. In the Western Cascade MPG, the Sandy and Clackamas Rivers were the only two populations identified in the original 1996 Status Review that appeared to be self-sustaining natural populations. Natural origin abundances in the Columbia Gorge MPG are low, with hatchery-origin fish contributing a large proportion of the total number of spawners, most notably in the Hood River (NMFS 2016d). Current abundance estimates of the Lower Columbia River ESU of coho salmon are presented in Table 82 and Table 83 below.

Table 82. Juvenile Abundance Estimates for the Lower Columbia River ESU of Coho salmon (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Juvenile	651,378
Listed Hatchery Intact Adipose	Juvenile	287,056
Listed Hatchery Adipose Clipped	Juvenile	7,055,635

Table 83. Average abundance estimates for Lower Columbia River Chinook salmon natural- and hatchery-origin spawners (NMFS 2020c).

Population Name	Years	Natural-origin Spawners	Hatchery-origin Spawners	% Hatchery Origin
Coastal Stratum – Fall run				
Youngs Bay	2012-2014	233	5,606	96.01%
Grays/Chinook salmon	2010-2014	100	357	78.12%
Big Creek	2012-2014	32	1,510	97.92%
Elochoman/Skamokowa	2010-2014	116	580	83.33%
Clatskanie	2012-2014	98	3,193	97.02%

Population Name	Years	Natural-origin Spawners	Hatchery-origin Spawners	% Hatchery Origin
Mill/Abernathy/Germany	2010-2014	92	805	89.74%
Cascade Stratum – Fall run				
Lower Cowlitz	2010-2013	723	196	21.33%
Upper Cowlitz	2010-2013	2,873	961	25.07%
Toutle	2010-2014	3,305	5,400	62.03%
Coweeman	2010-2014	385	963	71.44%
Kalama	2010-2014	803	8,892	91.72%
Lewis	2010-2014	2,178	943	30.21%
Washougal	2010-2014	192	116	37.66%
Clackamas	2012-2014	1,272	2,955	69.91%
Sandy	2012-2014	1,207	320	20.96%
Columbia Gorge Stratum – Fall run				
Lower Gorge	2003-2007	146	-	-
Upper Gorge	2010-2012	200	327	62.05%
White Salmon	2010-2014	829	246	22.88%
Cascade Stratum – Late fall run				
North Fork Lewis	2010-2014	12,330	0	0.00%
Cascade Stratum – Spring run				
Upper Cowlitz/Cispus	2010-2014	279	3,614	92.83%
Kalama	2011-2014	115	-	-
North Fork Lewis	2010-2014	217	0	0.00%
Sandy	2010-2014	1,731	1,470	45.92%
Gorge Stratum – Spring run				
White Salmon	2013-2014	13	140	91.50%
ESU Average		**29,469**	**38,594**	**56.70%**

The spatial structure of some populations is constrained by migration barriers (such as tributary dams) and development in lowland areas. Low abundance, past stock transfers, other legacy hatchery effects, and ongoing hatchery straying may have reduced genetic diversity within and among coho salmon populations. It is likely that hatchery effects have also decreased population productivity (NMFS 2016d).

This ESU includes all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia River up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon, as well as multiple artificial propagation programs. Most of the populations in the ESU contain a substantial number of hatchery-origin spawners. Myers et al. (2006) identified three MPGs (Coastal, Cascade, and Gorge), containing a total of 24 DIPs in the Lower Columbia River coho salmon ESU (NMFS 2016d).

Status

Recovery efforts have likely improved the status of a number of coho salmon DIPs, abundances are still at low levels and the majority of the DIPs remain at moderate or high risk. For the lower Columbia River region, land development and increasing human population pressures will likely continue to degrade habitat, especially in lowland areas. Although populations in this ESU have generally improved, especially in the 2013/14 and 2014/15 return years, recent poor ocean conditions suggest that population declines might occur in the upcoming return years. Regardless, this ESU is still considered to be at moderate risk (NMFS 2016d).

Critical Habitat

Critical habitat for the lower Columbia River coho salmon ESU was designated on February 24, 2016 (81 FR 9252). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Columbia River ESU of coho salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

Reduced complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migrating continues to be a concern for all four lower Columbia River listed species. Loss of habitat from conversion to agricultural or urbanized uses continues to be a particular concern throughout the lower Columbia River region, especially the loss of habitat complexity in the lower tributary/mainstem Columbia River interface, and concomitant changes in water temperature (LCFRB 2010; NMFS 2013c; ODFW 2010).

Recovery Goals

This species is included in the Lower Columbia River recovery plan (NMFS 2013a). Specific recovery goals are to improve all four viability parameters to the point that the Coast, Cascade, and Gorge strata achieve high probability of persistence. Protection of existing high functioning habitat and restoration of tributary habitat are noted needs, along with the reduction of hatchery and harvest impacts. Large improvements are needed in the persistence probability of most populations of this ESU.

6.2.22 Coho Salmon – Oregon Coast ESU

This ESU includes naturally spawned coho salmon originating from coastal rivers south of the Columbia River and north of Cape Blanco, and also coho salmon from one artificial propagation program: Cow Creek Hatchery Program (Figure 59). A physical description of coho salmon is presented in Section 6.2.9. The Oregon Coast ESU of coho salmon was listed as threatened under

the ESA on August 10, 1998 (63 FR 42587). The listing was revisited and confirmed as threatened on June 20, 2011 (76 FR 35755).

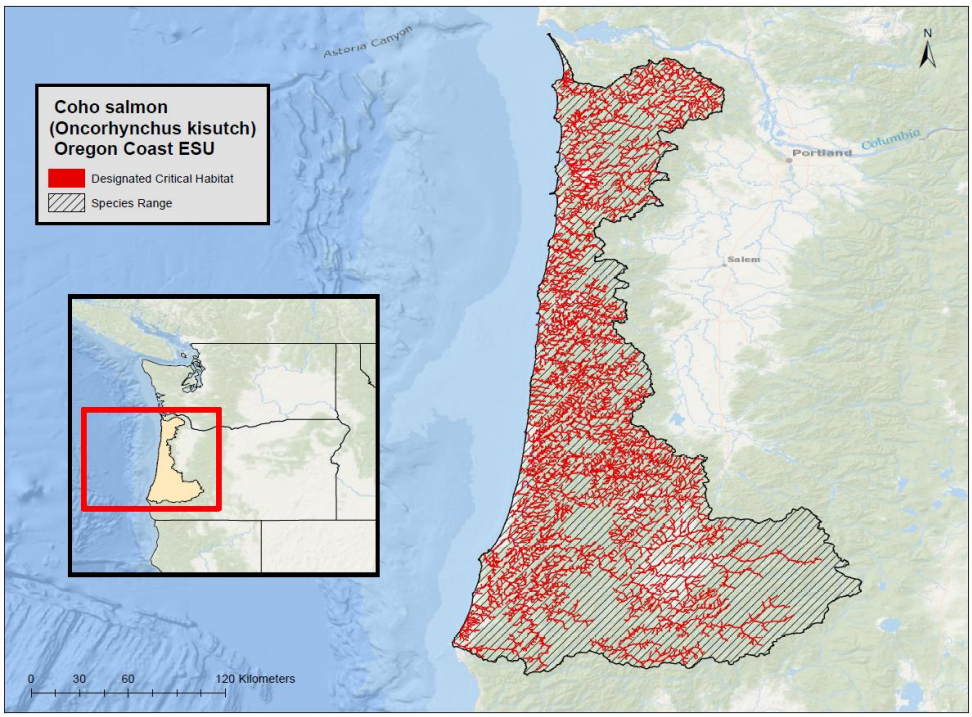


Figure 59. Geographic range and designated critical habitat of coho salmon, Oregon coast ESU.

Life History

The anadromous life cycle of coho salmon begins in their home stream where they emerge from eggs as 'alevins' (a larval life stage dependent on food stored in a yolk sac). These very small fish require cool, slow moving freshwater streams with quiet areas such as backwater pools, beaver ponds, and side channels (Reeves et al. 1989) to survive and grow through summer and winter seasons. Current production of coho salmon smolts in the Oregon Coast coho salmon ESU is particularly limited by the availability of complex stream habitat that provides the shelter for overwintering juveniles during periods when flows are high, water temperatures are low, and food availability is limited (ODFW 2007).

The Oregon Coast coho salmon follow a yearling-type life history strategy, with most juvenile coho salmon migrating to the ocean as smolts in the spring, typically from as late as March into June. Coho salmon smolts outmigrating from freshwater reaches may feed and grow in lower mainstem and estuarine habitats for a period of days or weeks before entering the nearshore ocean environment. The areas can serve as acclimation areas, allowing coho salmon juveniles to adapt to saltwater. Research shows that substantial numbers of coho salmon fry may also emigrate downstream from natal streams into tidally influenced lower river wetlands and estuarine habitat (Bass 2010; Chapman 1962; Koski 2009).

Oregon Coast coho salmon tend to make relatively short ocean migrations. Coho salmon from this ESU are present in the ocean from northern California to southern British Columbia, and even fish from a given population can be widely dispersed in the coastal ocean, but the bulk of the ocean harvest of coho salmon from this ESU are found off the Oregon coast. The majority of coho salmon adults return to spawn as 3–year-old fish, having spent about 18 months in freshwater and 18 months in salt water (Sandercock 1991). The primary exceptions to this pattern are ''jacks,'' sexually mature males that return to freshwater to spawn after only five to seven months in the ocean.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Oregon Coast ESU of coho salmon.

Results from the most recent NMFS Northwest Fisheries Science Center review show that while Oregon Coast coho salmon spawner abundance varies by time and population, the total abundance of spawners within the ESU has been generally increasing since 1999, with total abundance exceeding 280,000 spawners in three years between 2010 to 2015 (NWFSC 2015b).

Most independent populations in the ESU showed an overall increasing trend in abundance with synchronously high abundances in 2002-2003, 2009-2011, and 2014, and low abundances in 2007, 2009, and 2015. This synchrony suggests the overriding importance of marine survival to recruitment and escapement of Oregon Coast coho salmon (NWFSC 2015b). When future conditions are taken into account, the Oregon Coast coho salmon ESU, as a whole, is at moderate risk of extinction, but the recent risk trend is stable and improving (NWFSC 2015b). Current abundance estimates for natural and hatchery spawners as well as the expected number of outmigrants for the Oregon Coast ESU of coho salmon are presented in Table 84 below. The hatchery production goal is 60,000 adipose-fin-clipped yearling Oregon Coast ESU coho salmon (NMFS 2020c).

Table 84. Average abundance estimates for the Oregon Coast ESU coho salmon natural- and hatchery-origin spawners (NMFS 2020c).

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[a]	% Hatchery Origin	Expected Number of Outmigrants[b]
North Coast Stratum				
Necanicum River	1,139	5	0.42%	80,063
Nehalem River	7,073	11	0.16%	495,889
Tillamook Bay	4,771	19	0.39%	335,290
Nestucca River	2,320	2	0.09%	162,547
North Coast Dependents	602	3	0.49%	42,350
Mid-Coast Stratum				
Salmon River	924	9	0.98%	65,352
Siletz River	5,534	2	0.04%	387,545
Yaquina River	4,585	2	0.05%	321,141
Beaver Creek	1,634	1	0.09%	114,493
Alsea River	8,627	0	0.00%	603,904
Siuslaw River	12,994	0	0.00%	909,584
Mid Coast Dependents	1,190	7	0.56%	83,747
Lakes Stratum				
Siltcoos Lake	2,362	0	0.00%	165,333
Tahkenitch Lake	1,356	2	0.13%	95,077
Tenmile Lake	2,909	0	0.00%	203,660
Umpqua Stratum				
Lower Umpqua River	8,755	2	0.02%	612,987
Middle Umpqua River	3,080	0	0.00%	215,578
North Umpqua River	2,320	191	7.59%	175,760
South Umpqua River	3,683	299	7.52%	278,743
Mid-South Coast Stratum				
Coos River	6,320	0	0.00%	442,407
Coquille River	10,781	3	0.03%	754,870
Floras Creek	1,154	0	0.00%	80,785
Sixes River	200	0	0.00%	14,029
Mid-South Coast Dependents	5	1	16.36%	428
ESU Average	**94,320**	**559**	**0.59%**	**6,641,564**

[a] Five-year geometric mean of post-fishery spawners (2013-2017).

[b] Expected number of outmigrants=Total spawners*50% proportion of females*2,000 eggs per female*7% survival rate from egg to outmigrant.

While the 2008 biological review team status review concluded that there was low certainty that ESU-level genetic diversity was sufficient for long-term sustainability in the ESU (Wainwright et al. 2008), a 2015 NMFS Northwest Fisheries Science Center review suggests this is an

unlikely outcome. The observed upward trends in abundance and productivity and downward trends in hatchery influence make decreases in genetic or life history diversity or loss of dependent populations in recent years unlikely (NWFSC 2015b).

The geographic setting for the Oregon Coast coho salmon ESU includes the Pacific Ocean and the freshwater habitat (rivers, streams, and lakes) along the Oregon Coast from the Necanicum River near Seaside on the north to the Sixes River near Port Orford on the south. The Oregon/Northern California Coasts Technical Recovery Team identified 56 historical populations that function collectively to form the Oregon Coast coho salmon ESU. The team classified 21 of the populations as independent because they occur in basins with sufficient historical habitat to have persisted through several hundred years of normal variations in marine and freshwater conditions (NMFS 2016g).

Status

Findings by the NWFSC (2015b) and ODFW (2016) show many positive improvements to Oregon Coast coho salmon in recent years, including positive long-term abundance trends and escapement. Results from the NMFS Northwest Fisheries Science Center's recent review show that while Oregon Coast coho salmon spawner abundance varies by time and population, the total abundance of spawners within the ESU has generally increased since 1999, with total abundance exceeding 280,000 spawners in recent years. Overall, the NWFSC (2015b) found that increases in Oregon Coast coho salmon ESU scores for persistence and sustainability clearly indicate that the biological status of the ESU is improving, due in large part to management decisions (reduced harvest and hatchery releases). It determined, however, that Oregon Coast coho salmon abundance remains strongly correlated with marine survival rates.

Critical Habitat

NMFS published a final rule designating critical habitat for Oregon Coast coho salmon on February 11, 2008 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Oregon Coast coho salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

There are 80 watersheds within the range of the Oregon Coast coho salmon ESU. Eight watersheds received a low conservation value rating, 27 received a medium rating, and 45 received a high rating.

Recovery Goals

See the 2016 Recovery Plan for detailed descriptions of the recovery goals and delisting criteria (NMFS 2016g). In the simplest terms, NMFS will remove the Oregon Coast coho salmon from federal protection under the ESA when we determine that:

- The species has achieved a biological status consistent with recovery—the best available information indicates it has sufficient abundance, population growth rate, population spatial structure, and diversity to indicate it has met the biological recovery goals.
- Factors that led to ESA listing have been reduced or eliminated to the point where federal protection under the ESA is no longer needed, and there is reasonable certainty that the relevant regulatory mechanisms are adequate to protect Oregon Coast coho salmon sustainability.

6.2.23 Coho Salmon – Southern Oregon and Northern California Coasts ESU

This evolutionarily significant unit, or ESU, includes naturally spawned coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California (Figure 60). Also, coho salmon originate from three artificial propagation programs. A physical description of coho salmon is presented in Section 6.2.20. The Southern Oregon/Northern California Coast (SONCC) ESU of coho salmon was listed as threatened under the ESA on May 6, 1997 (62 FR 24588). The listing was revisited and confirmed as threatened on June 28, 2005(*Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs*).

Life History

A general summary of the life history of coho salmon is presented in Sections 6.2.20, 6.2.21, and 6.2.22.

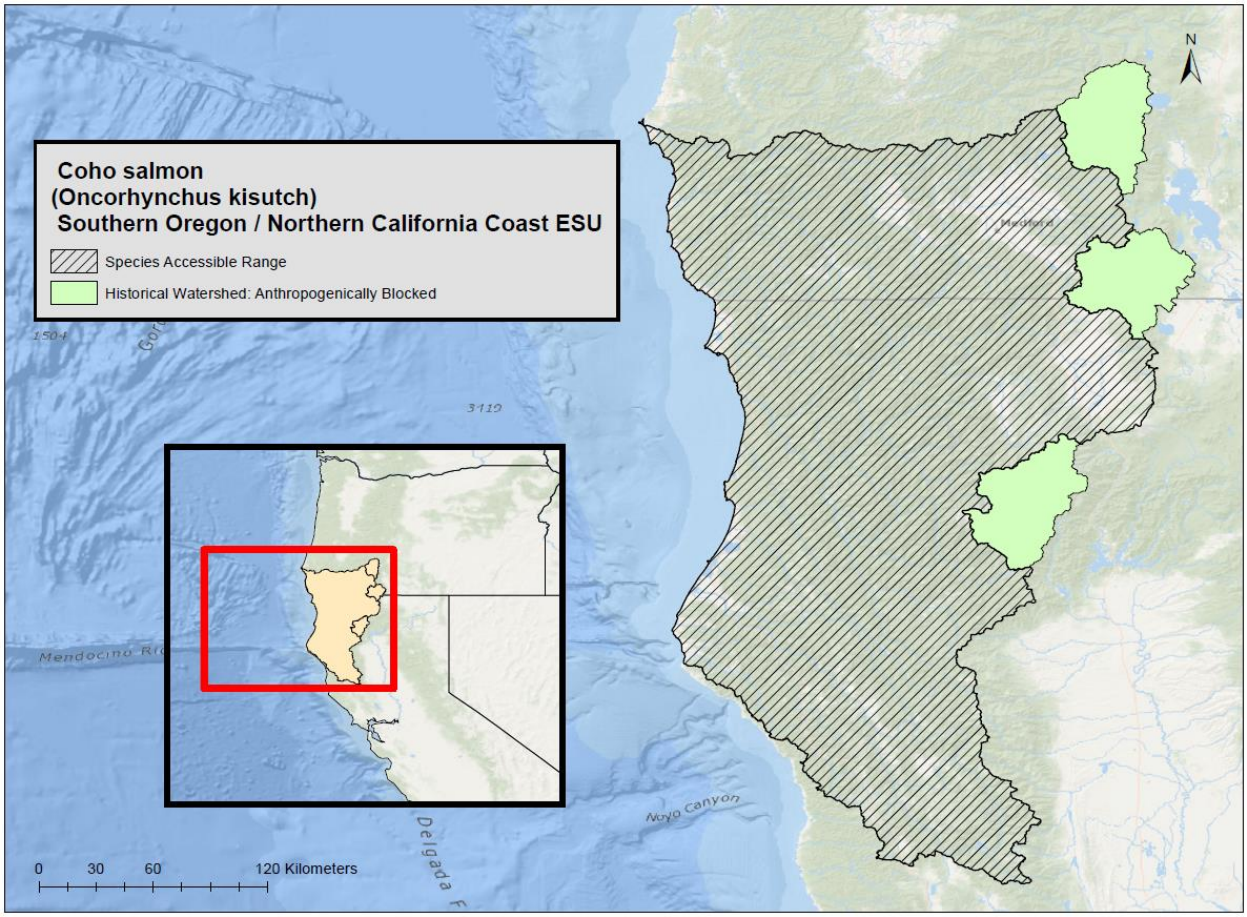


Figure 60. Geographic range of the Southern Oregon/Northern California ESU of Coho salmon.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to SONCC ESU of coho salmon.

Although long-term data on abundance of SONCC coho salmon are scarce, the best available data indicate that none of the seven diversity strata appear to support a single viable population, although all diversity strata are occupied (NMFS 2014b). Further, 24 out of 31 independent populations are at high risk of extinction and 6 are at moderate risk of extinction. Abundance estimates for adult SONCC ESU coho salmon are presented in Table 85 below. Current average abundance estimates for juvenile SONCC ESU coho salmon are 200,000 hatchery produced fish with clipped adipose fins, 575,000 hatchery produced fish with intact adipose fins, and 2,013,593 natural origin fish (NMFS 2020c).

Table 85. Average abundance estimates of the natural-origin and hatchery-produced adult Southern Oregon/Northern California Coast ESU coho salmon returning to the Rogue, Trinity, and Klamath rivers (NMFS 2020c).

YEAR	Rogue River		Trinity River		Klamath River		
	Hatchery	Natural	Hatchery	Natural	Shasta River[a]	Scott River[a]	Salmon River
2008	158	414	3,851	944	30	62	
2009	518	2,566	2,439	542	9	81	
2010	753	3,073	2,863	658	44	927	
2011	1,156	3,917	9,009	1,178	62	355	
2012	1,423	5,440	8,662	1,761		201	
2013	1,999	11,210	11,177	4,097			
2014	829	2,409	8,712	917			
Average [b]	**1,417**	**6,353**	**9,517**	**2,258**	**38**	**357**	**50 [c]**

[a] Hatchery proportion unknown, but assumed to be low.
[b] 3-year average of most recent years of data.
[c] Annual returns of adults are likely less than 50 per year.

The extinction risk of an ESU depends upon the extinction risk of its constituent independent populations; because the population abundance of most independent populations are below their depensation threshold, the SONCC coho salmon ESU is at high risk of extinction and is not viable (Williams et al. 2011c). Estimates from the Rogue River with its four independent populations indicate a small but significant positive trend ($p = 0.01$) over the past 35 years and a non-significant negative trend ($p > 0.05$) over the past 12 years or four generations (NMFS 2016e). The decline in abundance from historical levels and the poor status of population viability criteria are the main factors behind the extinction risk of the ESU.

Williams *et al.* (2006b) designated 45 populations of coho salmon in the SONCC coho salmon ESU as dependent or independent based on their historical population size. Two populations are both small enough and isolated enough that they are only intermittently present (McElhany et al. 2000; NMFS 2014b; Williams et al. 2006b). These populations were further grouped into seven diversity strata based on the geographical arrangement of the populations and basin-scale genetic, environmental, and ecological characteristics.

Status

Though population-level estimates of abundance for most independent populations are lacking, the best available data indicate that none of the seven diversity strata appears to support a single viable population as defined by the SONCC coho salmon technical recovery team's viability criteria (low extinction risk; Williams et al. (2008). Further, 24 out of 31 independent populations are at high risk of extinction and six are at moderate risk of extinction. Based on the above discussion of the population viability parameters, and qualitative viability criteria presented in Williams et al. (2008), NMFS concludes that the SONCC coho salmon ESU is currently not viable and is at high risk of extinction. The primary causes of the decline are likely

long-standing human-caused conditions (e.g., harvest and habitat degradation), which exacerbated the impacts of adverse environmental conditions (e.g., drought and poor ocean conditions) (60 FR 38011; July 25, 1995).

Critical Habitat

NMFS designated critical habitat for the SONCC ESU of coho salmon on May 5, 1999 (64 FR 24049). Critical habitat includes juvenile summer and winter rearing areas, juvenile migration corridors, areas for growth and development to adulthood, adult migration corridors, and spawning areas. The physical or biological features that characterize these sites include substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions. Designated critical habitat for the SONCC ESU of coho salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion and, therefore, will not be analyzed further in this opinion.

Critical habitat designated for the SONCC coho salmon is generally of good quality in northern coastal streams. Spawning PBFs have been degraded throughout the ESU by logging activities that has increased fines in spawning gravel. Rearing PBFs have been considerably degraded in many inland watersheds from the loss of riparian vegetation resulting in unsuitably high water temperatures. Rearing and juvenile migration PBFs have been reduced from the disconnection of floodplains and off-channel habitat in low gradient reaches of streams, consequently reducing winter rearing capacity.

Recovery Goals

A recovery plan is available for this species (NMFS 2014b). In 2016, we completed a 5-year review for this ESU in which we concluded that the ESU should remain listed as threatened; in the last five years there has not been improvement in the status of SONCC coho salmon or a significant change in risk to persistence of the ESU (NMFS 2016f).

6.2.24 Eulachon – Southern DPS

The eulachon is a small, cold-water species of anadromous fish, occupying the eastern Pacific Ocean in nearshore waters to depths of about 1,000 feet (300 meters) from California to the Bering Sea. Eulachon will return to their natal river spawn. Southern DPS eulachon are those that spawn in rivers south of the Nass River in British Columbia to the Mad River in California (Figure 61) (NMFS 2016f).

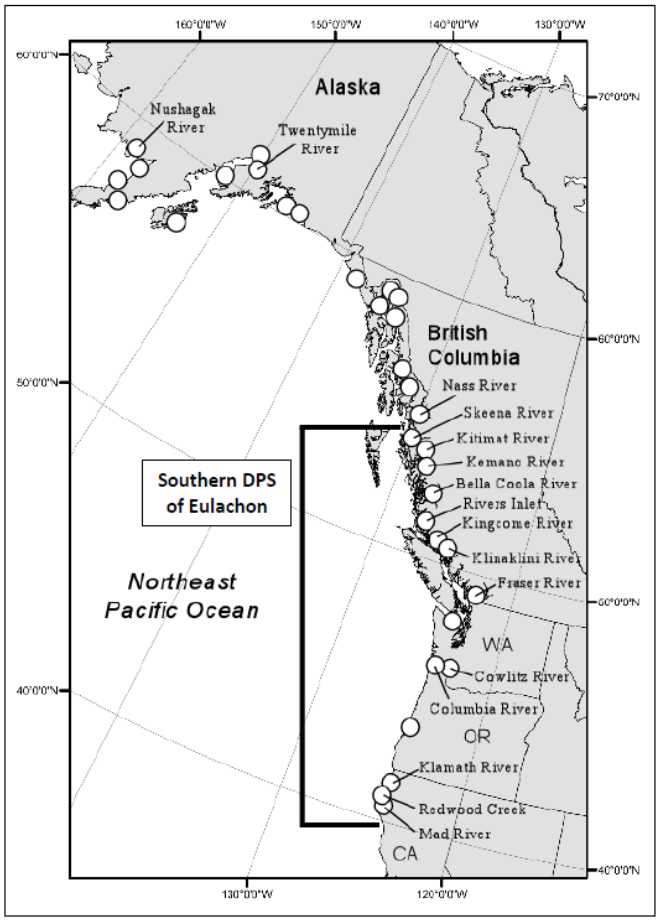


Figure 61. Map identifying the range of the eulachon Southern distinct population segment (NMFS 2016f).

Eulachon are a small (8.5 in (21.5 cm)) anadromous fish, with brown or blue backs, silver on their sides, and white underneath. The Southern DPS was first listed as threatened by NMFS on March 18, 2010 (75 FR 13012).

We used information available in the status review (Gustafson et al. 2010), the updated status review (Gustafson 2016), the 5-year review (NMFS 2016f), and recent scientific publications to summarize the life history, population dynamics and status of the species, as follows.

Life History

Although primarily marine, eulachon return to freshwater to spawn. For the Southern DPS eulachon, most spawning occurs in the Columbia River and its tributaries. Spawning usually occurs between ages two and five. Spawning is strongly influenced by water temperatures, and the timing of migration typically occurs between December and June, when water temperatures are between 0°C and 10°C (Gustafson 2016). In the Columbia River and further south, spawning occurs from late January to March (Hay and McCarter 2000). Further north, the peak of eulachon runs in Washington State is from February through March (Hay and McCarter 2000). Females

lay between 7,000 and 60,000 eggs over sand, course gravel or detrital substrate. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters. In their first year of life, juveniles are found along the continental shelf (Gustafson 2016; Wydoski and Whitney 1979). Adult eulachon are found in coastal and offshore marine habitats. With the exception of some individuals in Alaska, eulachon generally die after spawning (Gustafson 2016). The maximum known lifespan is nine years of age, but 20 to 30 percent of individuals live to four years and most individuals survive to three years of age, although spawning has been noted as early as two years of age. Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). The primary prey of adult eulachon are copepods and euphausiids, malacostracans and cumaceans.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Southern DPS eulachon.

For most Southern DPS eulachon spawning runs, abundance is unknown with the exception of the Columbia and Fraser River spawning runs. Beginning in 1995, the Canada's Department of Fisheries and Oceans (DFO) started annual surveys in the Fraser River. These surveys consisted of estimating larval density, measuring river discharge, and using estimates of relative fecundity to determine spawning biomass (NMFS 2020c). Beginning in 2011, Oregon Department of Fish and Wildlife (ODFW) and Washington Department of Fish and Wildlife (WDFW) began instituting similar monitoring in the Columbia River. From 2014 through 2018, the eulachon spawner population estimate for the Fraser River is 2,608,909 adults and for the Columbia River 16,188,081 adults (Table 86). The combined spawner estimate from the Columbia and Fraser rivers is 18,796,090 eulachon (NMFS 2020c).

Southern DPS eulachon are genetically distinct from eulachon in the northern parts of its range (i.e., Alaska). Recent genetic analysis indicates that the Southern DPS exhibits a regional population structure, with a three-population southern Columbia-Fraser group, coming from the Cowlitz, Columbia, and Fraser rivers (Candy et al. 2015; Gustafson 2016).

Adult and juvenile Southern DPS eulachon can be found in the Pacific Ocean, along the continental shelf, in waters from 50 to 200 meters deep (Gustafson 2016). Adults are most frequently found in the Columbia River and its tributaries (e.g., Cowlitz River, Sandy River), and sometimes in the Klamath River, California.

Status

Eulachon formerly experienced widespread, abundant runs and have been a staple of Native American diets for centuries along the northwest coast. However, such runs that were formerly present in several California rivers as late as the 1960s and 1970s (i.e., Klamath River, Mad

River and Redwood Creek) no longer occur (Larson and Belchik 2000). This decline likely began in the 1970s and continued until, in 1988 and 1989, the last reported sizeable run occurred in the Klamath River and no fish were found in 1996, although a moderate run was noted in 1999 (Moyle 2002). Eulachon have not been identified in the Mad River and Redwood Creek since the mid-1990s (Moyle 2002). The species is considered to be at moderate risk of extinction throughout its range because of a variety of factors, including predation, commercial and recreational fishing pressure (directed and bycatch), and loss of habitat. Climate change impacts on ocean conditions were classified as the most serious threat to the Southern DPS of eulachon by NOAA's Biological Review Team and were identified as a high research priority in the Recovery Plan (Gustafson et al. 2010; NMFS 2017e). Warmer water temperatures associated with climate change could alter the timing of spawning, and the availability of prey for larval and juvenile eulachon (NMFS 2016f). Further population decline is anticipated to continue as a result of climate change and bycatch in commercial fisheries. However, because of their fecundity, eulachon are assumed to have the ability to recover quickly if given the opportunity (Bailey and Houde 1989).

Table 86. Southern DPS eulachon spawning estimates for the lower Fraser River (British Columbia, Canada) and Columbia River (Oregon/Washington states, USA) (NMFS 2020c).

Year	Fraser River		Columbia River	
	Biomass estimate (metric tons)	Estimated spawner population	Biomass estimate (metric tons)	Estimated spawner population
2011	31	765,445	723	17,860,400
2012	120	2,963,013	810	20,008,600
2013	100	2,469,177	1,845	45,546,700
2014	66	1,629,657	3,412	84,243,100
2015	317	7,827,292	2,330	57,525,700
2016	44	1,086,438	877	21,654,800
2017	35	864,212	330	8,148,600
2018	408	10,074,243	53	1,300,000
2014-2018	**106**	**2,608,009**	**656**	**16,188,081**

Critical Habitat

On October 20, 2011, NMFS designated critical habitat for Southern DPS eulachon (76 FR 65324). Sixteen areas were designated in the states of Washington, Oregon, and California (Figure 62). These areas include: the Mad River, CA, Redwood Creek, CA, Klamath River, CA, Umpqua River/Winchester Bay, OR, Tenmile Creek, OR, Sandy River, OR, Lower Columbia River, OR and WA, Grays River, WA, Skamokawa Creek, WA, Elochoman River, WA, Cowlitz River, WA, Toutle River, WA, Kalama River, WA, Lewis River, WA, Quinault River, WA, and the Elwha River, WA. The designated areas are a combination of freshwater creeks and rivers

and their associated estuaries, comprising approximately 539 km (335 mi) of habitat. Designated critical habitat for Southern DPS eulachon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The physical or biological features essential to the conservation of the DPS include:

- Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles.
- Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted.
- Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

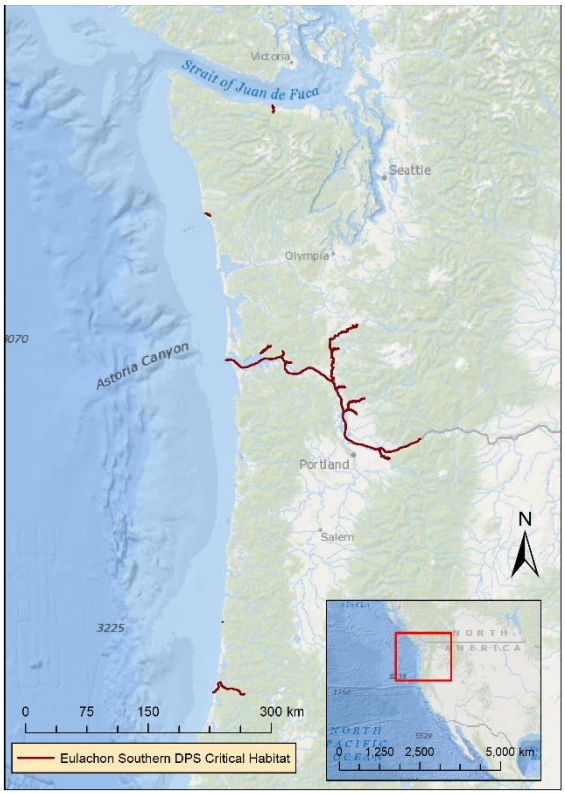


Figure 62. Map of designated critical habitat for the eulachon Southern distinct population segment.

Recovery Goals

See NMFS (2017e) for complete down listing/delisting criteria for the Southern DPS of eulachon. The following items were the top recovery goals and objectives listed in the 2017

Recovery Plan for the Southern DPS of eulachon identified to support in the recovery of the threatened population:

- Increase the abundance and productivity of eulachon;
- Ensure subpopulation viability;
- Protect and enhance the genetic, life history, and spatial diversity of eulachon throughout its geographical range;
- Conserve spatial structure and temporal distribution patterns;
- Conserve existing genetic and life history diversity and provide opportunities for interchange of genetic material between and within subpopulations;
- Reduce existing threats to warrant delisting of the species; and
- Eliminate or sufficiently reduce the severity of threats

6.2.25 Green Sturgeon – Southern DPS

The North American green sturgeon, *Acipenser medirostris*, is an anadromous fish that occurs in the nearshore Eastern Pacific Ocean from Alaska to Mexico (Moyle 2002) (Figure 63). Green sturgeon are long-lived, late-maturing, iteroparous, anadromous species that spawn infrequently in natal streams, and spend substantial portions of their lives in marine waters. Although they are members of the class of bony fishes, the skeleton of sturgeons is composed mostly of cartilage. Sturgeon have five rows of characteristic bony plates on their body (called scutes). Green sturgeon have an olive green to dark green back, a yellowish green-white belly, and a white stripe beneath the lateral scutes (Adams et al. 2002).

NMFS has identified two distinct population segments (DPS) of green sturgeon; northern and southern (Israel et al. 2009). In 2006, NMFS determined that the Southern DPS of green sturgeon warranted listing as a threatened species under the ESA (71 FR 17757). Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the US, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco Bay and Monterey Bay (Huff et al. 2012; Lindley et al. 2011; Lindley et al. 2008; Moser and Lindley 2007).

Life History

Green sturgeon can live to be 70 years old. Green sturgeon reach sexual maturity at approximately fifteen years of age (Van Eenennaam et al. 2006), and may spawn every two to four years throughout their long lives (Erickson and Webb 2007). Mora et al. (2018) reported a mean spawning periodicity of 3.69 years based on a tagging study of adult green sturgeon. The Southern DPS of green sturgeon spawn in cool (14-17°C), deep, turbulent areas with clean, hard substrates.

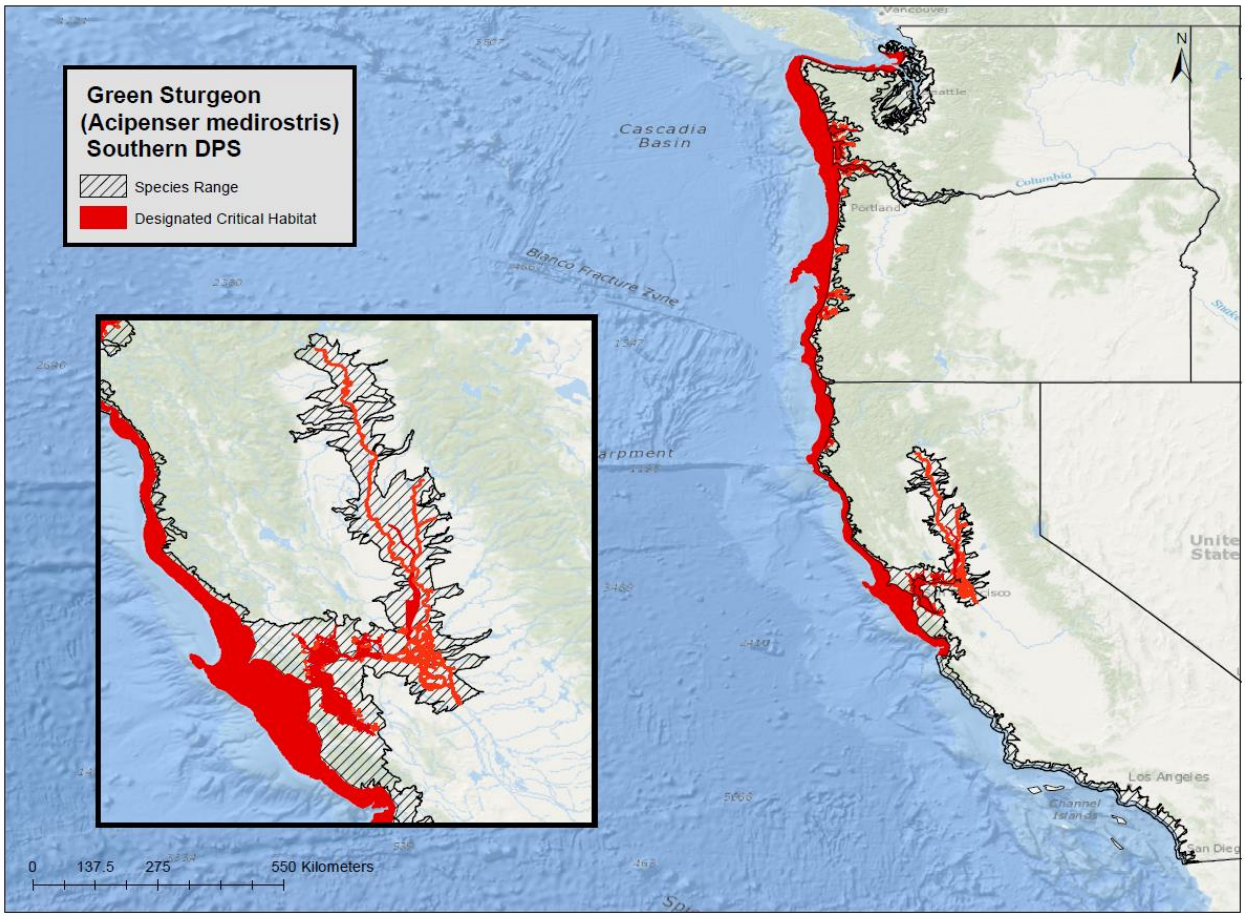


Figure 63. Geographic range (within the contiguous U.S., not including Alaska range) and designated critical habitat for the green sturgeon Southern DPS.

By far, the Sacramento River is the largest known spawning river for the southern DPS. Six discrete spawning sites have been identified in the upper Sacramento River between Gianella Bridge (River Kilometer 320.6) and the Keswick dam (River Kilometer 486) (Poytress et al. 2013). Some minor spawning takes place in the Feather River, with between 21 to 28 sturgeon observed in 2011, and fertilized eggs on egg mats found (Seesholtz et al. 2015). Spawning pairs of green sturgeon were captured on video at the foot of a dam in the Yuba River in 2011 (Bergman et al. 2011). In preparation for spawning, adult Southern DPS green sturgeon enter San Francisco Bay between mid-February and early-May, and migrate rapidly (on the order of a few weeks) up the Sacramento River (Heublein et al. 2009). Spawning occurs from April through early July, with peaks of activity that depend on a variety of factors including water temperature and water flow rates (Poytress et al. 2009; Poytress et al. 2010).

Green sturgeon larvae are different from all other sturgeon because of the absence of a distinct swim-up or post-hatching stage. Larvae grow fast; young fish grow to 74 millimeters 45 days after hatching (Deng 2000). Larvae and juveniles migrate downstream toward the Sacramento-

San Joaquin Delta/Estuary, where they rear for one to four years before migrating out to the Pacific Ocean as subadults (Nakamoto et al. 1995).

After spawning, green sturgeon typically congregate and hold for several months in a few deep pools in the upper mainstem Sacramento River near spawning sites and migrate back downstream when river flows increase in fall. Post-spawn fish re-enter the ocean during the winter months (November through January) and begin their seasonal marine migration north along the coast. Green sturgeon spend the majority of their lives in the open ocean where they exhibit migratory behavior not associated with spawning. Lindley et al. (2008) found that green sturgeon migrate annually along the continental shelf from U.S. to Canadian waters in the fall and an apparent return migration in the spring. Green sturgeon in marine habitats are primarily found at depths of 20–60 meters and from 9.5–16.0 degrees C (Huff et al. 2011). Huff et al. (2011) found that green sturgeon, on average, spent a longer duration in areas with high seafloor complexity, especially where a greater proportion of the substrate consists of boulders. During summer and fall, green sturgeon congregate in coastal bays and estuaries of Washington, Oregon, and California. In winter and spring, similar aggregations can be found from Vancouver Island to Hecate Strait, British Columbia, Canada (Lindley et al. 2008).

Adults captured in the Sacramento-San Joaquin Delta are benthic feeders on invertebrates including shrimp, mollusks, amphipods, and even small fish (Houston 1988; Moyle et al. 1992a). Juveniles in the Sacramento River delta feed on opossum shrimp, *Neomysis mercedis*, and *Corophium* amphipods (Radtke 1966). Green sturgeon in Willapa Bay, Washington, eat burrowing shrimp (*Neotrypaea californiensis*) (Borin et al. 2017).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Southern DPS of green sturgeon.

Mora et al. (2018) used dual-frequency identification sonar sampling in the Sacramento River for five years between 2010 and 2015 to estimate spawning run size and population size of the Southern DPS green sturgeon. Southern DPS spawning run size varied across years, from a minimum of 336 to a maximum of 1,236 individuals. The total population size for the Sacramento River was estimated at 17,548 individuals (95 percent confidence interval [CI] = 12,614 to 22,482). The study also estimated the number of juveniles (freshwater stage, less than 60 cm length, and one to three years of age), sub-adults (3-20 years and 60-165 cm length), and adults (greater than 165 cm in length and older than 20 years) in the river. There are an estimated 4,387 juveniles (95 percent CI = 2,595 to 6,179), an estimated 11,055 subadults (95 percent CI = 6,540 to 15,571), and an estimated 2,106 adults (95 percent CI = 1,246 to 2,966) in the Sacramento River (Mora et al. 2018).

Distribution within the Action Area

Southern DPS green sturgeon occur in both the offshore and inland portions of the NWTT action area. NMFS operates 115 tag detection receivers off the coast of Washington State. Although they have been detected on the outer continental shelf at depths up to 200 m, green sturgeon in marine habitats are primarily found at depths of 20–60 meters and from 9.5–16.0°C (Huff et al. 2011). Figure 64 below shows the number of green sturgeon detections at these receivers from May 2019 to September 2019. The number inside each circle is the number of unique individual green sturgeon (either DPS) detected at each receiver location, and the black line is the border of the NWTT action area. When analyzing green sturgeon tag detection data, it is important to note that only a small proportion of the Southern DPS has been tagged. Therefore, the actually number of fish from this DPS in the locations where the receivers can detect is several times greater than the number of tag detections in the database. Based on preliminary analysis, we expect about one-half of the green sturgeon found in this area to be from the ESA-listed Southern DPS, with the other half from the Northern DPS (M. Moser, NMFS, pers. comm. to R. Salz, NMFS, January 14, 2020). This is consistent with other findings which show that fish from both DPSs make these large coastal movements from their natal spawning grounds and can be found in roughly equal numbers of the Washington and Oregon coasts (Lindley et al. 2011; Richerson et al. 2019). Southern DPS green sturgeon have been detected by acoustic receivers off the coast of Washington (including within the Quinault Range Site) during the fall, winter, and spring months (J.M. Smith, NMFS unpublished data provided to R. Salz, NMFS, May 15, 2020). Animations of individual tagged fish in this area primarily show a southward movement along the coast from early May to about mid-June (J.M. Smith, NMFS, pers. comm. to R. Salz, NMFS, January 20, 2020). By late June, the tagged green sturgeon are no longer detected by the receivers in the action area off the coast of Washington, as most move into shallower, nearshore coastal waters during the summer months (Huff et al. 2011).

Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the U.S. west coast, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco Bay and Monterey Bay. Tagged Southern DPS green sturgeon subadults and adults have been detected in coastal marine waters from Monterey Bay, California to Graves Harbor, Alaska, including the Strait of Juan de Fuca and Puget Sound (74 FR 52299; AquaMaps 2016; Lindley et al. 2011), Washington estuaries within the action area for the proposed action. Recent tag detection results indicate that relatively large numbers of green sturgeon are migrating into the Strait of Juan de Fuca. Preliminary data from a hydrophone array (30 hydrophones) positioned across the Strait of Juan de Fuca showed a total of 64 individual green sturgeon detections from October 2012 to October 2014 (Moore 2020). Of the 64 green sturgeon detected at the Strait of Juan de Fuca array, 18 were identified as being part of the ESA-listed Southern DPS, 16 as being part of the Northern DPS, and 30 were of unknown origin (M. Moser, NMFS, pers. comm. to R. Salz, NMFS, February 20, 2020).

Lindley et al. (2011) reported that green sturgeon use the Puget Sound estuary at a low rate, but fish were detected within this estuary in both winter and summer months. Preliminary data from a hydrophone array (30 hydrophones) positioned across Admiralty Inlet, which connects the Strait of Juan de Fuca to Puget Sound, showed two green sturgeon detections from February 2014 through August 2019 (Moore 2020). One of these detections was a Southern DPS female, the other was of unknown origin (M. Moser, NMFS, pers. comm. to R. Salz, NMFS, February 20, 2020). There were no detections of tagged green sturgeon during this time frame at hydrophone arrays across Central Puget Sound or Tacoma Narrows (Moore 2020). NMFS also deployed an array of over 30 hydrophones surrounding the Hood Canal Bridge and four hydrophones spanning the outlet of Hood Canal from March through August of 2017 and March through September of 2018. There were no detections of tagged Southern DPS green sturgeon during this time frame at the Hood Canal array (Moore 2020).

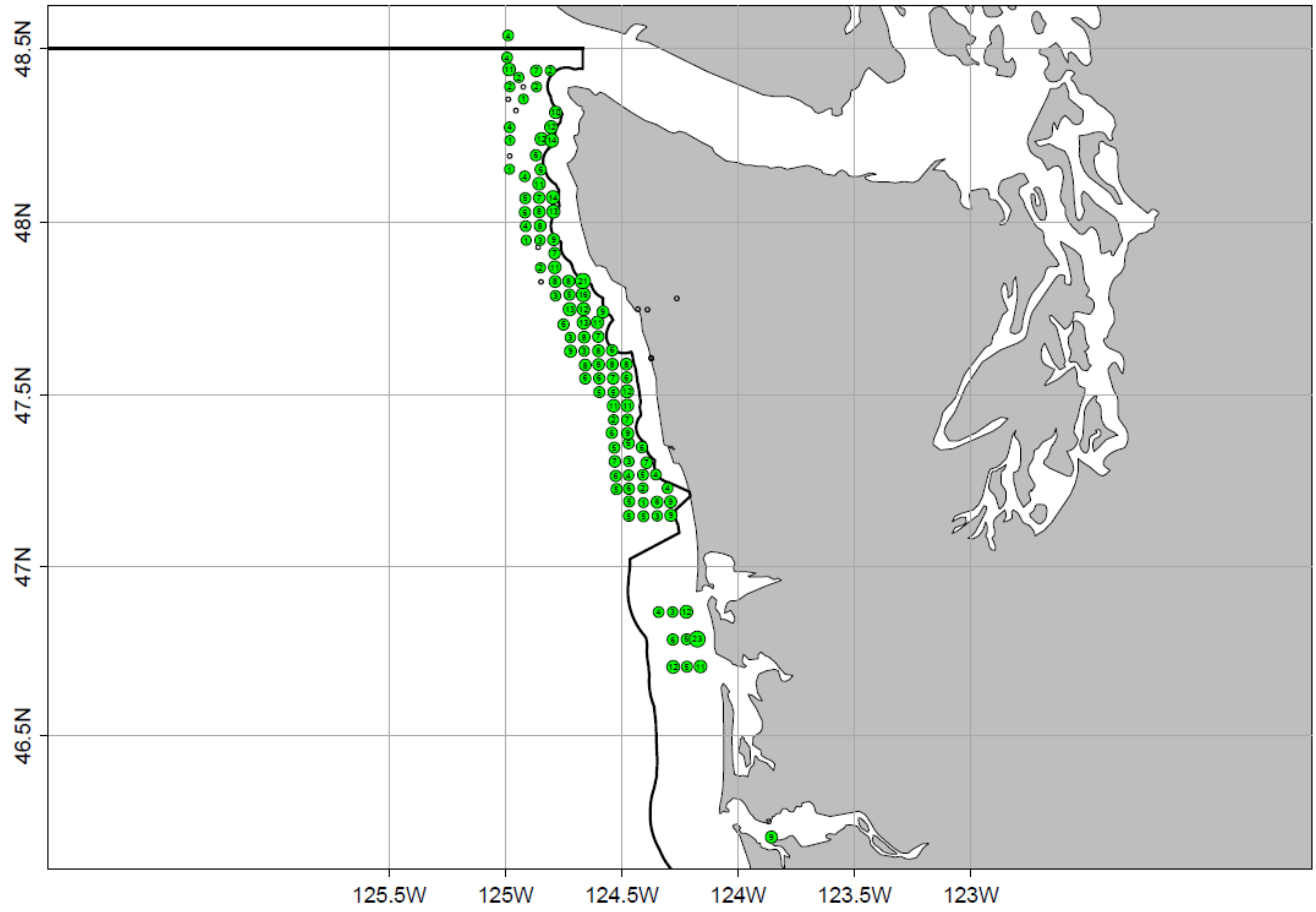


Figure 64. Number of individual green sturgeon detections at receivers off the coast of Washington State from May 2019 to September 2019 (unpublished map provided by Joe Smith, NMFS Northwest Fisheries Science Center, January 2, 2020).

Status

Green sturgeon stocks from the two identified DPSs (i.e., Southern and Northern) have been found to be genetically differentiated (Israel et al. 2009; Israel et al. 2004). The available genetic data do not change the status of the species or the imminence or magnitude of any threat; data only confirm the DPS structure and add detail to the DPS composition in different estuaries during the sampling periods (NMFS 2015e). Attempts to evaluate the status of Southern DPS green sturgeon have been met with limited success due to the lack of reliable long-term data. No estimate of intrinsic growth rates are available for Southern DPS green sturgeon. The final rule listing southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River caused primarily by impoundments. The species also faces threats from changes in water temperature, availability, and flow, and commercial and recreational bycatch (Doukakis et al. 2020; 71 FR 17757). Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015e). Rodgers et al. (2019) used a meta-analytical approach to summarize the mean effects of prominent stressors (elevated temperatures, salinity, low food availability and contaminants) on several physiological traits (growth, thermal tolerance, swimming performance and heat shock protein expression) of Southern DPS green sturgeon. All examined stressors significantly impaired green sturgeon growth, and additional stressor-specific costs were documented (Rodgers et al. 2019).

Critical Habitat

On October 9, 2009, NMFS designated critical habitat for the Southern DPS of green sturgeon (74 FR 52300). Critical habitat is designated in coastal U.S. marine waters within 60 fathoms depth from Monterey Bay, California (including Monterey Bay), north to Cape Flattery, Washington, including the Strait of Juan de Fuca, Washington, to its United States boundary; the Sacramento River, lower Feather River, and lower Yuba River in California; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; the lower Columbia River estuary; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor) (Figure 65) (USDC 2009).

Offshore coastal marine areas and the Strait of Juan de Fuca co-occur with portions of the NWTT action area. The physical and biological features for Southern DPS green sturgeon designated critical habitat in the coastal marine and estuarine areas are described in Table 87.

Prey species for juvenile, subadult, and adult green sturgeon within bays and estuaries primarily consist of benthic invertebrates and fish, including crangonid shrimp, burrowing thalassinidean shrimp, amphipods, isopods, clams, annelid worms, crabs, sand lances, and anchovies. These prey species are critical for the rearing, foraging, growth, and development of juvenile, subadult,

and adult green sturgeon within the bays and estuaries. Green sturgeon spend most of their lives in marine and estuarine waters along the coast where they require abundant food items to support subadults and adults over long-distance migrations.

Migratory corridor refers to a migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats. Safe and timely passage is defined to mean that human-induced impediments, either physical, chemical, or biological, do not alter the migratory behavior of the fish such that its survival or the overall viability of the species is compromised (e.g., an impediment that compromises the ability of fish to reach thermal refugia by the time they enter a particular life stage). Unimpeded passage is necessary in bays and estuaries for adult and subadult green sturgeon to access feeding areas, holding areas, and thermal refugia, and to ensure passage back out into the ocean.

Table 87. Physical and biological features of marine and estuarine critical habitat designated for Southern DPS green sturgeon and corresponding species life history events.

Physical or Biological Features		Species Life History Event
Site Type	Site Attribute	
Estuarine areas	Food resources Migratory corridor Sediment quality Water flow Water depth Water quality	Juvenile growth, development, seaward migration Subadult growth, development, seasonal holding, and movement between estuarine and marine areas Adult growth, development, seasonal holding, movements between estuarine and marine areas, upstream spawning movement, and seaward post-spawning movement
Coastal marine areas	Food resources Migratory corridor Water quality	Subadult growth and development, movement between estuarine and marine areas, and migration between marine areas Adult sexual maturation, growth and development, movements between estuarine and marine areas, migration between marine areas, and spawning migration

Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, refers to coastal marine waters with adequate dissolved oxygen levels and with acceptably low levels of contaminants (such as pesticides, polycyclic aromatic hydrocarbons [PAHs], elevated levels of heavy metals) that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon. Suitable water temperatures for juvenile green sturgeon should be below 24°C.

Water flow and water depth were also identified as important attributes of particular estuarine areas. The bays and estuaries adjacent to the Sacramento River (i.e., the Sacramento-San Joaquin

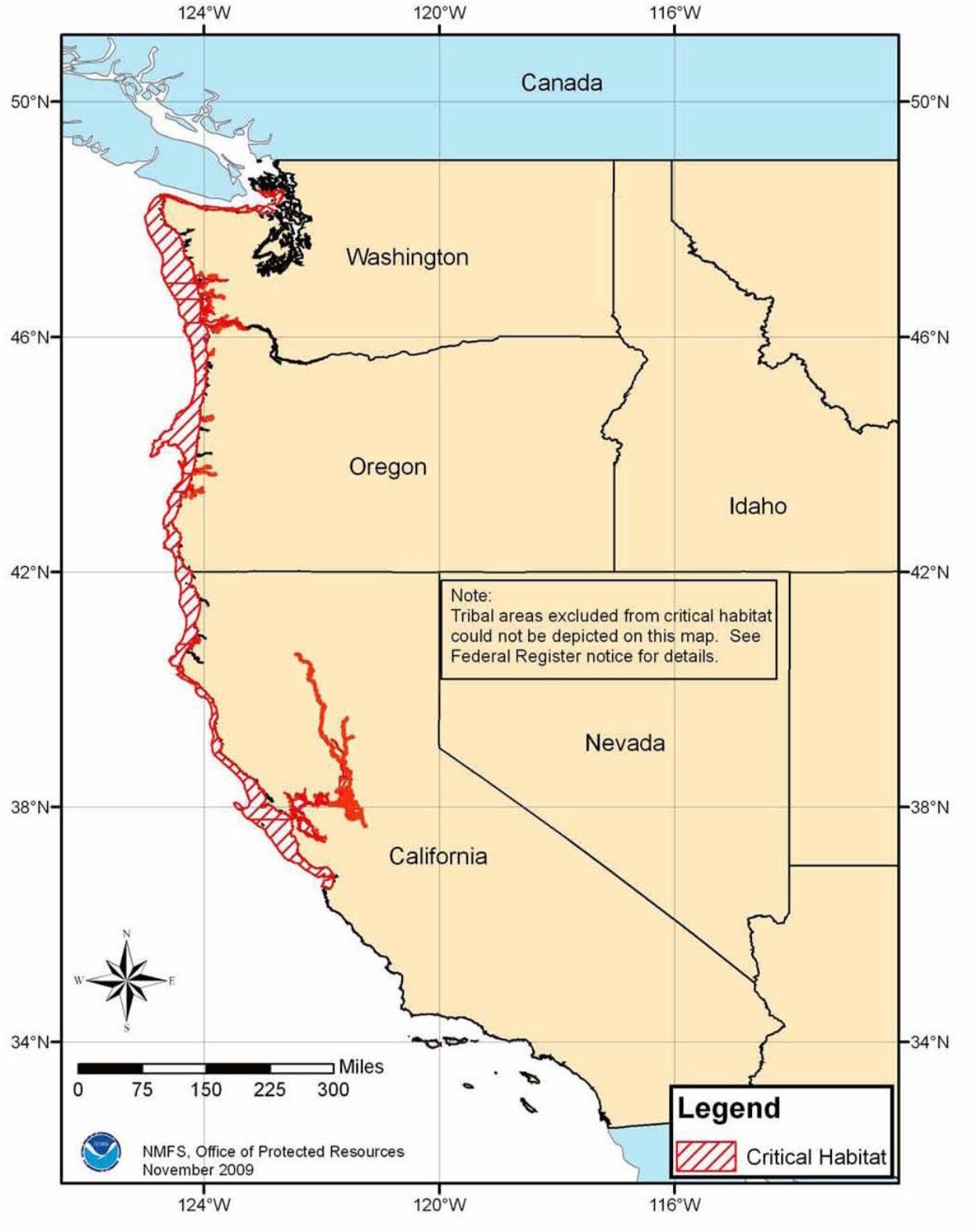


Figure 65. Green sturgeon Southern DPS critical habitat.

Delta and the Suisun, San Pablo, and San Francisco bays) must also have sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds. Subadult and adult green sturgeon occupy a diversity of waters depths within bays and estuaries for feeding and migration.

Sediment quality (i.e., chemical characteristics) is necessary for normal behavior, growth, and viability of all green sturgeon life stages. This includes sediments free of elevated levels of contaminants (e.g., elevated levels of selenium, PAHs, and organochlorine pesticides) that can cause adverse effects on all life stages of green sturgeon.

Several activities threaten Southern DPS green sturgeon PBFs in coastal bays and estuaries. The application of pesticides is likely to adversely affect prey resources and water quality within the bays and estuaries, as well as the growth and reproductive health of Southern DPS green sturgeon through bioaccumulation. Temperature and selenium are two environmental parameters that potentially affect reproduction and stock recruitment of female green sturgeon (Silvestre et al. 2010). Other activities of concern include those that disturb bottom substrates, adversely affect prey resources, or degrade water quality through re-suspension of contaminated sediments. Of particular concern are activities that affect prey resources. Prey resources are affected by: commercial shipping and activities generating point source pollution and non-point source pollution that discharge contaminants and result in bioaccumulation of contaminants in green sturgeon; disposal of dredged materials that bury prey resources; and bottom trawl fisheries that disturb the bottom (but result in beneficial or adverse effects on prey resources for green sturgeon).

Recovery Goals

NMFS (2018d) identified de-listing criteria crucial for the recovery of the Southern DPS of green sturgeon. These criteria for recovery are listed below:

- Abundance. The adult Southern DPS of green sturgeon census population remains at or above 3,000 for 3 generations (this equates to a yearly running average of at least 813 spawners for approximately 66 years). In addition, the effective population size must be at least 500 individuals in any given year and each annual spawning run must be comprised of a combined total, from all spawning locations, of at least 500 adult fish in any given year.
- Distribution. The Southern DPS of green sturgeon spawn successfully in at least two rivers within their historical range. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
- Productivity. A net positive trend in juvenile and subadult abundance is observed over the course of at least 20 years. Also, the population is characterized by a broad distribution of size classes representing multiple cohorts that are stable over the long term (20 years or more).

- Diversity. There is no net loss of Southern DPS green sturgeon diversity from current levels.
- Threat-Based Recovery Criteria
    - Access to spawning habitat is improved through barrier removal or modification in the Sacramento, Feather, and/or Yuba rivers such that successful spawning occurs annually in at least two rivers. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
    - Volitional passage is provided for adult green sturgeon through the Yolo and Sutter bypasses.
    - Water temperature and flows are provided in spawning habitat such that juvenile recruitment is documented annually. Recruitment is determined by the annual presence of age-0 juveniles in the lower Sacramento River or San Francisco Bay Delta Estuary. Flow and temperature guidelines have been derived from analysis of inter-annual spawning and recruitment success and are informing this criterion.
    - Adult contaminant levels are below levels that are identified as limiting population maintenance and growth.
    - Take of adults and subadults through poaching and state, federal, and tribal fisheries is minimal and does not limit population persistence and growth.

6.2.26 Sockeye Salmon – Snake River ESU

This evolutionarily significant unit, or ESU, includes naturally spawned anadromous and residual sockeye salmon originating from the Snake River basin (Figure 66), and also sockeye salmon from one artificial propagation program: Redfish Lake Captive Broodstock Program.

A general description of sockeye salmon is presented in Section 6.1.4. On November 20, 1991 NMFS listed the Snake River sockeye salmon ESU as endangered (56 FR 58619), (*Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs*)and reaffirmed the ESU's status as endangered on June 28, 2005.

Life History

The life history sockeye salmon is presented in Section 6.1.4.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Snake River ESU of sockeye salmon.

Adult returns over the last several years have ranged from a high of 1,579 fish in 2014 (including 453 natural-origin fish) to a low of 257 adults in 2012 (including 52 natural-origin fish). Sockeye salmon returns to Alturas Lake ranged from one fish in 2002 to 14 fish in 2010. No fish returned

to Alturas Lake in 2012, 2013, or 2014 (NMFS 2015b). Current abundance estimates for the Snake River ESU of sockeye salmon are presented in Table 88 below.

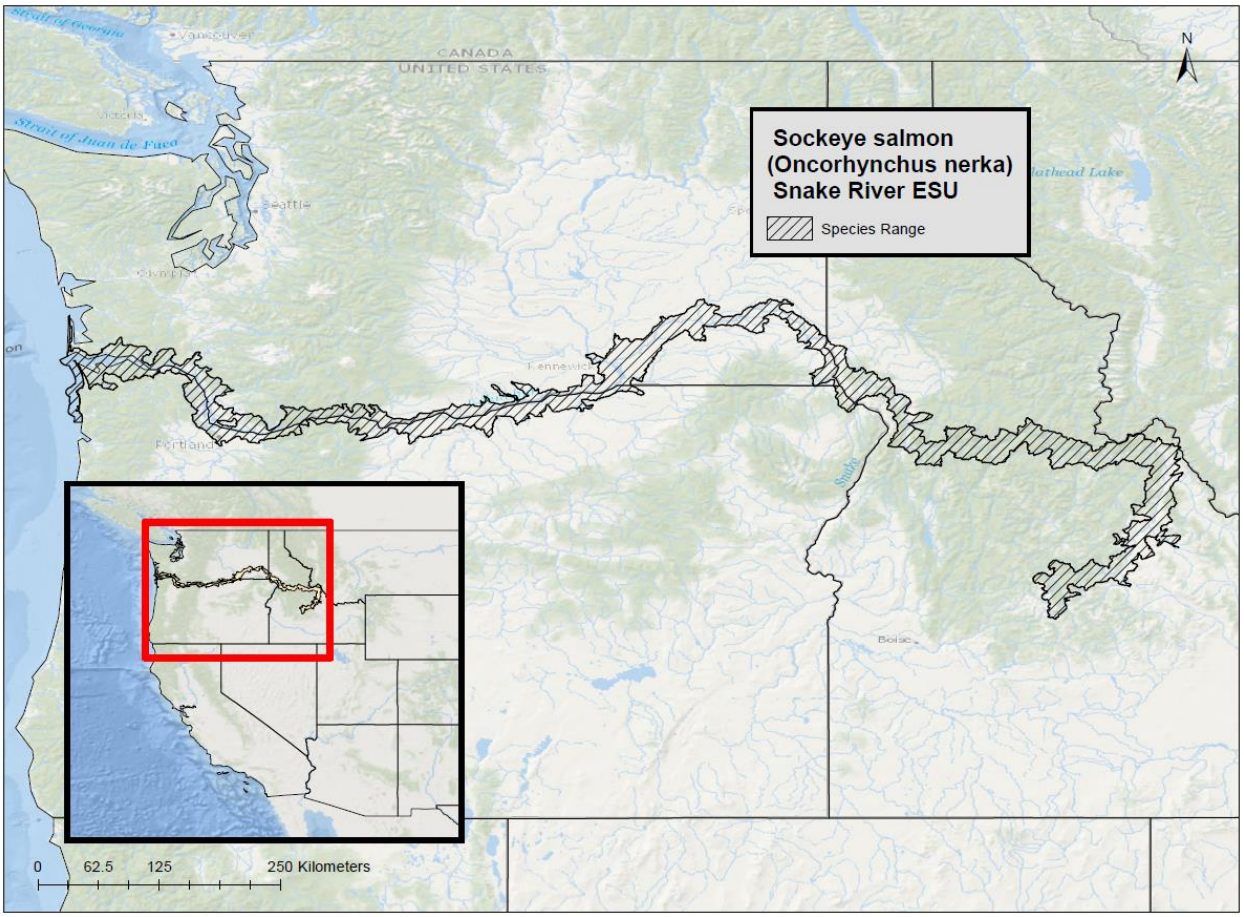


Figure 66. Geographic range of Sockeye salmon, Snake River ESU.

Table 88. Current Abundance Estimates for Snake River ESU Sockeye salmon (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	546
Natural	Juvenile	19,181
Listed Hatchery Adipose Clipped	Adult	4,004
Listed Hatchery Adipose Clipped	Juvenile	242,610

The large increases in returning adults in recent years reflect improved downstream and ocean survival as well as increases in juvenile production since the early 1990s. Although total sockeye salmon returns to the Sawtooth Valley in recent years have been high enough to allow for some

level of natural spawning in Redfish Lake, the hatchery program remains at its initial phase with a priority on genetic conservation and building sufficient returns to support sustained outplanting and recolonization of the species' historic range (NMFS 2015b; NWFSC 2015b).

For the Snake River ESU, the Sawtooth Hatchery is focusing on genetic conservation. An overrepresentation of genes from the anadromous population in Redfish Lake exists, but inbreeding is low, which is a sign of a successful captive broodstock program (NMFS 2015b; NWFSC 2015b) .

This species includes all anadromous and residual sockeye salmon from the Snake River basin, Idaho, and artificially-propagated sockeye salmon from the Redfish Lake Captive Broodstock Program (NMFS 2015b; NWFSC 2015b; USDC 2014). The ICTRT treats Sawtooth Valley Sockeye salmon as the single MPG within the Snake River Sockeye salmon ESU. The MPG contains one extant population (Redfish Lake) and two to four historical populations (Alturas, Petit, Stanley, and Yellowbelly Lakes) (NMFS 2015b). At the time of listing in 1991, the only confirmed extant population included in this ESU was the beach-spawning population of sockeye salmon from Redfish Lake, with about 10 fish returning per year (NMFS 2015b).

Status

The Snake River sockeye salmon ESU includes only one population comprised of all anadromous and residual sockeye salmon from the Snake River Basin, Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. Historical evidence indicates that the Snake River sockeye salmon once had a range of life history patterns, with spawning populations present in several of the small lakes in the Sawtooth Basin. NMFS listed the Snake River sockeye salmon ESU because of habitat loss and degradation from the combined effects of damming and hydropower development, overexploitation, fisheries management practices, and poor ocean conditions. Recent effects of climate change, such as reduced stream flows and increased water temperatures, are limiting Snake River ESU productivity (NMFS 2015b; NWFSC 2015b). Adults produced through the captive propagation program currently support the entire ESU. This ESU is still at extremely high risk across all four basic risk measures (abundance, productivity, spatial structure, and diversity) and would likely have a very low resilience to additional perturbations. Habitat improvement projects have slightly decreased the risk to the species, but habitat concerns and water temperature issues remain. Overall, although the status of the Snake River sockeye salmon ESU appears to be improving, there is no indication that the biological risk category has changed (NWFSC 2015b).

Critical Habitat

NMFS designated critical habitat for Snake River sockeye salmon on December 28, 1993 (58 FR 68543). The critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to salmon of this ESU (except reaches above impassable natural falls, and Dworshak and Hells

Canyon Dams). Specific PBFs were not designated in the critical habitat final rule; instead, four "essential habitat" categories were described: 1) spawning and juvenile rearing areas, 2) juvenile migration corridors, 3) areas for growth and development to adulthood, and 4) adult migration corridors. Designated critical habitat for the Snake River sockeye salmon does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

See the 2015 recovery plan for the Snake River sockeye salmon ESU for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2015b). Broadly, recovery plan goals emphasize restoring historical lake populations and improving water quality and quantity in lakes and migration corridors.

6.2.27 Steelhead – California Central Valley DPS

The Central Valley DPS of steelhead includes naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Sacramento and San Joaquin Rivers and their tributaries and excludes such fish originating from San Francisco and San Pablo Bays and their tributaries (Figure 67). Further the Central Valley DPS of steelhead includes steelhead from two artificial propagation programs.

A physical description of steelhead is presented in Section 6.1. On March 19, 1998 NMFS listed the California Central Valley DPS of steelhead as threatened (63 FR 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

The Central Valley DPS of steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems. The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000). The eggs hatch in three to four weeks at 50°F to 59°F, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

Steelhead typically migrate to marine waters after spending two years in fresh water. They reside in marine waters for typically two or three years prior to returning to their natal stream to spawn as four- or five-year olds. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002). Currently, Central Valley steelhead are considered "ocean-maturing" (also known as winter) steelhead, although summer steelhead may

have been present prior to construction of large dams (Moyle 2002). Ocean maturing steelhead enter fresh water with well-developed gonads and spawn shortly after river entry. Central Valley steelhead enter fresh water from August through April. They hold until flows are high enough in tributaries to enter for spawning (Moyle 2002). Steelhead adults typically spawn from December through April, with peaks from January through March in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961; McEwan 2001).

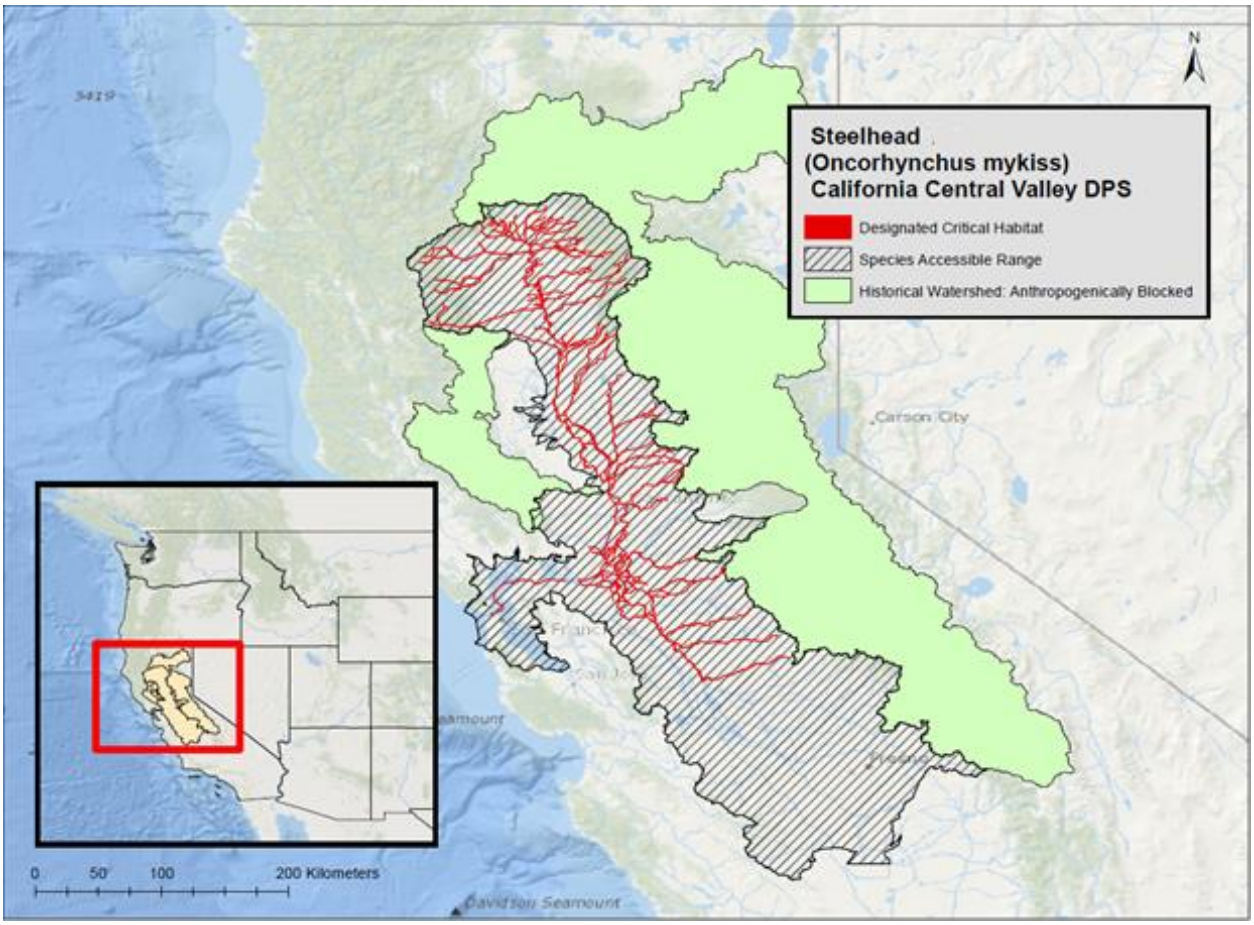


Figure 67. Geographic range and designated critical habitat of California Central Valley Steelhead.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to California Central Valley Steelhead.

Historic Central Valley steelhead run size may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead

populations in the upper Sacramento River have declined substantially. Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of Central Valley steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005). Current abundance estimates for the California Central Valley ESU of steelhead are presented in Table 89 below.

Table 89. Current Abundance Estimates for the California Central Valley ESU of Steelhead (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	1,686
Natural	Juvenile	630,403
Listed Hatchery Adipose Clipped	Adult	3,856
Listed Hatchery Adipose Clipped	Juvenile	1,600,653

California Central Valley steelhead lack annual monitoring data for calculating trends. However, the Red Bluff Diversion Dam counts and redd counts up to 1993 and later sporadic data show that the DPS has had a significant long-term downward trend in abundance (NMFS 2009a).

The Central Valley steelhead distribution ranges over a wide variety of environmental conditions and likely contains biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006). The loss of populations and reduction in abundances have reduced the large diversity that existed within the DPS. The genetic diversity of the majority of steelhead spawning runs within this DPS is also compromised by hatchery-origin fish.

Central Valley steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems.

Status

Many watersheds in the Central Valley are experiencing decreased abundance of California Central Valley steelhead. Dam removal and habitat restoration efforts in Clear Creek appear to be benefiting steelhead as recent increases in non-clipped (wild) abundance have been observed. Despite the positive trend in Clear Creek, all other concerns raised in the previous status review remain, including low adult abundances, loss and degradation of a large percentage of the historic spawning and rearing habitat, and domination of smolt production by hatchery fish. Many other planned restoration and reintroduction efforts have yet to be implemented or completed, or are focused on Chinook salmon, and have yet to yield demonstrable improvements in habitat, let alone documented increases in naturally produced steelhead. There are indications that natural production of steelhead continues to decline and is now at a very low levels. Their continued low numbers in most hatcheries, domination by hatchery fish, and relatively sparse

monitoring makes the continued existence of naturally reproduced steelhead a concern. California Central Valley steelhead is likely to become endangered within the foreseeable future throughout all or a significant portion of its range.

Critical Habitat

NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the California Central Valley steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The spawning PBFs are subject to variations in flows and temperatures, particularly over the summer months. The rearing PBFs are degraded by the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system and which typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Stream channels commonly have elevated temperatures. Both migration and rearing PBFs are affected by dense urbanization and agriculture along the mainstem and in the Delta which contribute to reduced water quality by introducing several contaminants. In the Sacramento River, the migration corridor for both juveniles and adults is obstructed by the RBDD gates which are down from May 15 through September 15. The migration PBF is also obstructed by complex channel configuration making it more difficult for steelhead to migrate successfully to the western Delta and the ocean. The estuarine PBF, which is present in the Delta, is affected by contaminants from agricultural and urban runoff and release of wastewater treatment plants effluent.

Recovery Goals

See the 2014 recovery plan for the California Central Valley steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2014c). The delisting criteria for this DPS are:

- One population in the Northwestern California Diversity Group at low risk of extinction
- Two populations in the Basalt and Porous Lava Flow Diversity Group at low risk of extinction
- Four populations in the Northern Sierra Diversity Group at low risk of extinction
- Two populations in the Southern Sierra Diversity Group at low risk of extinction
- Maintain multiple populations at moderate risk of extinction

6.2.28 Steelhead – Central California Coast DPS

The Central California Coast DPS of steelhead includes all naturally spawned populations of steelhead (and their progeny) in streams from the Russian River to Aptos Creek, Santa Cruz County, California (inclusive). It also includes the drainages of San Francisco and San Pablo Bays (Figure 68).

A physical description of steelhead is presented in Section 6.1. On August 18, 1997 NMFS listed the Central California Coast DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

The Central California Coast DPS of steelhead is entirely composed of winter-run fish. Adults return to the Russian River and migrate upstream from December to April, and smolts emigrate between March and May (Hayes et al. 2004; Shapovalov and Taft 1954). Most spawning takes place from January through April. The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000).

The eggs hatch in three to four weeks at 50 degrees Fahrenheit to 59 degrees Fahrenheit, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

Steelhead typically migrate to marine waters after spending two years in fresh water. They reside in marine waters for typically two or three years prior to returning to their natal stream to spawn as four- or five-year olds. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002).While age of smoltification typically ranges for one to four years, recent studies indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age two (Sogard et al. 2009).

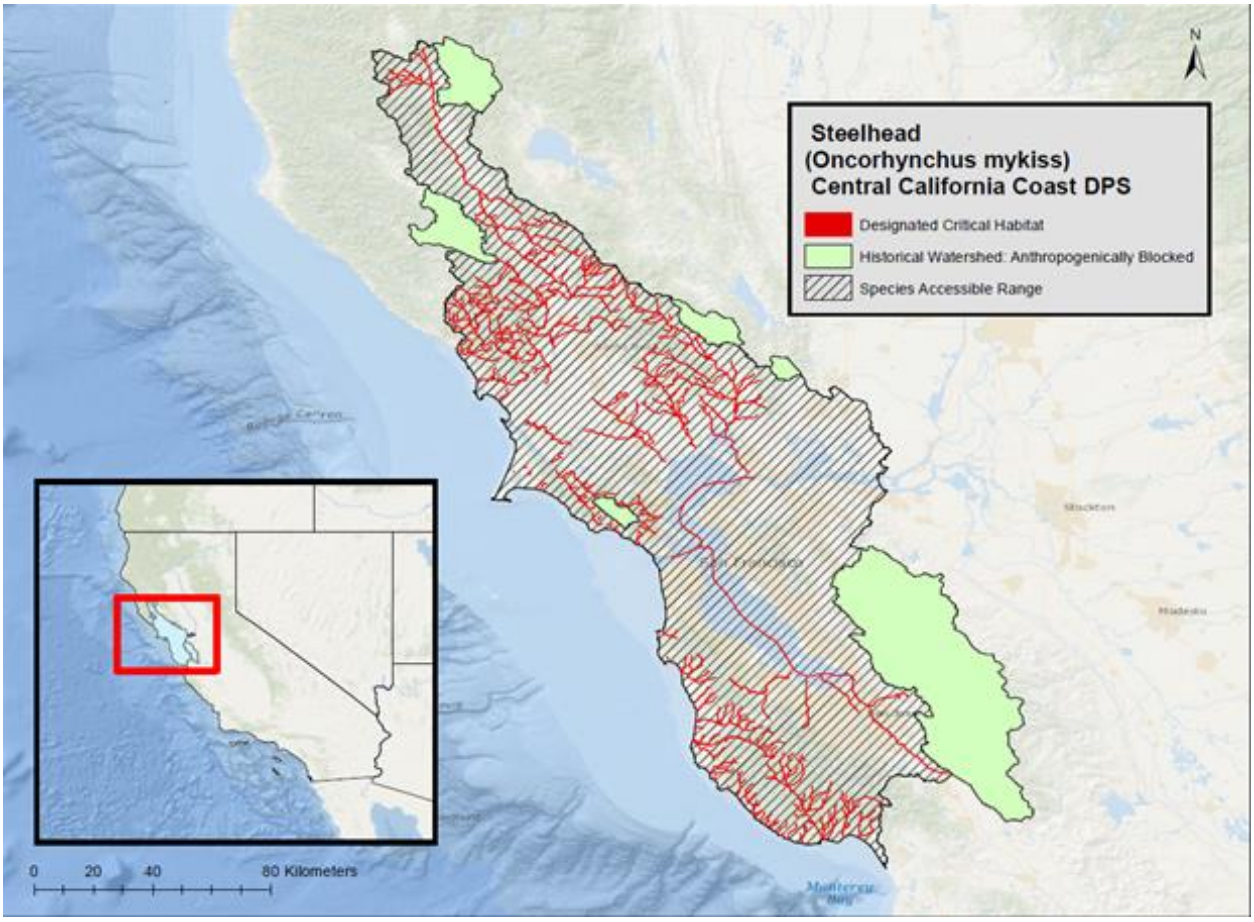


Figure 68. Geographic range and designated critical habitat of Central California Coast Steelhead.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, and genetic diversity as it relates to Central California Coast steelhead.

Historically, the entire Central California Coast steelhead DPS may have consisted of an average runs size of 94,000 adults in the early 1960s (Good et al. 2005). Current abundance estimates for the California Central Coast ESU of steelhead are presented in

Table 90 below. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good et al. 2005).

Table 90. Current Abundance Estimates for the California Central Coast ESU of Steelhead (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	2,187
Natural	Juvenile	248,771
Listed Hatchery Adipose Clipped	Adult	3,866
Listed Hatchery Adipose Clipped	Juvenile	648,891

Though the information for individual populations is limited, available information strongly suggests that no population is viable. Long-term population sustainability is extremely low for the southern populations in the Santa Cruz mountains and in the San Francisco Bay (NMFS 2008a). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good et al. 2005).

The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners. Due to this, Russian River winter-run steelhead may be able to be sustained over the long-term but hatchery management has eroded the population's genetic diversity (Bjorkstedt et al. 2005; NMFS 2008a).

Status

The Central California Coast steelhead consisted of nine historic functionally independent populations and 23 potentially independent populations (Bjorkstedt et al. 2005). Of the historic functionally independent populations, at least two are extirpated while most of the remaining are nearly extirpated. Current runs in the basins that originally contained the two largest steelhead populations for the DPS, the San Lorenzo and the Russian Rivers, both have been estimated at less than 15 percent of their abundances just 30 years earlier (Good et al. 2005). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of Central California Coast steelhead (NOAA 2013). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby et al. 1996; NMFS 2008a).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Central California Coast steelhead does not

overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

Streams throughout the critical habitat have reduced quality of spawning PBFs; sediment fines in spawning gravel have reduced the ability of the substrate attribute to provide well oxygenated and clean water to eggs and alevins. High proportions of fines in bottom substrate also reduce forage by limiting the production of aquatic stream insects adapted to running water. Elevated water temperatures and impaired water quality have further reduced the quality, quantity and function of the rearing PBFs within most streams. These impacts have diminished the ability of designated critical habitat to conserve the Central California Coast steelhead.

Recovery Goals

See the 2016 recovery plan for the Central California Coast steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. Recovery plan objectives are to:

- Reduce the present or threatened destruction, modification, or curtailment of habitat or range;
- Ameliorate utilization for commercial, recreational, scientific, or educational purposes;
- Abate disease and predation;
- Establish the adequacy of existing regulatory mechanisms for protecting Central California Coast steelhead now and into the future (i.e., post-delisting);
- Address other natural or manmade factors affecting the continued existence of Central California Coast steelhead;
- Ensure Central California Coast steelhead status is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

6.2.29 Steelhead – Lower Columbia River DPS

The Lower Columbia River DPS of steelhead includes naturally spawned steelhead originating below natural and manmade impassable barriers from rivers between the Cowlitz and Wind Rivers (inclusive) and the Willamette and Hood Rivers (inclusive) and excludes such fish originating from the upper Willamette River basin above Willamette Falls (Figure 69). The Lower Columbia River DPS also includes steelhead from seven artificial propagation programs.

A physical description of steelhead is presented in Section 6.1. On March 19, 1998 NMFS listed the Lower Columbia River DPS of steelhead as threatened (63 FR 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

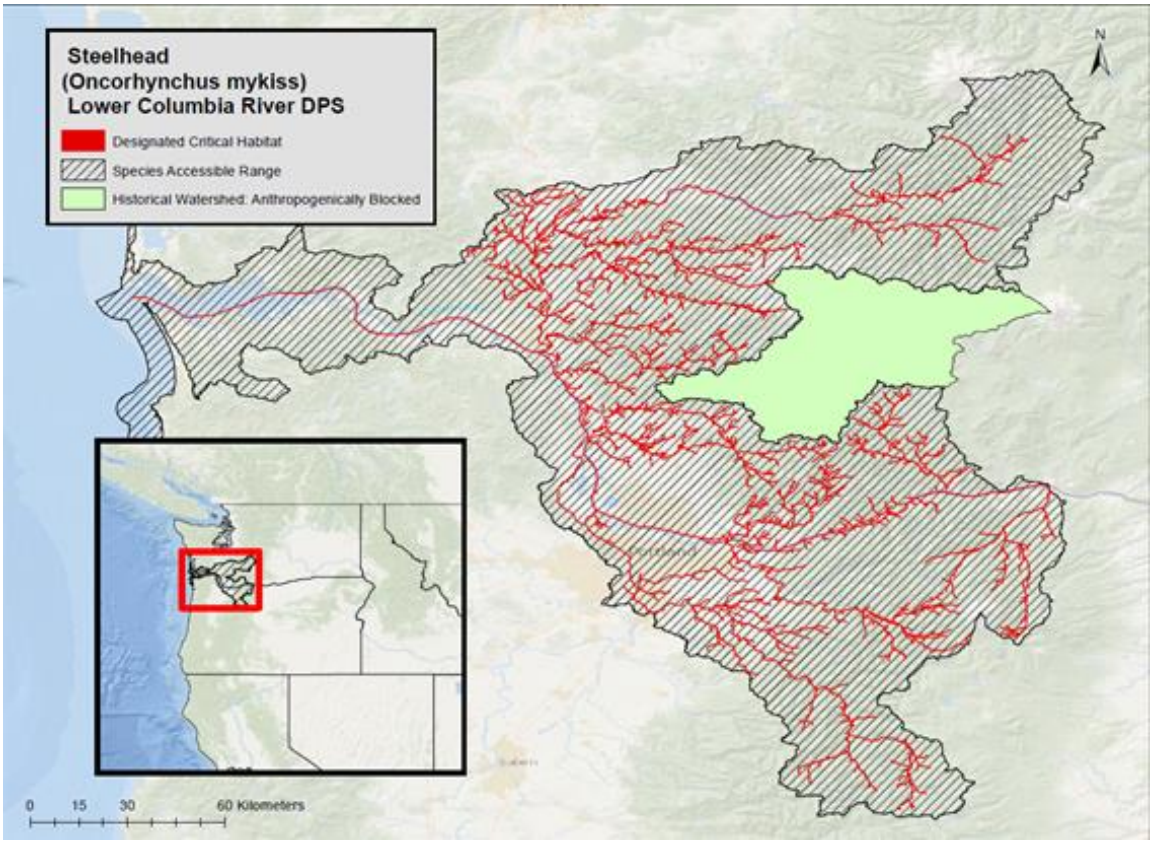


Figure 69. Geographic range and designated critical habitat of Lower Columbia River steelhead.

Life History

The Lower Columbia River steelhead DPS includes both summer- and winter-run stocks. Summer-run steelhead return sexually immature to the Columbia River from May to November, and spend several months in fresh water prior to spawning. Winter-run steelhead enter fresh water from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer-run steelhead tend to spawn at higher elevations than the winter-run. The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000). The eggs hatch in three to four weeks at 50°F to 59°F, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The

smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

The majority of juvenile lower Columbia River steelhead remain for two years in freshwater environments before ocean entry in spring. Both winter- and summer-run adults normally return after two years in the marine environment. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to lower Columbia River steelhead.

The Winter-run Western Cascade MPG includes native winter-run steelhead in 14 DIPs from the Cowlitz River to the Washougal River. Abundances have remained fairly stable and have remained low, averaging in the hundreds of fish. Notable exceptions to this were the Clackamas and Sandy River winter-run steelhead populations, that are exhibiting recent rises in NOR abundance and maintaining low levels of hatchery-origin steelhead on the spawning grounds (NMFS 2016d). In the Summer-run Cascade MPG, there are four summer-run steelhead populations. Absolute abundances have been in the hundreds of fish. In the Winter-run Gorge MPG both the Lower and Upper Gorge population surveys for winter steelhead are very limited and abundance levels in the Hood River have been low but relatively stable. In the Summer-run Gorge MPG adult abundance in the Wind River remains stable, but at a low level (hundreds of fish). Current abundance estimates for the Lower Columbia River DPS of steelhead are presented in Table 91 below. From 2015-2019, the geometric means for the releases from these hatcheries are 1,197,156 LHAC and 9,138 LHIA LCR steelhead annually (Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020). To estimate abundance of juvenile natural LCR steelhead, we calculate the geometric mean for outmigrating smolts over the past five years (2015-2019) by using annual abundance estimates provided by the NMFS' Northwest Fisheries Science Center (Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020). For juvenile natural-origin LCR steelhead, an estimated average of 352,146 juvenile steelhead outmigrated over the last five years.

Table 91. Current Abundance Estimates for the Lower Columbia River DPS of Steelhead (NMFS 2019c; Zabel 2015; Zabel 2017a; Zabel 2017b; Zabel 2018; Zabel 2020).

Production	Life Stage	Abundance
Natural	Adult	12,920
Natural	Juvenile	352,146

Production	Life Stage	Abundance
Listed Hatchery Adipose Clipped and Intact	Adult	22,297
Listed Hatchery Adipose Clipped	Juvenile	1,197,156
Listed Hatchery Intact Adipose	Juvenile	9,138

Population trends for the Winter-run Western Cascade MPG are fairly stable. Long and short term trends for three independent populations within the Summer-run Cascade MPG are positive; though the 2014 surveys indicate a drop in abundance for all three. Population trends in the Winter-run Gorge MPG is relatively stable. The overall status of the Summer-run Gorge MPG is uncertain.

Total steelhead hatchery releases in the Lower Columbia River Steelhead DPS have decreased since the last status review, declining from a total (summer and winter run) release of approximately 3.5 million to 3 million from 2008 to 2014. Some populations continue to have relatively high fractions of hatchery-origin spawners, whereas others (e.g., Wind River) have relatively few hatchery origin spawners.

There are four MPGs comprised of 23 DIPs, including six summer-run steelhead populations and 17 winter-run populations (NWFSC 2015b). Summer steelhead spawning areas in the Lower Columbia River are found above waterfalls and other features that create seasonal barriers to migration. There have been a number of large-scale efforts to improve accessibility (one of the primary metrics for spatial structure) in this ESU. Trap and haul operations were begun on the Lewis River in 2012 for winter-run steelhead, reestablishing access to historically-occupied habitat above Swift Dam. In 2014, 1033 adult winter steelhead (integrated program fish) were transported to the upper Lewis River; however, juvenile collection efficiency is still below target levels. In addition, there have been a number of recovery actions throughout the ESU to remove or improve culverts and other small-scale passage barriers.

Status

The Lower Columbia River steelhead had 17 historically independent winter steelhead populations and six independent summer steelhead populations (McElhany et al. 2003; Myers et al. 2006). All historic Lower Columbia River steelhead populations are considered extant. However, spatial structure within the historically independent populations, especially on the Washington side, has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydropower development. The majority of winter-run steelhead populations in this DPS continue to persist at low abundances (NWFSC 2015b). Hatchery interactions remain a concern in select basins, but the overall situation is somewhat improved compared to prior reviews. Summer-run steelhead DIPs were similarly stable, but at low abundance levels. Habitat degradation continues to be a concern for most populations. Even with

modest improvements in the status of several winter-run populations, none of the populations appear to be at fully viable status, and similarly none of the MPGs meet the criteria for viability. The DPS therefore continues to be at moderate risk (NWFSC 2015b).

Critical Habitat

Critical habitat was designated for the Lower Columbia River steelhead on September 2, 2005(*Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California*). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Lower Columbia River steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

Critical habitat is affected by reduced quality of rearing and juvenile migration PBFs within the lower portion and alluvial valleys of many watersheds; contaminants from agriculture affect both water quality and food production in these reaches of tributaries and in the mainstem Columbia River. Several dams affect adult migration PBFs by obstructing the migration corridor. Watersheds which consist of a large proportion of federal lands such as is the case with the Sandy River watershed, have relatively healthy riparian corridors that support attributes of rearing PBFs such as cover, forage, and suitable water quality.

Recovery Goals

The Lower Columbia River DPS of steelhead are included in the Lower Columbia River recovery plan (NMFS 2013a). For this DPS, threats in all categories must be reduced, but the most crucial elements are protecting favorable tributary habitat and restoring habitat in the Upper Cowlitz, Cispus, North Fork Toutle, Kalama and Sandy subbasins (for winter steelhead), and the East Fork Lewis, and Hood, subbasins (for summer steelhead). Protection and improvement is also need among the South Fork Toutle and Clackamas winter steelhead populations.

6.2.30 Steelhead – Middle Columbia River DPS

The Middle Columbia River DPS of steelhead includes naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Columbia River and its tributaries upstream of the Wind and Hood Rivers (exclusive) to and including the Yakima River and excludes such fish originating from the Snake River Basin (Figure 70). Further, this DPS includes steelhead from seven artificial propagation programs.

A physical description of steelhead is presented in Section 6.1. On March 25, 1999 NMFS listed the Middle Columbia River (MCR) DPS of steelhead as threatened (64 FR 14517) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

Middle Columbia River steelhead populations are mostly of the summer-run type. Adult steelhead enter fresh water from June through August. The only exceptions are populations of inland winter-run steelhead which occur in the Klickitat River and Fifteenmile Creek (Busby et al. 1996). The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000). The eggs hatch in three to four weeks at 50°F to 59°F and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

The majority of juveniles smolt and out-migrate as two-year olds. Most of the rivers in this region produce about equal or higher numbers of adults having spent one year in the ocean as adults having spent two years. However, summer-run steelhead in Klickitat River have a life cycle more like LCR steelhead whereby the majority of returning adults have spent two years in the ocean (Busby et al. 1996). Adults may hold in the river up to a year before spawning. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002).

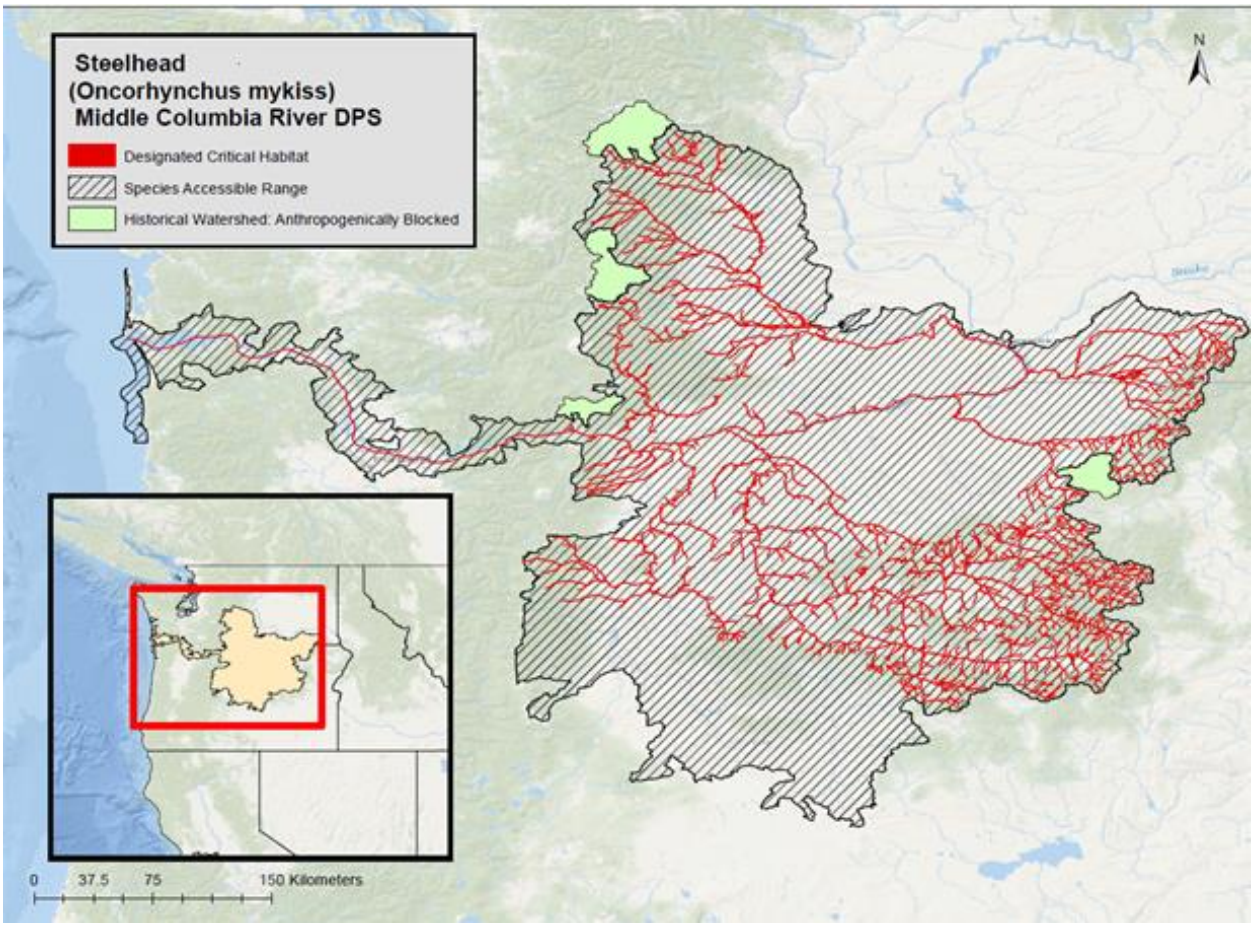


Figure 70. Geographic range and designated critical habitat of Middle Columbia River steelhead.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to Middle Columbia River steelhead.

Historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby et al. 1996). The five-year average (geometric mean) return of natural Middle Columbia River steelhead for 1997 to 2001 was up from basin estimates of previous years. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good et al. 2005). The five-year average for these basins is 298 and 1,492 fish, respectively (Good et al. 2005). Current abundance estimates for the Middle Columbia River DPS of steelhead are presented in Table 92 below.

Table 92. Current Abundance Estimates for the Middle Columbia River DPS of Steelhead (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	5,052
Natural	Juvenile	407,697
Listed Hatchery Adipose Clipped	Adult	448
Listed Hatchery Adipose Clipped	Juvenile	444,973
Listed Hatchery Intact Adipose	Adult	112
Listed Hatchery Intact Adipose	Juvenile	110,469

There have been improvements in the viability ratings for some of the component populations, but the Middle Columbia River Steelhead DPS is not currently meeting the viability criteria described in the Mid-Columbia Steelhead Recovery Plan.

The ICTRT identified 17 extant populations in this DPS (ICTRT 2003; McClure et al. 2005). The populations fall into four MPGs: Cascade eastern slope tributaries (five extant and two extirpated populations), the, the John Day River (five extant populations), the Walla Walla and Umatilla rivers (three extant and one extirpated populations), and the Yakima River (four extant populations.

Status

Within the Middle Columbia River DPS of steelhead, the ICTRT identified 16 extant populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek) (ICTRT 2003). There are two extinct populations in the Cascades Eastern Slope major population group: the White Salmon River and the Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on geographical proximity, topography, distance, ecological similarities or differences. Using criteria for abundance and productivity, the ICTRT modeled a gaps analysis for each of the four MPGs in this DPS under three different ocean conditions and a base hydro condition (most recent 20-year survival rate). The results showed that none of the MPGs would be able to achieve a 5 percent or less risk of extinction over 100 years without recovery actions. It is important to consider that significant gaps in factors affecting spatial structure and diversity also contribute to the risk of extinction for these fish.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, and freshwater migration corridors. The PBFs that characterize these sites include water quality and quantity, natural

cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Middle Columbia River steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The current condition of critical habitat designated for the Middle Columbia River steelhead is moderately degraded. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. Reduced quality of the rearing PBFs has diminished its contribution to the conservation value necessary for the recovery of the species. Several dams affect adult migration PBFs by obstructing the migration corridor.

Recovery Goals

See the 2009 recovery plan for the Middle Columbia River steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2009b).

6.2.31 Steelhead – Northern California DPS

The Northern California DPS of steelhead includes naturally spawned steelhead originating below natural and manmade impassable barriers in California coastal river basins from Redwood Creek to and including the Gualala River (Figure 71).

A physical description of steelhead is presented in Section 6.1. On June 7, 2000 NMFS listed the Northern California DPS of steelhead as threatened (65 FR 36074) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

The Northern California DPS of steelhead includes both winter- and summer –run steelhead. In the Mad and Eel Rivers, immature steelhead may return to fresh water as "half-pounders" after spending only two to four months in the ocean. Generally, a half-pounder will overwinter in fresh water and return to the ocean in the following spring.

Juvenile out-migration appears more closely associated with size than age but generally, throughout their range in California, juveniles spend two years in fresh water (Busby et al. 1996). Smolts range from 14-21 cm in length. Juvenile steelhead may migrate to rear in lagoons throughout the year with a peak in the late spring/early summer and in the late fall/early winter period (Shapovalov and Taft 1954; Zedonis 1992).

Steelhead spend anywhere from one to five years in salt water, however, two to three years are most common (Busby et al. 1996). Ocean distribution is not well known but coded wire tag recoveries indicate that most NC steelhead migrate north and south along the continental shelf (Barnhart 1986).

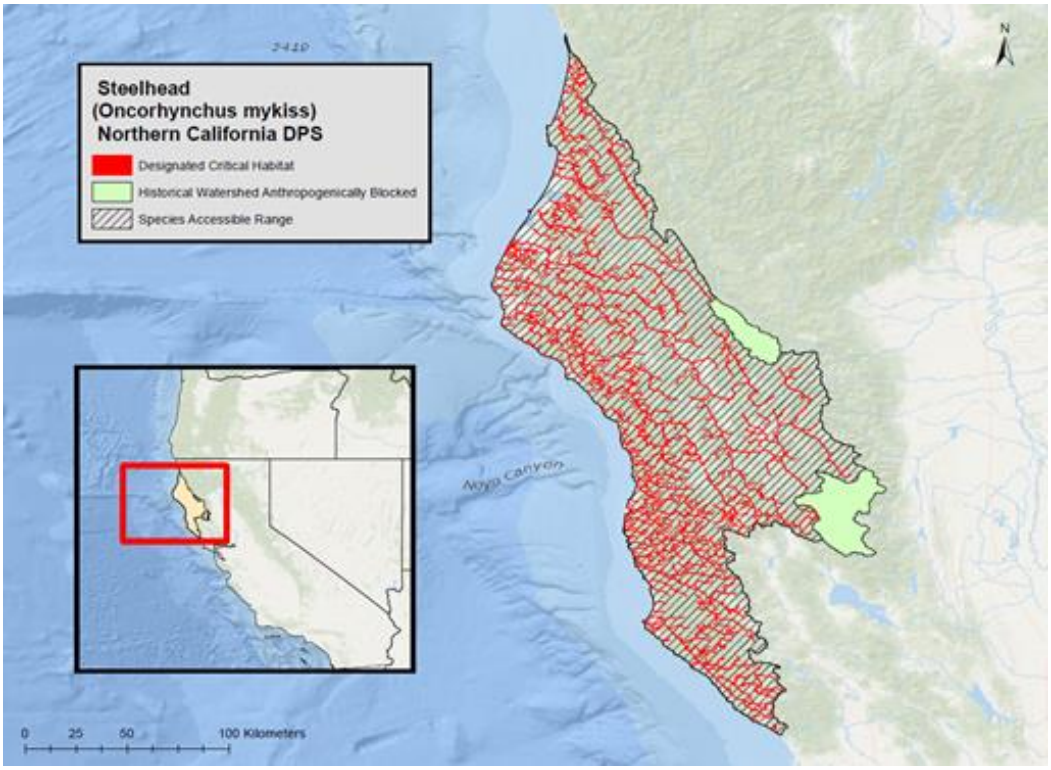


Figure 71. Geographic range and designated critical habitat of Northern California DPS steelhead.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Northern California DPS of steelhead.

Most populations for which there are population estimates available remain well below viability targets; however, the short-term increases observed for many populations, despite the occurrence of a prolonged drought in northern California, suggests this DPS is not at immediate risk of extinction. Current abundance estimates for the Northern California DPS of steelhead are presented in Table 93 below.

Table 93. Current Abundance Estimates for the Northern California DPS of Steelhead (NMFS 2019d).

Production	Life Stage	Abundance
Natural	Adult	7,221
Natural	Juvenile	821,389

Overall, the available data for winter-run populations— predominately in the North Coastal, North-Central Coastal, and Central Coastal strata— indicate that all populations are well below viability targets, most being between five percent and 13 percent of these goals. For the two Mendocino Coast populations with the longest time series, Pudding Creek and Noyo River, the 13-year trends have been negative and neutral, respectively (Spence 2016). However, the short-term (six-year) trend has been generally positive for all independent populations in the North-Central Coastal and Central Coastal strata, including the Noyo River and Pudding Creek (Spence 2016). Data from Van Arsdale Station likewise suggests that, although the long-term trend has been negative, run sizes of natural-origin steelhead have stabilized or are increasing (Spence 2016). Thus, we have no strong evidence to indicate conditions for winter-run populations in the DPS have worsened appreciably since the last status review (NMFS 2016c). Summer-run populations continue to be of significant concern because of how few populations currently exist. The Middle Fork Eel River population has remained remarkably stable for nearly five decades and is closer to its viability target than any other population in the DPS (Spence 2016). Although the time series is short, the Van Duzen River appears to be supporting a population numbering in the low hundreds. However, the Redwood Creek and Mattole River populations appear small, and little is known about other populations including the Mad River and other tributaries of the Eel River (i.e., Larabee Creek, North Fork Eel, and South Fork Eel).

Artificial propagation was identified as negatively affecting wild stocks of salmonids through interactions with non-native fish, introductions of disease, genetic changes, competition for space and food resources, straying and mating with native populations, loss of local genetic adaptations, mortality associated with capture for broodstock and palliating the destruction of habitat and concealing problems facing wild stocks.

Status

The available data for winter-run populations— predominately in the North Coastal, North-Central Coastal, and Central Coastal strata— indicate that all populations are well below viability targets, most being between five percent and 13 percent of these goals. For the two Mendocino Coast populations with the longest time series, Pudding Creek and Noyo River, the 13-year trends have been negative and neutral, respectively (Spence 2016). However, the short-term (six-year) trend has been generally positive for all independent populations in the North-Central Coastal and Central Coastal strata, including the Noyo River and Pudding Creek (Spence 2016). Data from Van Arsdale Station likewise suggests that, although the long-term trend has been negative, run sizes of natural-origin steelhead have stabilized or are increasing (Spence 2016). Summer-run populations continue to be of significant concern because of how few populations currently exist. The Middle Fork Eel River population has remained remarkably stable for nearly five decades and is closer to its viability target than any other population in the DPS (Spence 2016). Although the time series is short, the Van Duzen River appears to be supporting a population numbering in the low hundreds. However, the Redwood Creek and Mattole River populations appear small, and little is known about other populations including the

Mad River and other tributaries of the Eel River (i.e., Larabee Creek, North Fork Eel, and South Fork Eel). Most populations for which there are population estimates available remain well below viability targets; however, the short-term increases observed for many populations, despite the occurrence of a prolonged drought in northern California, suggests this DPS is not at an immediate risk of extinction.

Critical Habitat

NMFS designated critical habitat for Northern California DPS steelhead on September 2, 2005(*Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California*). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Northern California DPS steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The current condition of critical habitat designated for the Northern California DPS steelhead is moderately degraded. Nevertheless, it does provide some conservation value necessary for species recovery. Within portions of its range, especially the interior Eel River, rearing PBF quality is affected by elevated temperatures by removal of riparian vegetation. Spawning PBF attributes such as the quality of substrate supporting spawning, incubation, and larval development have been generally degraded throughout designated critical habitat by silt and sediment fines in the spawning gravel. Bridges and culverts further restrict access to tributaries in many watersheds, especially in watersheds with forest road construction, thereby reducing the function of adult migration PBFs.

Recovery Goals

See the 2016 recovery plan for the Northern California steelhead DPS for complete down-listing/delisting criteria for recovery goals for the DPS (NMFS 2016g).

6.2.32 Steelhead – Puget Sound DPS

This distinct population segment, or DPS, includes naturally spawned anadromous *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Georgia Strait (Figure 72). Also, steelhead from six artificial propagation programs.

A physical description of steelhead is presented in Section 6.1. On May 11, 2007 NMFS listed the Puget Sound DPS of steelhead as threatened (72 FR 26722).

Life History

The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS 2005d). Spawning occurs from January to mid-June, with peak spawning occurring from mid-April through May. Prior to spawning, maturing adults hold in pools or in side channels to avoid high winter flows. Less information exists for summer-run steelhead as their smaller run size and higher altitude headwater holding areas have not been conducive for monitoring. Based on information from four streams, adult run time occur from mid-April to October with a higher concentration from July through September (NMFS 2005d).

The majority of juveniles reside in the river system for two years with a minority migrating to the ocean as one or three-year olds. Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from one to three years in the ocean (Busby et al. 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to Puget Sound steelhead.

Abundance of adult steelhead returning to nearly all Puget Sound rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s. Inspection of geometric means of total spawner abundance from 2010 to 2014 indicates that 9 of 20 populations evaluated had geometric mean abundances fewer than 250 adults and 12 of 20 had fewer than 500 adults.

Smoothed trends in abundance indicate modest increases since 2009 for 13 of the 22 DIPs. Between the two most recent five-year periods (2005-2009 and 2010-2014), the geometric mean of estimated abundance increased by an average of 5.4 percent. For seven populations in the Northern Cascades MPG, the increase was 3 percent; for five populations in the Central & South Puget Sound MPG, the increase was 10 percent; and for six populations in the Hood Canal & Strait of Juan de Fuca MPG, the increase was 4.5 percent. However, several of these upward trends are not statistically different from neutral, and most populations remain small. Long-term (15-year) trends in natural spawners are predominantly negative (NWFSC 2015a). Current abundance estimates for the Puget Sound DPS of steelhead are presented in Table 94 and Table 95 below.

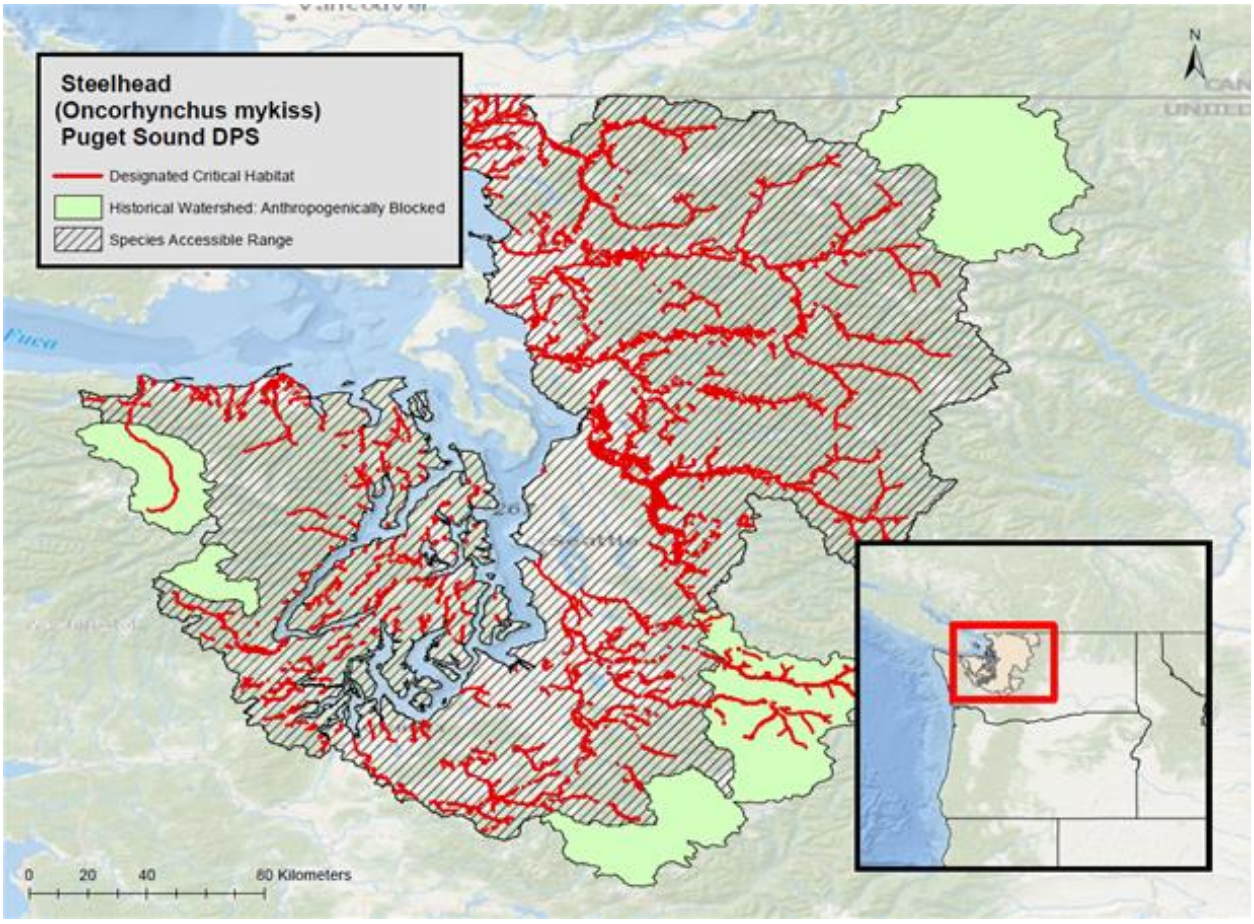


Figure 72. Geographic range and designated critical habitat of Puget Sound DPS steelhead.

Table 94. Expected 2019 Puget Sound steelhead listed hatchery releases (NMFS 2020c).

Subbasin	Artificial propagation program	Brood year	Run Timing	Clipped Adipose Fin	Intact Adipose Fin
Dungeness/Elwha	Dungeness	2018	Winter	10,000	-
	Hurd Creek	2018	Winter	-	34,500
Duwamish/Green	Flaming Geyser	2018	Winter	-	15,000
	Icy Creek	2018	Summer	50,000	-
			Winter	-	28,000
	Soos Creek	2018	Summer	50,000	-
Puyallup	White River	2018	Winter	-	35,000
Total Annual Release Number				**110,000**	**112,500**

Table 95. Abundance of PS steelhead spawner escapements (natural-origin and hatchery-production combined) from 2012-2016 (NMFS 2020c).

Demographically Independent Populations	Spawners	Expected Number of Outmigrants[b]
Central and South Puget Sound MPG		
Cedar River	3	391
Green River	977	111,179
Nisqually River	759	86,323
N. Lake WA/Lake Sammamish	-	-
Puyallup/Carbon River	603	68,646
White River	629	71,638
Hood Canal and Strait of Juan de Fuca MPG		
Dungeness River[c]	26	2,984
East Hood Canal Tribs.	89	10,120
Elwha River	878	99,954
Sequim/Discovery Bay Tribs.	19	2,186
Skokomish River	862	98,066
South Hood Canal Tribs.	73	8,304
Strait of Juan de Fuca Tribs.	173	19,697
West Hood Canal Tribs.	122	13,858
North Cascades MPG		
Nooksack River	1,790	203,631
Pilchuck River	868	98,709
Samish River/ Bellingham Bay Tribs.	977	111,167
Skagit River	8,038	914,353
Snohomish/Skykomish Rivers	1,053	119,762
Snoqualmie River	824	93,772
Stillaguamish River	476	54,170
Tolt River	70	7,988
TOTAL	**19,313**	**2,196,901**

Only two hatchery stocks genetically represent native local populations (Hamma and Green River natural winter-run). The remaining programs, which account for the vast preponderance of production, are either out-of-DPS derived stocks or were within-DPS stocks that have diverged substantially from local populations. The WDFW estimated that 31 of the 53 stocks were of native origin and predominantly natural production (Washington Department of Fish and Wildlife (WDFW) 1993).

Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run

steelhead in the rest of the DPS. The Elwha River run, however, is descended from introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s.

Status

For all but a few putative demographically independent populations of steelhead in Puget Sound, estimates of mean population growth rates obtained from observed spawner or redd counts are declining—typically three to 10 percent annually. Extinction risk within 100 years for most populations in the DPS is estimated to be moderate to high, especially for draft populations in the putative South Sound and Olympic major population groups. Collectively, these analyses indicate that steelhead in the Puget Sound DPS remain at risk of extinction throughout all or a significant portion of their range in the foreseeable future, but are not currently in danger of imminent extinction. The Biological Review for the latest 5-Year Review of the Puget Sound DPS of steelhead identified degradation and fragmentation of freshwater habitat, with consequent effects on connectivity, as the primary limiting factors and threats facing the Puget Sound steelhead DPS. The status of the listed Puget Sound steelhead DPS has not changed substantially since the 2007 listing. Most populations within the DPS are showing continued downward trends in estimated abundance, a few sharply so. The limited available information indicates that this DPS remains at a moderate risk of extinction.

Critical Habitat

NMFS designated critical habitat for Puget Sound steelhead on February 2, 2016 (81 FR 9251). The specific areas designated for Puget Sound steelhead include approximately 2,031 stream miles (3,269 km) within the geographical area presently occupied by this DPS (Figure 72). Designated critical habitat for the Puget Sound steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. There are 66 watersheds within the range of the Puget Sound steelhead DPS. Nine watersheds received a low conservation value rating, 16 received a medium rating, and 41 received a high rating.

Recovery Goals

NMFS published a final recovery plan for the Puget Sound ESU of steelhead on December 20, 2019 (NMFS 2019f). The recovery plan's primary goals are as follows:

- The Puget Sound steelhead DPS achieves biological viability and the ecosystems upon which the DPS depends are conserved such that it is sustainable and persistent and no longer needs federal protection under the ESA; and

- The five listing factors from the ESA, section 4 (a)(1) are addressed. The five listing factors from the ESA, section 4(a)(1), include:
    - The present or threatened destruction, modification, or curtailment of the species' habitat or range;
    - Overutilization for commercial, recreational, scientific, or educational purposes;
    - Disease or predation;
    - Inadequacy of existing regulatory mechanisms; and
    - Other natural or human-made factors affecting the species' continued existence.

Delisting criteria for the Puget Sound DPS of steelhead are detailed in NMFS (2019f).

6.2.33 Steelhead – Snake River Basin DPS

The Snake River Basin DPS of steelhead includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Snake River Basin (Figure 73), and also steelhead from six artificial propagation programs.

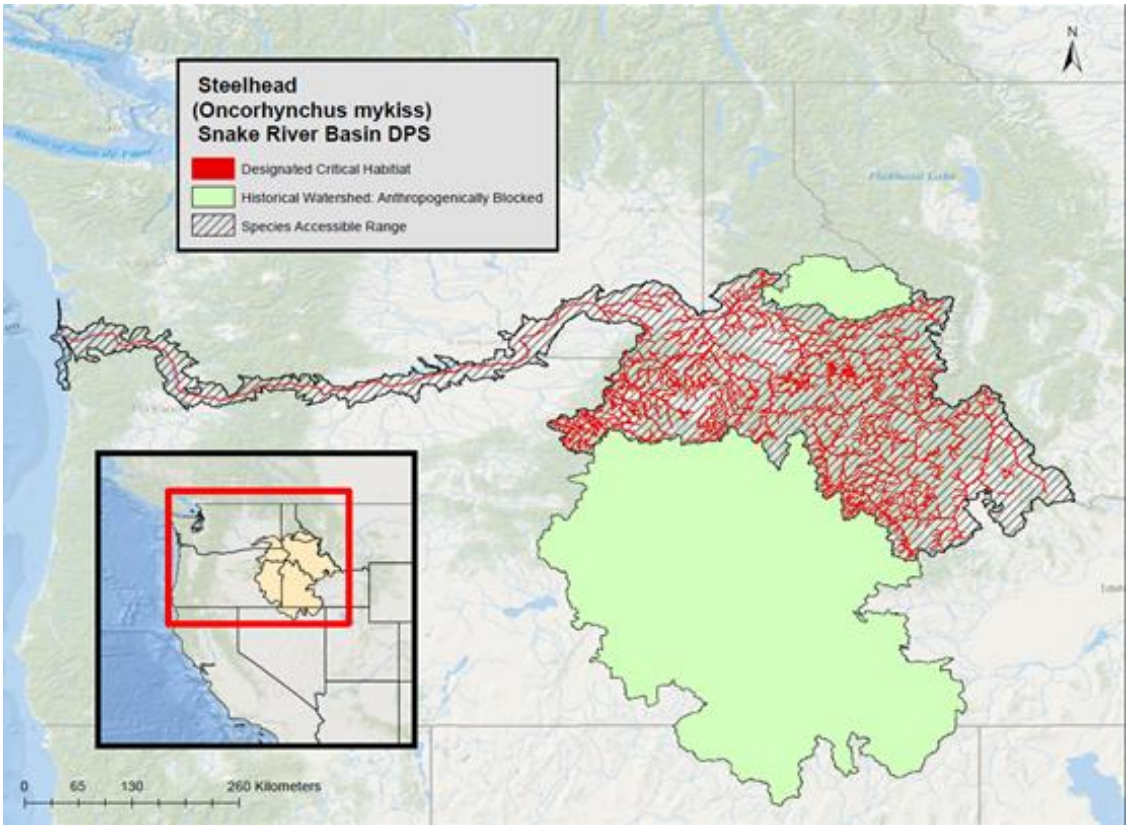


Figure 73. Geographic range and designated critical habitat of Snake River Basin steelhead.

A physical description of steelhead is presented in Section 6.1. On August 18, 1997 NMFS listed the Snake River Basin DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

Snake River Basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, Snake River Basin steelhead spawn the following spring (March to May). Managers recognize two life history patterns within this DPS primarily based on ocean age and adult size upon return: A-run or B-run. A-run steelhead are typically smaller, have a shorter freshwater and ocean residence (generally one year in the ocean), and begin their up-river migration earlier in the year. B-run steelhead are larger, spend more time in fresh water and the ocean (generally two years in ocean), and appear to start their upstream migration later in the year. Snake River Basin steelhead usually smolt after two or three years.

The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000). The eggs hatch in three to four weeks at 50°F to 59°F, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

The majority of juveniles smolt and out-migrate as two-year olds. Adults may hold in the river up to a year before spawning. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to Snake River Basin steelhead.

There is uncertainty for wild populations of Snake River Basin DPS steelhead given limited data for adult spawners in individual populations. Regarding population growth rate, there are mixed long- and short-term trends in abundance and productivity. Overall, the abundances remain well below interim recovery criteria. Current abundance estimates for the Snake River Basin DPS of steelhead are presented in Table 96 below.

Table 96. Current Abundance Estimates for the Snake River Basin DPS of Steelhead (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	10,547
Natural	Juvenile	798,341
Listed Hatchery Adipose Clipped	Adult	79,510
Listed Hatchery Adipose Clipped	Juvenile	3,300,152
Listed Hatchery Intact Adipose	Adult	16,137
Listed Hatchery Intact Adipose	Juvenile	705,490

Status

Four out of the five MPGs are not meeting the specific objectives in the draft recovery plan being written by NMFS based on the updated status information available for this review, and the status of many individual populations remains uncertain (NWFSC 2015b). The Grande Ronde MPG is tentatively rated as viable; more specific data on spawning abundance and the relative contribution of hatchery spawners for the Lower Grande Ronde and Wallowa populations would improve future assessments. A great deal of uncertainty still remains regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites within individual populations.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Snake River Basin steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The current condition of critical habitat designated for Snake River Basin steelhead is moderately degraded. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. These factors have substantially reduced the rearing PBFs contribution to the conservation value

necessary for species recovery. Several dams affect adult migration PBFs by obstructing the migration corridor.

Recovery Goals

NMFS published a final recovery plan for the Snake River Basin DPS of steelhead on November 30, 2017 (NMFS 2017f). The ESA recovery goal for Snake River Basin steelhead is that: The ecosystems upon which the steelhead depend are conserved such that the DPS is self-sustaining in the wild and no longer need ESA protection. More information on the Snake River Basin DPS' recovery goals and delisting criteria are found in NMFS (2017f).

6.2.34 Steelhead South-Central California DPS

The South-Central California Coast DPS of steelhead includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Pajaro River to (but not including) the Santa Maria River. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins overlapping within the range of this DPS include the inland basins of the Pajaro River and the Salinas River.

A physical description of steelhead is presented in Section 6.2.27. On August 18, 1997 NMFS listed the South-Central California Coast DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 5248).

Life History

Only winter steelhead are found in the South-Central California Coast DPS of steelhead. Most spawning takes place from January through April. The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30 degrees Fahrenheit to 52 degrees Fahrenheit (CDFW 2000). The eggs hatch in three to four weeks at 50 degrees Fahrenheit to 59 degrees Fahrenheit, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

Steelhead typically migrate to marine waters after spending two years in fresh water. They reside in marine waters for typically two or three years prior to returning to their natal stream to spawn as four- or five-year olds. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002). There is limited life history information for steelhead in this DPS.

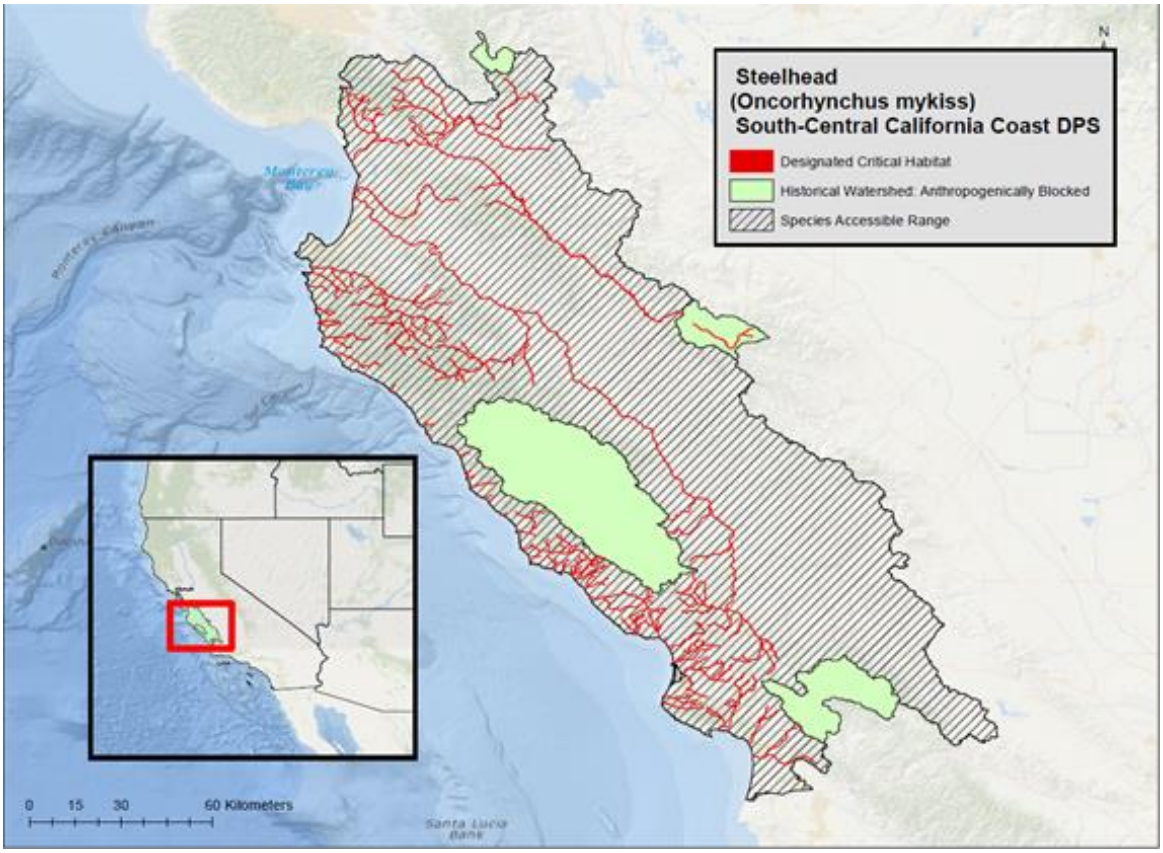


Figure 74. Geographic range and designated critical habitat of South-Central California Coast steelhead.

Population Dynamics

The data summarized in the most recent status review indicate small (generally <10 fish) but surprisingly persistent annual runs of anadromous *O. mykiss* are currently being monitored across a limited but diverse set of basins within the range of this DPS, but interrupted in years when the mouth of the coastal estuaries fail to open to the ocean due to low flows (Williams et al. 2011c). Current abundance estimates for the South-Central California Coast DPS of steelhead are presented in Table 97 below.

Table 97. Current Abundance Estimates for the South-Central California Coast DPS of Steelhead (NMFS 2019d).

Production	Life Stage	Abundance
Natural	Adult	695
Natural	Juvenile	79,057

Status

Following the dramatic rise in South-Central California's human population after World War II and the associated land and water development within coastal drainages (particularly major dams and water diversions), steelhead abundance rapidly declined, leading to the extirpation of populations in many watersheds and leaving only sporadic and remnant populations in the remaining, more highly modified watersheds such as the Salinas River and Arroyo Grande Creek watersheds (NMFS 2013e). A substantial portion of the upper watersheds, which contain the majority of historical spawning and rearing habitats for anadromous O. mykiss, remain intact (though inaccessible to anadromous fish) and protected from intensive development as a result of their inclusion in the Los Padres National Forest (NMFS 2013e).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005(*Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California*). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the South-Central California Coast steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

Migration and rearing PBFs are degraded throughout critical habitat by elevated stream temperatures and contaminants from urban and agricultural areas. Estuarine PBFs are impacted by most estuaries being breached, removal of structures, and contaminants.

Recovery Goals

See the 2013 recovery plan for the South-Central California Coast steelhead DPS (NMFS 2013e) for complete down-listing/delisting criteria for recovery goals for the species.

6.2.35 Steelhead – Upper Columbia River DPS

The Upper Columbia River DPS of steelhead includes naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Columbia River and its tributaries upstream of the Yakima River to the U.S.-Canada border. Also, the Upper Columbia River DPS includes steelhead from six artificial propagation programs.

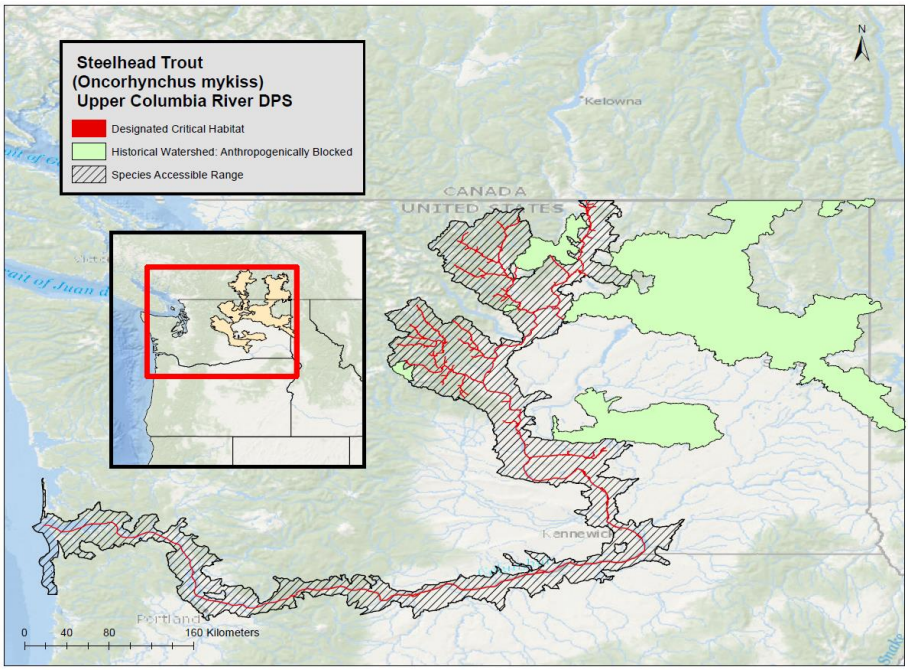


Figure 75. Geographic range and designated critical habitat of upper Columbia River steelhead.

A physical description of steelhead is presented in Section 6.1. On August 18, 1997 NMFS listed the Upper Columbia River DPS of steelhead as endangered (62 FR 43937) and reaffirmed the DPS's status as endangered on January 5, 2006 (71 FR 834).

Life History

All Upper Columbia River steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinter in mainstem reservoirs, passing over upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend one to seven years rearing in fresh water before migrating to sea. Smolt out migrations are predominantly year class two and three (juveniles), although some of the oldest smolts are reported from this DPS at seven years. Most adult steelhead return to fresh water after one or two years at sea.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to Upper Columbia River steelhead.

The most recent estimates of natural-origin spawner abundance for each of the four populations in the Upper Columbia River DPS of steelhead show fairly consistent patterns throughout the years. None of the populations have reached their recovery goal numbers during any of the years

(500 for the Entiat, 2,300 for the Methow, 2,300 for the Okanogan, and 3,000 for Wenatchee). Current abundance estimates for the Upper Columbia River DPS of steelhead are presented in Table 98 below.

Table 98. Current Abundance Estimates for the Upper Columbia River DPS of Steelhead (NMFS 2020c).

Production	Life Stage	Abundance
Natural	Adult	3,988
Natural	Juvenile	169,120
Listed Hatchery Adipose Clipped	Juvenile	662,848
Listed Hatchery Intact Adipose	Adult	2,403
Listed Hatchery Intact Adipose	Juvenile	144,067

Upper Columbia River steelhead populations have increased relative to the low levels observed in the 1990s, but natural origin abundance and productivity remain well below viability thresholds for three out of the four populations. In spite of recent increases, natural origin abundance and productivity remain well below viability thresholds for three out of the four populations, and the Okanogan River natural-origin spawner abundance estimates specifically are well below the recovery goal for that population. Three of four extant natural populations are considered to be at high risk of extinction and one at moderate risk.

All populations are at high risk for diversity, largely driven by chronic high levels of hatchery spawners within natural spawning areas and lack of genetic diversity among the populations.

The Upper Columbia River steelhead DPS is composed of three MPGs, two of which are isolated by dams. With the exception of the Okanogan population, the Upper Columbia River populations were rated as low risk for spatial structure.

Status

Current estimates of natural origin spawner abundance increased relative to the levels observed in the prior review for all three extant populations, and productivities were higher for the Wenatchee and Entiat and unchanged for the Methow (NWFSC 2015b). However abundance and productivity remained well below the viable thresholds called for in the Upper Columbia Recovery Plan for all three populations. Short-term patterns in those indicators appear to be largely driven by year-to year fluctuations in survival rates in areas outside of these watersheds. All three populations continued to be rated at low risk for spatial structure but at high risk for diversity criteria. Although the status of the ESU is improved relative to measures available at the time of listing, all three populations remain at high risk (NWFSC 2015b).

Critical Habitat

Critical habitat was designated for the Upper Columbia River DPS of steelhead on September 2, 2005 (70 FR 52630). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Upper Columbia River steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The current condition of critical habitat designated for the upper Columbia River steelhead is moderately degraded. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PBFs by obstructing the migration corridor.

Recovery Goals

See the 2007 recovery plan for the Upper Columbia River steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2007b).

6.2.36 Steelhead – Upper Willamette River DPS

This DPS includes naturally spawned anadromous winter-run *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from the Willamette River and its tributaries upstream of Willamette Falls to and including the Calapooia River (Figure 76).

A physical description of steelhead is presented in Section 6.1. On March 25, 1999 NMFS listed the Upper Willamette River DPS of steelhead as threatened (64 FR 14517) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

Life History

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al. 1985). Upper Willamette River steelhead do not ascend to their spawning areas until late March or April, which is late compared to other West Coast winter steelhead. Spawning occurs from April to June 1. The unusual run timing may be an adaptation for ascending the Willamette Falls, which may have facilitated reproductive isolation of the stock. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell et al. 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. As with other coastal steelhead, the majority of juvenile smolts outmigrate after two years; adults return to their natal rivers to spawn after spending two years in the ocean. Repeat

spawners are predominantly female and generally account for less than 10 percent of the total run size (Busby et al. 1996).

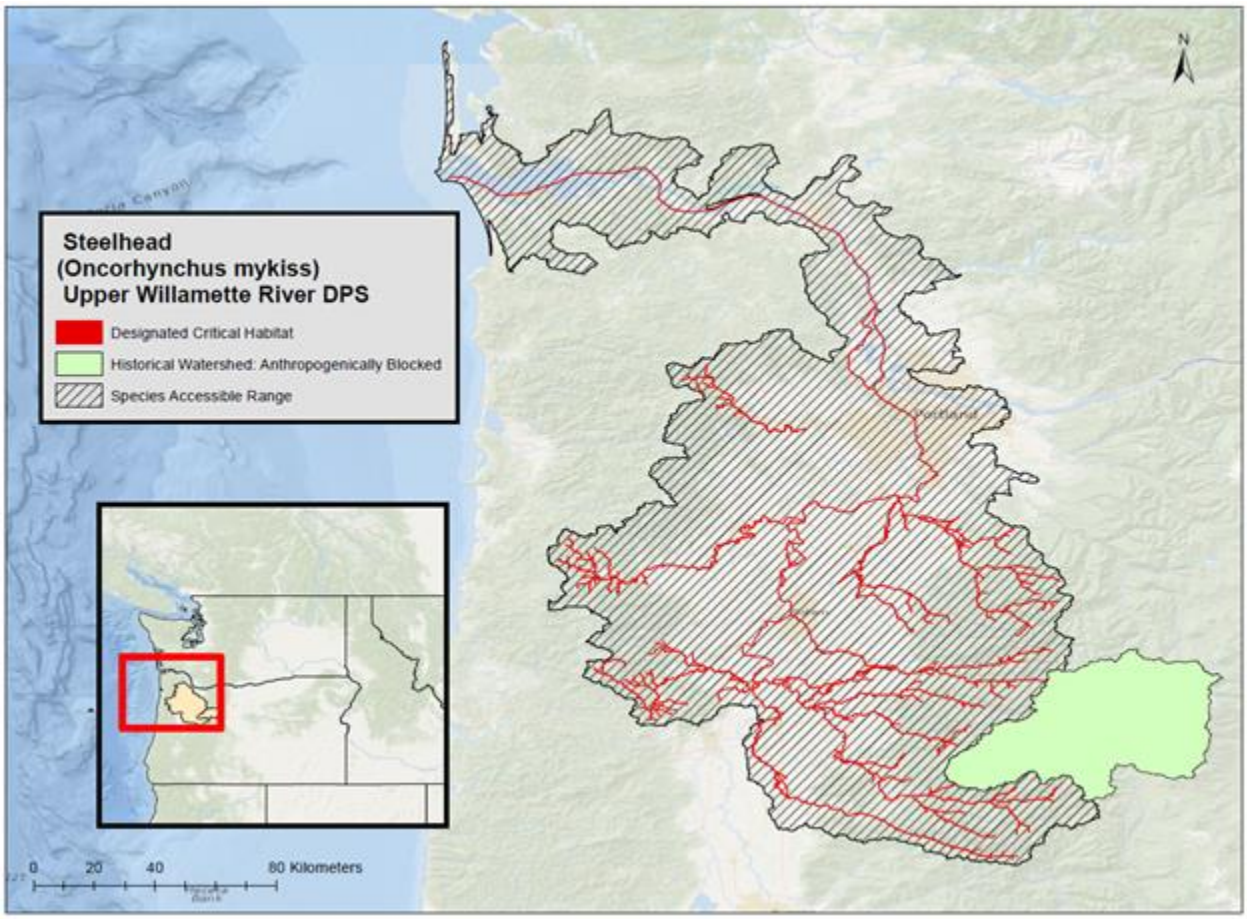


Figure 76. Geographic range and designated critical habitat of upper Willamette River steelhead.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to upper Willamette River steelhead.

For the Upper Willamette steelhead DPS, the declines in abundance noted during the previous status review continued through 2010 to 2015, and accessibility to historical spawning habitat remains limited, especially in the North Santiam River. Although the recent magnitude of these declines is relatively moderate, NMFS Northwest Fisheries Science Center (NWFSC 2015b) notes that continued declines would be a cause for concern.

Recent estimates of escapement in the Molalla River indicate abundance is stable but at a depressed level, and the lack of migration barriers indicates this limitation is likely due to habitat degradation (NWFSC 2015b). In the North Santiam, radio-tagging studies and counts at Bennett Dam between 2010 and 2014 estimate the average abundance of returning winter-run adults is following a long-term negative trend (NWFSC 2015b). In the South Santiam live counts at Foster Dam indicate a negative trend in abundance from 2010-2014, and redd survey data indicate consistent low numbers of spawners in tributaries (NWFSC 2015b). Radio-tagging studies in the Calapooia from 2012-2014 suggest that abundances have been depressed but fairly stable, however long-term trends in redd counts conducted since 1985 are generally negative (NWFSC 2015b). Current abundance estimates for the Upper Willamette River DPS of steelhead are presented in Table 99 below.

Table 99. Current Abundance Estimates for the Upper Willamette River DPS of Steelhead (NMFS 2019c).

Production	Life Stage	Abundance
Natural	Adult	2,912
Natural	Juvenile	143,898

Genetic analysis suggests that there is some level introgression among native late-winter steelhead and summer-run steelhead (Van Doornik et al. 2015), and up to approximately 10 percent of the juvenile steelhead at Willamette Falls and in the Santiam Basin may be hybrids (Johnson et al. 2013). While winter-run steelhead have largely maintained their genetic distinctiveness over time (Van Doornik et al. 2015), there are still concerns that hybridization will decrease the overall productivity of the native population. In addition, releases of large numbers of hatchery-origin summer steelhead may temporarily exceed rearing capacities and displace winter-run juvenile steelhead (NWFSC 2015b).

There are four DIPs within the Upper Willamette River DPS of steelhead. Historical observations, hatchery records, and genetics suggest that the presence of Upper Willamette River DPS steelhead in many tributaries on the west side of the upper basin is the result of recent introductions. Nevertheless, the Willamette/Lower Columbia Technical Recovery Team recognized that although west side Upper Willamette River DPS steelhead does not represent a historical population, those tributaries may provide juvenile rearing habitat or may be temporarily (for one or more generations) colonized during periods of high abundance. Hatchery summer-run steelhead that are released in the subbasins are from an out-of-basin stock, and are not part of the DPS, nor are stocked summer steelhead that have become established in the McKenzie River (ODFW and NMFS 2011).

Status

Four basins on the east side of the Willamette River historically supported independent populations for the Upper Willamette River DPS steelhead, all of which remain extant. Data indicate that currently the two largest populations within the DPS are the Santiam River populations. Mean spawner abundance in both the North and South Santiam River is about 2,100 native winter-run steelhead. However, about 30 percent of all habitat has been lost due to human activities (McElhany et al. 2007). The North Santiam population has been substantially affected by the loss of access to the upper North Santiam basin. The South Santiam subbasin has lost habitat behind non-passable dams in the Quartzville Creek watershed. Notwithstanding the lost spawning habitat, the DPS continues to be spatially well distributed, occupying each of the four major subbasins.

Overall, the declines in abundance noted during the previous review continued through the period 2010-2015 (NWFSC 2015b). There is considerable uncertainty in many of the abundance estimates, except for perhaps the tributary dam counts. Radio-tagging studies suggest that a considerable proportion of winter-run steelhead ascending Willamette Falls do not enter the DIPs that constitute this DPS; these fish may be nonnative early winter-run steelhead that appear to have colonized the western tributaries, misidentified summer-run steelhead, or late winter-run steelhead that have colonized tributaries not historically part of the DPS.

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005(*Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California*). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Willamette River steelhead does not overlap spatially with the NWTT action area and, therefore, will not be analyzed further in this opinion.

The current condition of critical habitat designated for the upper Willamette River steelhead is degraded and provides a reduced the conservation value necessary for species recovery. Critical habitat is affected by reduced quality of juvenile rearing and migration PBFs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PBFs by obstructing the migration corridor.

Recovery Goals

See the 2011 recovery plan for the Upper Willamette River steelhead DPS (NMFS 2011f) for complete down-listing/delisting criteria for recovery goals for the species.

6.2.37 Bocaccio – Puget Sound/Georgia Basin DPS

The bocaccio is a long-lived species of rockfish, occupying the eastern Pacific Ocean in waters from California to Alaska. Puget Sound/Georgia Basin DPS bocaccio are those that reside in Puget Sound/Georgia Basin (Figure 77).



Figure 77. Map identifying the range of the Puget Sound/Georgia Basin distinct population segment bocaccio.

Bocaccio are a large (up to three feet) Pacific rockfish, olive to burnt orange-brown, with a distinctively long jaw. The Puget Sound/Georgia Basin DPS was first listed as endangered by NMFS on April 28, 2010 (75 FR 22276). The listing was updated on January 23, 2017 (82 FR 7711), when NMFS amended the listing description to include fish residing within the Puget Sound/Georgia Basin rather than fish originating from the Puget Sound/Georgia Basin.

We used information available in the status review (NMFS 2016m) and recent scientific publications to summarize the life history, population dynamics, and status of the species, as follows.

Life History

Bocaccio larvae (<1.2 in) and pelagic juvenile bocaccio (1.2-3.6 in) have been found at varying depth ranges within the water column and are often associated with floating kelp mats, algae and seagrass (NMFS 2016m; NMFS 2017h). Once juvenile bocaccio reach 1 to 3.5 inches, they move into shallow nearshore waters, with rocky or cobble substrates, preferably with kelp (Love et al. 1991; Love et al. 2002b). Subadult and adult bocaccio (> 3.6 in) may be found in depths

ranging from 30-425 meters and are often associated with rocky habitat and complex structure (NMFS 2017h).

Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54 to 61 cm TL), they produce 20,000 to 2.3 million eggs annually, with the number increasing as females age and grow larger (Echeverria 1987; Hart 1973; Love et al. 2002b). However, either sex has been known to attain sexual maturity as small as 35 centimeters or three years of age. In recent years as populations have declined, average age at sexual maturity may have declined as well (Echeverria 1987; Hart 1973; Love et al. 2002b; MacCall 2002). Mating occurs between August and November, with larvae born between January and April (NMFS 2016m).

Upon birth, bocaccio larvae measure four to five mm in length. These larvae move into pelagic waters as juveniles when they are 1.5 to 3 cm and remain in oceanic waters from 3.5 to 5.5 months after birth (usually until early June), where they grow at ~0.5 to one mm per day (NMFS 2016m). However, growth can vary from year-to-year (Woodbury and Ralston 1991). Larval rockfish are extremely fragile and thus have high natural mortality rates. Their small size, relative inability to store food within their gut, and slow swimming speeds likely contribute to this high mortality rate by making them vulnerable to predators and starvation (NMFS 2014a). Poor larval survival in most years provides evidence that rockfish populations persist through what has been termed "the storage hypothesis" (Tolimieri and Levin 2005) where recruitment is generally poor because larval survival and settlement are dependent upon the vagaries of climate, abundance of predators, oceanic currents, and chance events (NMFS 2014a).

Once individuals are three to four centimeters in length, they return to nearshore waters, where they settle into bottom habitats. Females tend to grow faster than males, but fish may take five years to reach sexual maturity (MacCall 2003). Individuals continue to grow until they reach maximum sizes of 91 cm, or 9.6 kg, at an estimated maximum age of 50 years (Andrews et al. 2005; Eschmeyer et al. 1983; Halstead et al. 1990; Love et al. 2002b; Piner et al. 2006; Ralston and Ianelli 1998). Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates. Pelagic juveniles consume fish larvae, copepods, and krill, while older, nearshore juveniles and adults prey upon rockfishes, hake, sablefish, anchovies, lanternfish, and squid (Love et al. 2002b; Reilly et al. 1992)

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Puget Sound/Georgia Basin DPS bocaccio.

In 2013, the Washington Department of Fish and Wildlife published abundance estimates from a remotely operated vehicle survey conducted in 2008 in the San Juan Island area (NMFS 2016m; WDFW 2017). The survey produced an estimate of 4,606 (100 percent variance) based on a very small sample size (four fish observed along a single transect). No abundance estimate could be obtained in the 2010 ROV survey because this species was not encountered. A single bocaccio

encountered in the 2015 ROV survey produced a statistically invalid population estimate for that portion of the DPS lying south of the entrance to Admiralty Inlet and east of Deception Pass.

The rate of decline for rockfish (in general) in Puget Sound has been estimated at 3.1 to 3.8 percent annually for the period 1977 to 2014 (NMFS 2016m). Although there is a lack of long-term trend information on the Puget Sound/Georgia Basin DPS bocaccio abundance, among rockfish in Puget Sound bocaccio appear to have undergone a particularly sharp decline. This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climatic/oceanographic conditions (MacCall and He 2002).

Puget Sound/Georgia Basin bocaccio occupy the inland marine waters east of the Strait of Juan de Fuca and south of the northern Georgia Strait. Puget Sound/Georgia Basin DPS bocaccio are distinct from bocaccio elsewhere in its range, likely due to its inhabitance of a geographically isolated area. There is no genetic information available for bocaccio in Puget Sound/Georgia Basin (NMFS 2016m).

Status

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 1998a). Bocaccio are long-lived fishes, taking several years to reach sexual maturity and becoming more fecund with age (Dorn 2002). As harvesting targeted the largest individuals available, bocaccio have become less capable of recovering population numbers (Love et al. 1998b). At present, in the complete absence of directed or bycatch fishing pressure, it is estimated that bocaccio populations would have to have frequent good recruitment to restrain their present decline (Tolimieri and Levin 2005). In addition, bocaccio reproduction appears to be characterized by frequent recruitment failures, punctuated by occasional high success years (Love et al. 1998b; MacCall and He 2002). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes. Recruitment success appears to be linked to oceanographic/climatic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998b; MacCall 1996; Moser et al. 2000; Sakuma and Ralston 1995). Harvey *et al.* (2006) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery.

Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS of bocaccio, canary rockfish, and yelloweye rockfish was finalized in 2014 (79 FR 68041). The critical habitat designation was updated in 2017 when canary rockfish were delisted (82 FR 7711). Rockfish critical habitat is spread amongst five interconnected, biogeographic basins (San Juan/Strait of Juan de Fuca basin, Main basin, Whidbey basin, South Puget Sound, and Hood Canal) based upon presence and distribution of adult and juvenile rockfish, geographic conditions, and habitat features (Figure 78). The specific areas in the final designation include 590.4 mi^2 (1529 km^2) of nearshore habitat

(sites less than 30 meters deep) for juvenile bocaccio and 414.1 mi^2 (1072.5 km^2) of deepwater habitat (i.e., sites greater than 30 meters deep) for adult and juvenile yelloweye rockfish and adult bocaccio.

NMFS excluded some particular areas from designation because the benefits of exclusion outweigh the benefits of inclusion, and exclusion of those areas will not result in the extinction of the species. Excluded areas covered 64.1 lineal mi (103.1 km) of marine habitat adjacent to Indian lands and approximately 35.8 sq mi (92.7 sq km) of marine habitat area (15.7 sq mi of nearshore, 20.1 sq mi of deepwater) controlled by the Navy. Listed rockfish habitat within areas controlled by the Navy represents approximately eight percent of the nearshore area and approximately six percent of the deepwater area determined to have essential features (79 FR 68041). The following Department of Defense areas were not included as critical habitat:

- Naval Restricted Areas: Admiralty Inlet; Port Gardner, Everett Naval Base; Hood Canal, Bangor; Port Orchard; Sinclair Inlet; Dabob Bay, Whitney Point; Carr Inlet; Port Townsend, Indian Island, Walan Point; Puget Sound, Manchester Fuel Depot; Small Arms Danger Zone off Western Side of Naval Air Station Whidbey Island and additional Accident Potential Zone restricted areas; Strait of Juan de Fuca Naval Air-to-Surface Weapon Range Restricted Area
- Hood Canal and Dabob Bay Naval Non-Explosive Torpedo Testing Area
- NAS Whidbey Island, Crescent Harbor

Juvenile bocaccio settlement habitats located in the nearshore with substrates such as sand, rock and/or cobble compositions that also support kelp are essential for conservation because these features enable forage opportunities and refuge from predators and enable behavioral and physiological changes needed for juveniles to occupy deeper adult habitats (82 FR 7711). The PBFs for juvenile bocaccio in nearshore habitat are: (i) Quantity, quality, and availability of prey species to support individual growth, survival, and feeding opportunities; and (ii) Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities.

Benthic habitats and sites deeper than 30 meters that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for adult bocaccio to avoid predation, seek food and persist for decades (82 FR 7711). PBFs for adult bocaccio in deepwater habitat include the two above for juvenile bocaccio related to prey and water quality, as well as the following: (iii) the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance.

Specific threats to bocaccio critical habitat include degradation of rocky habitat, loss of eelgrass and kelp, introduction of non-native species that modify habitat, and degradation of water quality.

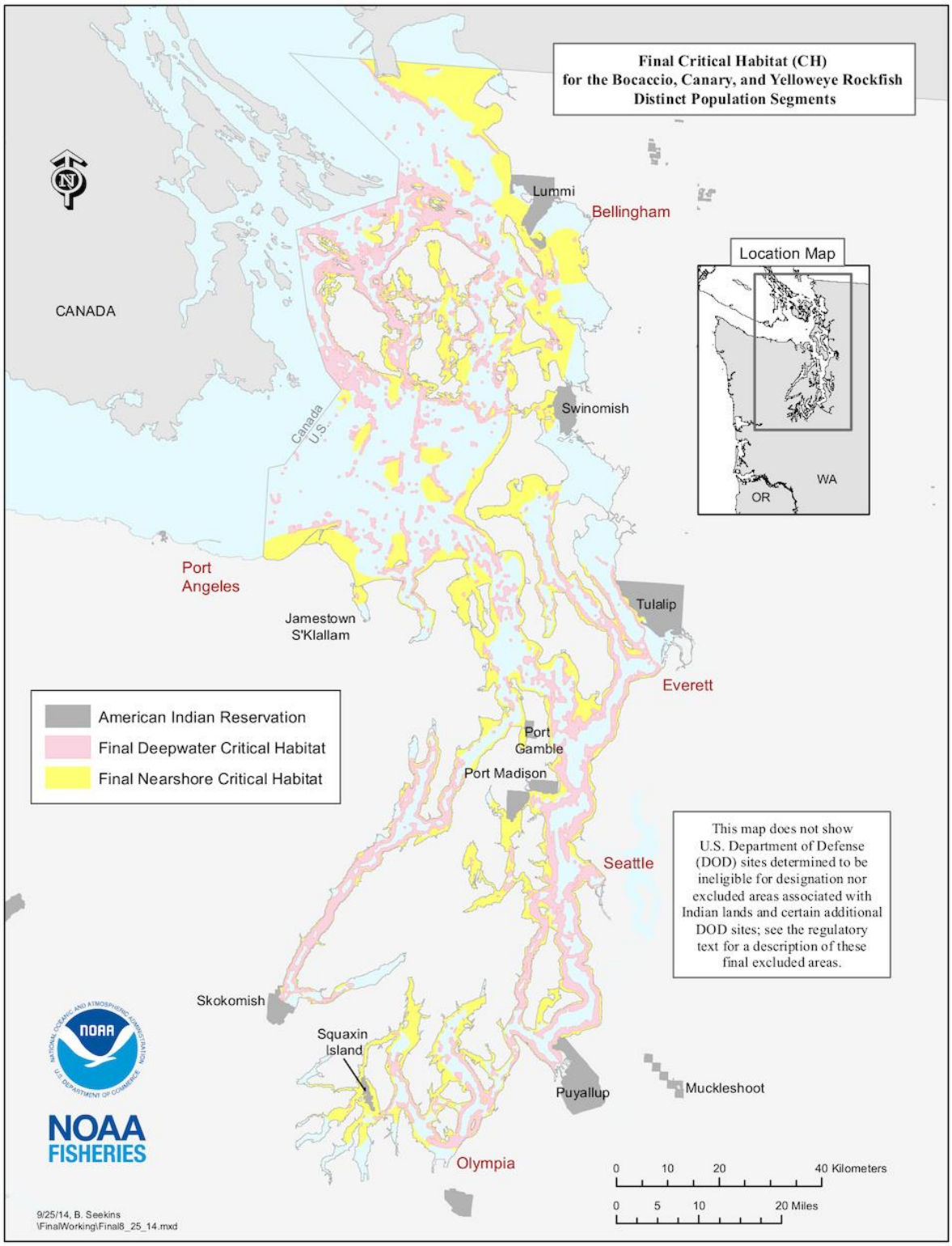


Figure 78. Designated critical habitat for ESA-listed rockfishes.

Recovery Goals

See NMFS (2017h) for complete down listing/delisting criteria for each of the Puget Sound/Georgia Basin DPS of bocaccio's respective recovery goals. The following items were the top recovery objectives identified to support in the Recovery Plan:

(1) Improve our knowledge of the current and historical status of the yelloweye rockfish and bocaccio and their habitats.
(2) Reduce or eliminate existing threats to listed rockfish from fisheries/anthropogenic mortality.
(3) Reduce or eliminate existing threats to listed rockfish habitats and restore important rockfish habitat.

6.2.38 Yelloweye Rockfish – Puget Sound/Georgia Basin DPS

Yelloweye rockfish occur throughout most of the eastern Pacific Ocean ranging from northern Baja California to the Aleutian Islands, Alaska. Puget Sound/Georgia Basin yelloweye rockfish were listed on the ESA as threatened on April 28, 2010 (75 FR 22276). The Puget Sound/Georgia Basin DPS is located along the coastal/inlet waters off the state of Washington and province of British Columbia (Figure 79) and is the only population listed on the Endangered Species Act.

Life History

Yelloweye rockfish larvae are born at 4-5 mm in length and maintain a pelagic existence for the first few weeks of life, before moving to nearshore habitats and settling into rocky reef habitat at about 25 mm in length (DeLacy et al. 1964; Love et al. 2002b; Matarese et al. 1989; Moser 1996). Larval rockfish are extremely fragile and thus have high natural mortality rates. Their small size, relative inability to store food within their gut, and slow swimming speeds likely contribute to this high mortality rate by making them vulnerable to predators and starvation (NMFS 2014a). Poor larval survival in most years provides evidence that rockfish populations persist through what has been termed "the storage hypothesis" (Tolimieri and Levin 2005) where recruitment is generally poor because larval survival and settlement are dependent upon the vagaries of climate, abundance of predators, oceanic currents, and chance events (NMFS 2014a).

As juveniles (1.2-3.6 in), yelloweye rockfish shift to deeper waters ranging from about 30-73 meters in rocky habitats with complex structure (NMFS 2017h). Subadults and adults also prefer deep water (30-425 meters) in rocky habitats with complex structure. Once adult habitat is established, individuals tend to remain at a particular site (NMFS 2008d). Yelloweye growth is thought to vary by latitudinal gradient, with individuals in more northerly regions growing faster and larger. Year class strength appears to be most strongly linked to survival of the larval stage (Laidig et al. 2007). In general, sexual maturity appears to be reached by 50 percent of individuals by 15-20 years of age and 40-50 cm in length (Yamanaka and Kronlund 1997). As

with other rockfish, yelloweyes can be long-lived (reported oldest age is 118 years) (Munk 2001). Maximum size has been reported as 910 cm, but asymptotic size in Alaskan waters for both males and females was estimated to be 690 cm and 659-676 mm along British Columbia (Clemens and Wilby 1961; Love et al. 2005; Rosenthal et al. 1982; Westrheim and Harling 1975; Yamanaka et al. 2006).

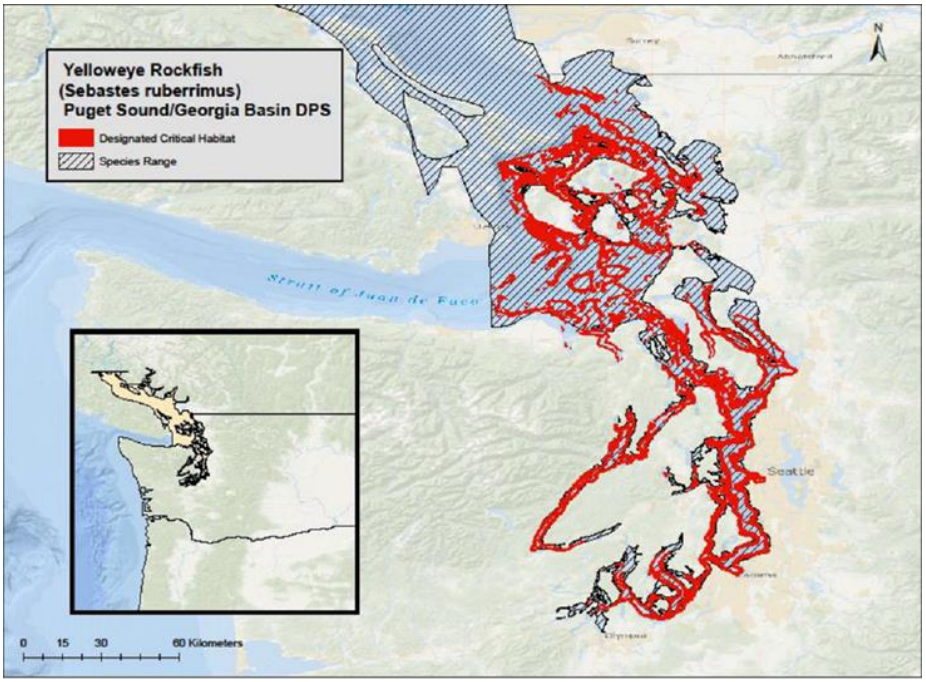


Figure 79. Yelloweye Rockfish, Puget Sound/Georgia Basin DPS range and designated critical habitat.

Population Dynamics

The apparent steep reduction of ESA-listed rockfish in Puget Sound proper (and their consequent fragmentation) has led to concerns about the viability of these populations (Drake et al. 2010). Recreationally caught yelloweye rockfish in the 1970s spanned a broad size range. By the 2000s, fewer older fish in the population were observed (Drake et al. 2010). However, overall fish numbers in the database were also much lower, making it difficult to determine if clear size truncation occurred. With age truncation, the reproductive burden may have shifted to younger and smaller fish. This could alter larval release timing and condition, which may create a mismatch with habitat conditions and potentially reduce offspring viability (Drake et al. 2010).

Spatial distribution provides a protective measure from larger scale anthropogenic changes that damage habitat suitability, such as oil spills or hypoxia, which can occur within one basin but not necessarily the other basins. When localized depletion of rockfish occurs, it can reduce stock resiliency, especially when exacerbated by the natural hydrologic constrictions within Puget Sound (NMFS 2017h). Combining this with limited adult movement, yelloweye rockfish

population viability may be highly influenced by the probable localized loss of populations within the DPS, thus decreasing spatial structure and connectivity.

Rockfish diversity characteristics include fecundity, larvae release timing, larvae condition, morphology, age at reproductive maturity, physiology, and molecular genetic characteristics. The leading factors affecting diversity are the relatively small home ranges of juveniles and subadults and low population size of all life stages (Love et al. 2002a). Yelloweye rockfish spatial structure and connectivity are likely threatened by the apparently severe reduction of fish numbers throughout Hood Canal and South Puget Sound. At 2,330 km^2, Puget Sound is a small geographic area compared with the entire yelloweye rockfish range in the northeastern Pacific.

Results from a recent genetic study comparing yelloweye rockfish individuals from within the PS/GB DPS (n=52) to those outside the DPS (n=52) provided multiple results (NMFS 2016n). First, yelloweye rockfish in inland Canadian waters as far north as Johnstone Strait were genetically similar to those within the PS/GB DPS. Currently, these areas are not included within the boundaries of the DPS. Second, a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait. Lastly, individuals within Hood Canal are genetically differentiated from the rest of the DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (NMFS 2016n).

Yelloweye rockfish were 2.4 percent of the rockfish harvest in the North Sound during the 1960s, 2.1 percent of the harvest during the 1980s, and further decreased to an average of one percent from 1996 to 2002 (Palsson et al. 2009). In Puget Sound proper, yelloweye rockfish were 4.4 percent of the rockfish harvest during the 1960s, 0.4 percent during the 1980s, and 1.4 percent from 1996 to 2002 (Palsson et al. 2009). By the 2000s, evidence of fewer older fish in the population prevailed. Since overall fish numbers in the database were also much lower, it is difficult to determine if size truncation occurred.

The Washington Department of Fish and Wildlife has generated several population estimates of the Puget Sound/Georgia Basin yelloweye rockfish DPS in recent years. ROV surveys in the San Juan Island region in 2008 (focused on rocky substrate) and 2010 (across all habitat types) estimated a population of 47,407±11,761 and 114,494±31,036 individuals, respectively. A 2015 ROV survey of that portion of the DPSs south of the entrance to Admiralty Inlet encountered 35 yelloweye rockfish, producing a preliminary population estimate of 66,998 ± 7,370 individuals (WDFW 2017).

Productivity measures a population's growth rate through all or a portion of its life-cycle. Yelloweye rockfish life-history traits suggest generally low inherent productivity levels because they are long-lived, mature slowly, and have sporadic episodes of successful reproduction (Drake et al. 2010). Adult yelloweye rockfish typically occupy relatively small ranges (Love et al. 2002) and may not move to find suitable mates. So as the density of mature fish has decreased, productivity may have also been impacted by Allee effects. Further, past commercial and

recreational fishing may have depressed the DPS to a threshold beyond which optimal productivity is unattainable (Drake et al. 2010). Also, historic over-fishing may have had dramatic impacts on population size or age structure.

Status

Puget Sound/Georgia Basin yelloweye rockfish abundance is much less than it was historically. The life history of this species, including long lived, slow growth, and late maturity, combined with low survival rates of young make recovery especially challenging. Yelloweye rockfish face several threats including bycatch in commercial and recreational harvest, non-native species introductions, and habitat degradation. Results from a recent genetic study comparing yelloweye rockfish individuals from within the Puget Sound/Georgia Basin DPS to those outside the DPS concluded that a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait (NMFS 2016n). Further, individuals within Hood Canal are genetically differentiated from the rest of the Puget Sound/Georgia Basin DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (NMFS 2016n). NMFS has determined that this DPS is likely to be in danger of extinction in the foreseeable future throughout all of its range; and in its 2016 status review (NMFS 2016n), NMFS has recommended no change in the Puget Sound/Georgia Basin yelloweye rockfish's threatened classification.

Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS of bocaccio, canary rockfish, and yelloweye rockfish was finalized in 2014 (79 FR 68041). For yelloweye rockfish, specific areas in the final designation include 414.1 mi^2 (1072.5 km^2) of deepwater habitat (i.e., sites greater than 30 meters deep) for adult and juvenile life stages (see Figure 78 above). For details on yelloweye rockfish critical habitat, including PBFs, military excluded areas, and specific threats, see the critical habitat subsection in the bocaccio status of species section (Section 6.2.37 above).

Recovery Goals

See NMFS (2017h) for complete down listing/delisting criteria for each of the Puget Sound/Georgia Basin DPS of yelloweye rockfish's respective recovery goals. The following items were the top recovery objectives identified in the Recovery Plan:

(1) Improve our knowledge of the current and historical status of the yelloweye rockfish and bocaccio and their habitats.
(2) Reduce or eliminate existing threats to listed rockfish from fisheries/anthropogenic mortality.
(3) Reduce or eliminate existing threats to listed rockfish habitats and restore important rockfish habitat.

7 ENVIRONMENTAL BASELINE

The "environmental baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The following information summarizes the principal natural and human-caused phenomena in the NWTT action area believed to affect the survival and recovery of ESA-listed species (from Section 6.3 above) in the wild.

7.1 Global Climate Change

There is a large and growing body of literature on past, present, and anticipated future impacts of global climate change, exacerbated and accelerated by human activities. Effects of climate change include sea level rise, increased frequency and magnitude of severe weather events, changes in air and water temperatures, and changes in precipitation patterns, all of which are likely to impact ESA resources. NOAA's climate information portal provides basic background information on these and other measured or anticipated climate change effects (see https://www.climate.gov). This section provides some examples of impacts to ESA-listed species and their habitats that have occurred or may occur as the result of climate change. We address climate change as it has affected ESA-listed species and continues to affect species, and we look to the foreseeable future to consider effects that we anticipate will occur as a result of ongoing activities. While the consideration of future impacts may also be suited to our cumulative effects analysis (Section 9), it is discussed here to provide a comprehensive analysis of the effects of climate change. While it is difficult to accurately predict the consequences of climate change to a particular species or habitat, a range of consequences are expected that are likely to change the status of the species and the condition of their habitats both within and outside of the action area.

In order to evaluate the implications of different climate outcomes and associated impacts throughout the 21st century, many factors have to be considered. The amount of future greenhouse gas emissions is a key variable. Developments in technology, changes in energy generation and land use, global and regional economic circumstances, and population growth must also be considered. A set of four scenarios was developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections are employed consistently across the various branches of climate science. The scenarios are referred to as representative concentration pathways (RCPs), which capture a range of potential greenhouse gas emissions pathways and associated atmospheric concentration levels through 2100 (IPCC 2014). The RCP scenarios drive climate model projections for temperature, precipitation, sea level, and other variables: RCP2.6 is a stringent mitigation scenario; RCP2.5 and RCP6.0 are intermediate scenarios; and RCP8.5 is a scenario with no mitigation or reduction in the use of fossil fuels. The IPCC future global climate predictions (2014 and 2018) and national and regional climate predictions included in the Fourth National Climate Assessment for U.S. states and territories (2018) use the RCP scenarios.

The increase of global mean surface temperature change by 2100 is projected to be 0.3 to 1.7°C under RCP2.6, 1.1 to 2.6°C under RCP 4.5, 1.4 to 3.1°C under RCP6.0, and 2.6 to 4.8°C under RCP8.5 with the Arctic region warming more rapidly than the global mean under all scenarios (IPCC 2014). The Paris Agreement aims to limit the future rise in global average temperature to 2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios such as RCP8.5 (Hayhoe et al. 2018). As there remains a fair amount of uncertainty regarding the implementation of mitigation measures with the goal of curbing pollutants contributing to global climate change, our ESA analyses are conducted under the status quo conditions outlined in RCP8.5.

The globally-averaged combined land and ocean surface temperature data, as calculated by a linear trend, show a warming of approximately 1.0°C from 1901 through 2016 (Hayhoe et al. 2018). The IPCC Special Report on the Impacts of Global Warming (2018) (IPCC 2018) noted that human-induced warming reached temperatures between 0.8 and 1.2°C above pre-industrial levels in 2017, likely increasing between 0.1 and 0.3°C per decade. Warming greater than the global average has already been experienced in many regions and seasons, with most land regions experiencing greater warming than over the ocean (Allen et al. 2018). Annual average temperatures have increased by 1.8°C across the contiguous U.S. since the beginning of the 20th century with Alaska warming faster than any other state and twice as fast as the global average since the mid-20th century (Jay et al. 2018). Global warming has led to more frequent heatwaves in most land regions and an increase in the frequency and duration of marine heatwaves (Allen et al. 2018). Average global warming up to 1.5°C as compared to pre-industrial levels is expected to lead to regional changes in extreme temperatures, and increases in the frequency and intensity of precipitation and drought (Allen et al. 2018).

Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Since the early 1980s, the annual minimum sea ice extent (observed in September each year) in the Arctic Ocean has decreased at a rate of 11 to 16 percent per decade (Jay et al. 2018). Further, ocean acidity has increased by 26 percent since the beginning of the industrial era (IPCC 2014) and this rise has been linked to climate change. Climate change is also expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (IPCC 2014).

Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (Evans and Bjørge 2013; IPCC 2014; Kintisch 2006; Learmonth et al. 2006; MacLeod et al. 2005; McMahon and Hays 2006; Robinson et al. 2005). Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Becker et al. 2018; Silber et al. 2017; Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring. For example, in sea turtles, sex is determined by the ambient sand temperature (during the middle third of incubation) with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25 to 35°C (Ackerman 1997). Increases in

global temperature could skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007a; NMFS and USFWS 2007c; NMFS and USFWS 2013a; NMFS and USFWS 2013b; NMFS and USFWS 2015). These impacts will be exacerbated by sea level rise. The loss of leatherback nesting habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. According to Holsman et al. (2019), in the North Pacific, some fish and crab species in the Bering Sea may move into northern Bering Sea waters where fishing is more limited. As a result, small boat fisheries and shore-based subsistence and recreational fishers in this region are likely vulnerable to climate change. Modeling conducted by Woodworth-Jefcoats et al. (2017) showed that increasing temperatures under RCP8.5 may alter the spatial distribution of tuna and billfish species richness across the North Pacific. The models also projected that zooplankton densities would decline across this region. Such declines would be amplified relative to declines in phytoplankton densities. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback turtles were predicted to gain core habitat area, whereas loggerhead turtles and blue whales were predicted to experience losses in available core habitat. McMahon and Hays (2006) predicted increased ocean temperatures will expand the distribution of leatherback turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean.

MacLeod (2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans will be affected by climate change, with 47 percent predicted to experience unfavorable conditions (e.g., range contraction). Willis-Norton et al. (2015) acknowledged there will be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback turtles in the eastern South Pacific Ocean. Using climate velocity trajectories under RCP8.5, Molinos et al. (2016) found that net marine species richness would peak at latitudes around 40 degrees North and 30 degrees South, while widespread species richness loss would occur near the equator, especially in the Central Indo-Pacific.

Similarly, climate-related changes in important prey species populations are likely to affect predator populations. For example, blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et

al. 1999; Payne et al. 1986; Payne et al. 1990). Pecl and Jackson (2008) predicted climate change will likely result in squid that hatch out smaller and earlier, undergo faster growth over shorter life-spans, and mature younger at a smaller size. This could have negative consequences for species such as sperm whales, whose diets can be dominated by cephalopods. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperatures regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009).

This review provides some examples of impacts to ESA-listed species and their habitats that may occur as the result of climate change. While it is difficult to accurately predict the consequences of climate change to a particular species or habitat, a range of consequences are expected that are likely to change the status of the species and the condition of their habitats.

Hazen et al. (2012) predicted up to 35 percent change in core habitat for some key Pacific species based on climate change scenarios predicated on the rise in average sea surface temperature by 2100. Climate-mediated changes in the distribution and abundance of keystone prey species like krill and in cephalopod populations worldwide will likely affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. Blue whales, as predators that specialize in eating krill, seem likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1990); if they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations seem likely to experience declines similar to those observed in other krill predators, which would cause dramatic declines in their population sizes or would increase the year-to-year variation in population size; either of these outcomes would dramatically increase the extinction probabilities of these whales. Sperm whales, whose diets can be dominated by cephalopods, would have to re-distribute following changes in the distribution and abundance of their prey. This statement assumes that projected changes in global climate would only affect the distribution of cephalopod populations, but would not reduce the number or density of cephalopod populations. If, however, cephalopod populations collapse or decline dramatically, sperm whale populations are likely to collapse or decline dramatically as well.

Southern Resident killer whales might shift their distribution in response to climate-related changes in their salmon prey (NMFS 2019e). Climatic conditions affect salmonid abundance, productivity, spatial structure, and diversity through direct and indirect impacts at all life stages (e.g., Crozier et al. 2008; Independent Science Advisory Board 2007; Lindley et al. 2007; Moyle et al. 2013; Wainwright and Weitkamp 2013). Studies examining the effects of long-term climate change to salmon populations have identified a number of common mechanisms by which climate variation is likely to influence salmon sustainability. These include direct effects of temperature such as mortality from heat stress, changes in growth and development rates, and disease resistance (NMFS 2019e). Changes in the flow regime (especially flooding and low flow events) also affect survival and behavior. Expected behavioral responses include shifts in seasonal timing of important life history events, such as the adult migration, spawn timing, fry emergence timing, and the juvenile migration. Indirect effects on salmon mortality, growth rates and movement behavior are also expected to follow from changes in the freshwater habitat

structure and the invertebrate and vertebrate community, which governs food supply and predation risk (Crozier et al. 2008; Independent Science Advisory Board 2007; Petersen and Kitchell 2001).

Crozier et al. (2019) conducted an extensive analysis on ESA-listed salmonid and steelhead vulnerability to climate change. Nearly all listed populations faced high exposures to projected increases in stream temperature, sea surface temperature, and ocean acidification. The highest vulnerability scores for extrinsic effects (anthropogenic stressors) occurred in interior and southern regions where climate is expected to change the most. Populations ranked as the most vulnerable to climate change overall were California Central Valley Chinook salmon, California and southern Oregon coho salmon, Snake River Basin sockeye salmon, and Columbia and Willamette River spring-run Chinook salmon (Crozier et al. 2019).

In the marine ecosystem, salmon may be affected by warmer water temperatures, increased stratification of the water column, intensity and timing changes of coastal upwelling, loss of coastal habitat due to sea level rise, ocean acidification, and changes in water quality and freshwater inputs (Independent Science Advisory Board 2007; Mauger et al. 2015). Salmon marine migration patterns could be affected by climate-induced contraction of thermally suitable habitat. Abdul-Aziz et al. (2011) modeled changes in summer thermal ranges in the open ocean for Pacific salmon under multiple IPCC warming scenarios. For chum salmon, pink, coho salmon, sockeye salmon and steelhead, they predicted contractions in suitable marine habitat of 30-50% by the 2080s, with an even larger contraction (86-88%) for Chinook salmon under the medium and high emissions scenarios. Northward range shifts are a climate response expected in many marine species, including salmon (Cheung et al. 2015). However, salmon populations are strongly differentiated in the northward extent of their ocean migration, and hence will likely respond individualistically to widespread changes in sea surface temperature (NMFS 2019e). In a meta-analytical review of multiple peer-reviewed papers on green sturgeon, Rodgers et al. (2019) reported that elevated temperatures significantly reduce growth and hatching success and increase the incidence of larval deformities.

The adaptive capacity of threatened and endangered salmonid species is depressed due to reductions in population size, habitat quantity and diversity, and loss of behavioral and genetic variation (NMFS 2019e). Without these natural sources of resilience, systematic changes in local and regional climatic conditions due to anthropogenic global climate change are more likely to reduce long-term viability and sustainability of salmon populations, although the character and magnitude of these effects will likely vary within and among ESUs (NMFS 2019e). Muñoz et al. (2015) reported finding a constraint on the upper limit of thermal tolerance in the Quinsam River juvenile Chinook salmon population. Although fish in this study exhibited both physiological and genetic capacities to increase their thermal tolerance in response to rising temperatures, results suggest that Pacific salmon populations are physiologically susceptible to the projected increases in river temperatures associated with climate change. Based on the observed constraint on thermal tolerance and present-day river temperatures, Muñoz et al. (2015) predict a 17 percent chance of catastrophic loss in the studied population by 2100 based on the average

warming projection, with this chance increasing to 98 percent in the maximum warming scenario.

Anthropogenic climate change is also linked to food web and salinity fluctuations in estuarine environments as a result of sea level rise and seawater intrusion coupled with smaller snowpack and lower spring freshwater flows. Larger and less stable salinity regimes coupled with altered food web dynamics may have direct physiological consequences for green sturgeon juveniles in addition to indirectly affecting the quality and quantity of their prey organisms (Haller et al. 2015). In a meta-analytical review of multiple peer-reviewed papers on green sturgeon, Rodgers et al. (2019) reported that, on average, exposure to elevated salinity levels negatively affected growth, and that plasma osmolality and muscle moisture are significantly increased in response to salinity exposure. Haller et al. (2015) studied the effect of nutritional status on the osmoregulation of green sturgeon. The largest disturbances caused by feed restriction were observed at the highest salinity treatments across all feeding regimes, and the interaction between feed restriction and acute salinity exposure at the highest salinity treatment resulted in high mortality rates during the first 72 hours of salinity exposure (Haller et al. 2015). Sardella et al. (2014) studied the physiological responses of green sturgeon to potential global climate change stressors. They found that while sturgeon can acclimate to changes in salinity, salinity fluctuations resulted in substantial cellular stress.

Effects of ocean acidification on ESA-listed fish most likely occur through ecological mechanisms mediated by changes to the food web (Busch et al. 2013; Crozier et al. 2019). Taxa directly affected by declining marine pH include invertebrates such as pteropods, crabs, and krill. Physiological effects of acidification may also impair olfaction, which could hinder salmonid homing ability, along with other developmental effects (Crozier et al. 2019).

Tolimieri and Levin (2005) examined the effects of climate variability on bocaccio recruitment. They found that the dynamics of bocaccio populations were governed by rare recruitment events, and that these rare events resulted when specific climate conditions (such as various combinations of temperature and upwelling regimes) occurred at different times in their early life history. The coincidence of such climate patterns only occurred 15 percent of the time. Field and Ralston (2005) noted that recruitment of all species of rockfish appeared to be correlated at large scales and hypothesized that such synchrony was the result of large-scale climate forcing. Exactly how climate influences listed rockfish in the Puget Sound/Georgia Basin is unknown; however, given the general importance of climate to Puget Sound and to rockfish, it is likely that climate strongly influences the dynamics of the listed species (Drake et al. 2010). Climate change impacts on ocean conditions were classified as the most serious threat to the Southern DPS of eulachon by NOAA's Biological Review Team (Gustafson et al. 2010; NMFS 2017e).

7.2 Oceanic Temperature Regimes

Oceanographic conditions in the Pacific Ocean can be altered due to periodic shifts in atmospheric patterns caused by the Southern oscillation in the Pacific Ocean, which leads to El Niño and La Niña events and the Pacific decadal oscillation, and the North Atlantic oscillation. These climatic events can alter habitat conditions and prey distribution for ESA-listed species in

the action areas (Beamish 1993; Hare and Mantua 2001; Mantua et al. 1997); (Benson and Trites 2002; Mundy 2005; Mundy and Cooney 2005; Stabeno et al. 2004). For example, decade-scale climatic regime shifts have been related to changes in zooplankton in the North Atlantic Ocean (Fromentin and Planque 1996), and decadal trends in the North Atlantic oscillation (Hurrell 1995) can affect the position of the Gulf Stream (Taylor et al. 1998) and other circulation patterns in the North Atlantic Ocean that act as migratory pathways for various marine species, especially fish.

The north Pacific heatwave first detected in 2013 to 2014, known colloquially as "The Blob," was associated with a large region of anomalously warm water. This warm water region was located along the west coast of North America and spread as far south as the Baja California Peninsula, where water temperatures were as much as four degrees Celsius higher than normal at depths from zero to 300 meters. This warm water region was accompanied by a strong El Niño event from 2015 to 2016. Data collected from scat and lanugo samples from 2013 to 2016 suggested that Guadalupe fur seals shifted their foraging areas further north and/or offshore during this timeframe, possibly as far north as northern California (Amador-Capitanachi et al. 2020). This northward shift may have resulted from decreased prey availability in more southern latitudes where increases in water temperature were more pronounced (Amador-Capitanachi et al. 2020).

The Pacific decadal oscillation is the leading mode of variability in the North Pacific and operates over longer periods than either El Niño or La Niña/Southern Oscillation events and is capable of altering sea surface temperature, surface winds, and sea level pressure (Mantua and Hare 2002; Stabeno et al. 2004). During positive Pacific decadal oscillations, the northeastern Pacific experiences above average sea surface temperatures while the central and western Pacific Ocean undergoes below-normal sea surface temperatures (Royer 2005). Warm Pacific decadal oscillation regimes, as occurs in El Niño events, tends to decrease productivity along the U.S. west coast, as upwelling typically diminishes (Childers et al. 2005; Hare et al. 1999). Sampling of oceanographic conditions just south of Seward, Alaska has revealed anomalously cold conditions in the Gulf of Alaska from 2006 through 2009, suggesting a shift to a colder Pacific decadal oscillation phase. Cartwright et al. (2019) observed a 73 percent decrease in sightings of mother-calf pairs of humpback whales belonging to the Hawaii DPS between 2013 and 2018 during a positive shift in the Pacific decadal oscillation. This coincided with a buildup of warm water in the central, north, and eastern Pacific, which may have suppressed coastal upwelling and productivity, and therefore the availability of humpback whale prey, in these regions. However, more research needs to be done to determine what effects these phase shifts have on the dynamics of prey populations important to ESA-listed cetaceans throughout the Pacific action area. A shift to a colder or warmer decadal oscillation phase would be expected to impact prey populations, although the magnitude of this effect is uncertain.

Montgomery (2020) found that ocean ecosystem indicators in years of ocean residency are correlated with eulachon abundance in the Columbia River. Large-scale and bottom-up

indicators such as the status of the Pacific Decadal Oscillation and prey abundance describe much of the variation in eulachon abundance. When eulachon are in the ocean when it is a cooler phase, such as in 2011 to 2013, returns two to three years later are higher than when the ocean is in a warmer phase. Time series analysis also indicates eulachon abundance correlates strongly with ocean conditions in the two and three years prior to their return, suggesting dominant life histories of two- and three-year ocean types (Montgomery 2020).

The California Current Large Marine Ecosystem (CCLME) is a productive coastal upwelling ecosystem, where wind-driven upwelling brings enriched cool water to the surface that supports a diverse array of species and sustains important fisheries (Santora et al. 2020). During 2014–2016, a marine heatwave occurred in the North Pacific that resulted in an unprecedented multi-year warming event. The impacts of the marine heatwave were wide ranging, but notably caused a sustained bloom of toxic Pseudo-nitzschia diatoms that led to the persistence of domoic acid (a neurotoxin impacting marine wildlife; e.g., shellfish poisoning, record changes in biodiversity of pelagic species, and an unprecedented delay in the opening of the commercial Dungeness crab (*Metacarcinus magister*) fishery in California (a fixed-gear trap fishery with vertical lines) (Santora et al. 2020). The marine heatwave resulted not only in significant economic loss to fishing communities as a result of closures of shellfish and some finfish fisheries, but also coincided with an alarming rise in whale entanglements, mainly humpback whales (see Section 7.4.1 *Fishing Gear Interactions* below). An evaluation of the regional distribution and spatial intensity of krill (measured by acoustics) and midwater trawl catches of anchovy indicates changes in the availability of prey used by humpback whales preceding and during the marine heatwave (Santora et al. 2020). Humpback whales are flexible foragers that perform rapid distribution changes in response to prey abundance and aggregation intensity, and switch from feeding on krill to schooling fish (Santora et al. 2020). Findings by Santora et al. (2020) suggest that changes in humpback whale distribution and movements (shifts from onshore to offshore feeding), triggered by changes in prey type and availability caused by the marine heatwave, resulted in an increased vulnerability of humpbacks to fishing gear entanglement. The authors conclude that ecosystem shifts and forage availability are a plausible, although unconfirmed, explanation for the increased entanglements during the Pacific marine heatwave, in conjunction with the delayed fishing season.

Harvey (2005) created a generic bioenergetics model for rockfish, finding that productivity of rockfish is highly influenced by climate conditions, such that El Niño-like conditions generally lowered growth rates and increased generation time. The negative effect of the warm water conditions associated with El Niño appears to be common across rockfishes (Moser et al. 2000).

7.3 Whaling

Large whale population numbers in the action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. Prior to current prohibitions on whaling, such as the International Whaling Commission's 1966 moratorium, most large whale species had been depleted to the extent it was necessary to list them as endangered under the Endangered Species Act of 1966. For example, from 1900 to 1965 nearly 30,000 humpback whales were

captured and killed in the Pacific Ocean with an unknown number of additional animals captured and killed before 1900 (Perry et al. 1999). Sei whales are estimated to have been reduced to 20 percent (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). In addition, 9,500 blue whales were reported killed by commercial whalers in the North Pacific between 1910 and 1965 (Ohsumi and Wada. 1972); 46,000 fin whales between 1947and 1987 (Rice 1984); and 25,800 sperm whales (Barlow et al. 1997). North Pacific right whales once numbered 11,000 animals but commercial whaling has now reduced their population to less than 100.

7.4 Unusual Mortality Events

Under the MMPA, an unusual mortality event (UME) is defined as "a stranding that is unexpected; involves a significant die-off of any marine mammal population; and demands immediate response." In the past, an UME was declared for fin and humpback whales in the Pacific Ocean and Gulf of Alaska, from April 23, 2015 to April 16, 2016, where a total of 46 fin and humpback whales were found dead (NMFS 2019a). A primary cause for the UME was not identified but ecological factors, including El Niño (see Section 7.2), were likely contributors (NMFS 2019a).

Between 1989 and 2011, a total of 118 dead stranded animals were found along the Washington and Oregon coastline (Northwest Region Stranding Database; Wilkinson 2013). Between June 20 and November 1, 2007, 19 Guadalupe fur seals stranded on the Washington and Oregon outer coasts, prompting NOAA to declare an Unusual Mortality Event on October 19, 2007 (Lambourn et al. 2012). The UME was officially closed on December 11, 2009. In 2012, approximately 58 Guadalupe fur seals stranded on the outer coasts of Washington and Oregon (Lambourn 2013 pers. comm.). This is three times the number of strandings that prompted the Unusual Mortality Event in 2007. Of all the strandings reported off Washington and Oregon (1989 to 2012), most occurred from mid-May through August with occasional reports between October and December (Northwest Region Stranding Database; Lambourn et al. 2012; Wilkinson 2013).

An UME was declared for Guadalupe fur seals beginning in January 2015, and continuing to the present (2015 to 2020) (NMFS 2020a). The UME was declared due to the increased stranding of Guadalupe fur seals in California, and was expanded to include Oregon and Washington due to the elevated number of strandings there. Strandings began in California in January 2015, were eight times higher than the historical average, and continued to remain well above average through 2019 (Figure 80) (NMFS 2020a). Strandings in Oregon and Washington have been well above typical numbers since 2015 (Figure 81); strandings in these two states were five times higher than the historical average in 2019 (NMFS 2020a). Guadalupe fur seal strandings generally peak in April through June each year. Stranded individuals were mostly weaned pups and juveniles, aged one to two years old. Most stranded individuals showed signs of malnutrition

and had secondary bacterial and parasitic infections. As this UME is currently on-going, we expect Guadalupe fur seals to continue to be impacted.

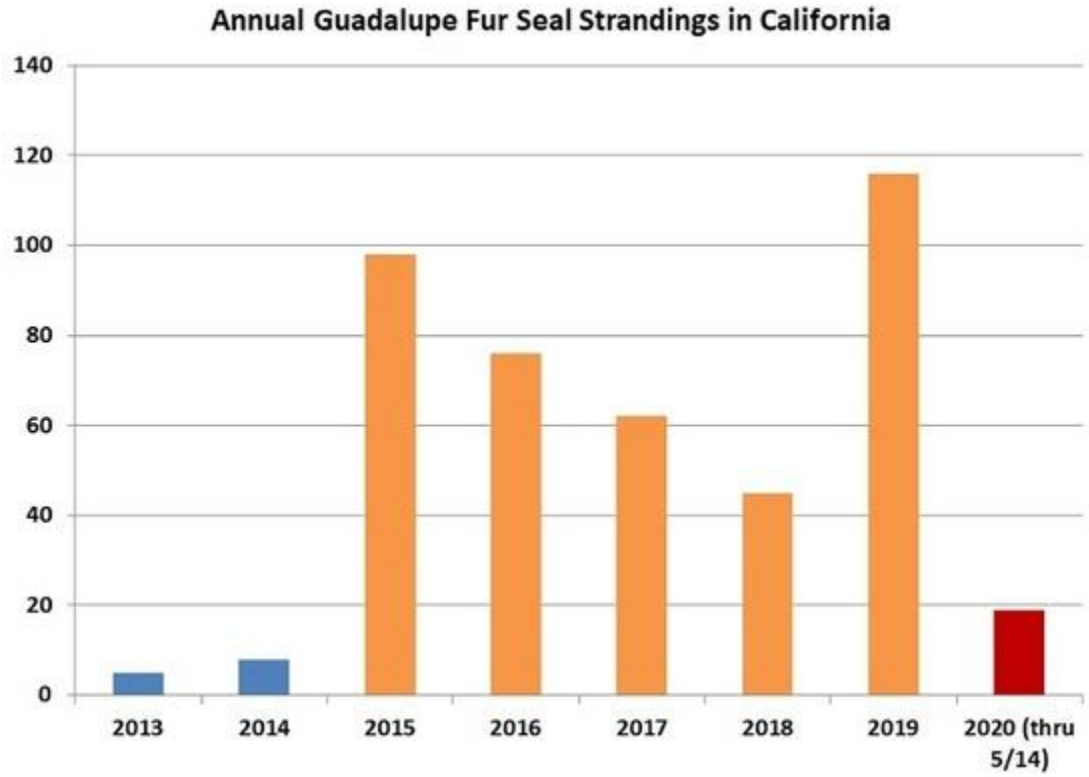


Figure 80. Guadalupe fur seal annual strandings in California, 2013 to present. Orange and red bars indicate unusual mortality event years (NMFS 2020a).



Figure 81. Guadalupe fur seal annual strandings in Oregon and Washington, 2013 to present. Orange/red – Oregon; Blue/light blue – Washington (NMFS 2020a).

7.5 Fisheries Bycatch and Gear Interactions

In this section we address the impacts to ESA-listed fishes, sea turtles, and marine mamals in the action from fisheries bycatch and interactions with commercial fishing gear.

7.5.1 Fisheries Bycatch of Endangered and Threatened Fishes

ESA-listed salmon are incidentally caught in several fisheries that operate in the NWTT action area targeting non-listed salmon or other species. These include:

- Groundfish fisheries off the coasts of Washington, Oregon and California that operate under the Pacific Coast Groundfish Fishery Management Plan;
- Coastal pelagic species (i.e., northern anchovy, squid, Pacific sardine, Pacific mackerel, and jack mackerel) managed by the Pacific Fisheries Management Council under the Coastal Pelagic Species Fisheries Management Plan;
- Commercial salmon fisheries that operate under the Pacific Salmon Treaty;
- Salmon fisheries that are managed by the U.S. Pacific Fisheries Management Council under the Pacific Coast Management Plan;
- Salmon fisheries managed by the U.S. Fraser River Panel;
- Recreational fisheries that operate in the ocean and inland portions of the action area

- Tribal ceremonial and subsistence (gillnet, dip net and hook and line) fisheries in Puget Sound

Fisheries management plans (FMPs) developed for federally regulated fisheries with ESA-listed species bycatch are required to undergo section 7 consultation, including a NMFS issued opinion and an ITS. The ITS includes the anticipated amount of take (lethal and nonlethal) and reasonable and prudent measures with specific terms and conditions for mitigating and minimizing the adverse effects of the proposed action on ESA-listed species and designated critical habitat. Section 7 consultations also evaluate the secondary effects of fisheries removals on ESA-listed species that prey on fish (e.g., Southern Resident killer whales).

Pacific salmon fisheries provide for commercial, recreational, and tribal harvest in ocean and inland waters. A recent Section 7 consultation on this fishery determined that the action was not likely to jeopardize ESA-listed salmonids. It was determined that the fisheries will be managed to keep catch levels within Chinook salmon overall quotas (NMFS 2020b). Commercial ocean fisheries targeting Pacific salmon primarily use troll or hook-and-line gear, but gill nets are also used in commercial and tribal freshwater fisheries in inland waters. The broad geographic range and migration routes of salmon, from the inland tributaries to offshore areas, require comprehensive management by several stakeholder groups representing federal, state, tribal, and Canadian interests (NMFS 2019e).

While management of fishing activities have largely been focused on sustainability and protecting ESA-listed salmonids, management of salmon fisheries with respect to endangered Southern Resident killer whales is also part of the consultation process to evaluate impacts to fish stocks (listed or non-listed) that affect prey available to the Southern Resident killer whales (NMFS 2019e). A growing body of evidence documents how Southern Resident killer whales are affected by limitations of their primary prey, Chinook salmon (Matkin et al. 2017). Availability of Chinook salmon for Southern Resident killer whales is likely affected by multiple factors including sound, competition from other salmon predators (e.g., other resident killer whales and pinnipeds) and fisheries harvest (Chasco et al. 2017b). Both directed and incidental fishing activities may reduce the biomass available to Southern Resident killer whales by removing prey or by selecting for the larger salmon that are preferred by Southern Resident killer whales (NMFS 2008e). Reductions in Chinook salmon prey available to whales from fishery removals vary from year to year and by season and location. In years prior to ESA listings for salmon, fishery reductions were as high as 20-30 percent in some seasons and locations (NMFS 2019e). More recently with ESA considerations for salmon and whales, seasonal reductions in inland and coastal waters have ranged from 0-15 percent reductions. NMFS is currently working on a comprehensive analysis that assesses the effects of fisheries on Chinook salmon availability throughout the Southern Resident killer whales' geographic range, using a retrospective Fishery Regulation Assessment Model (FRAM) based analysis similar to those used in previous fisheries consultations (NMFS 2008b; NMFS 2008c; NMFS 2011d; NMFS 2018c).

The majority of the ongoing impacts that the Southeast Alaska salmon fisheries have on prey availability in the action area would occur in the coastal waters of Washington and Oregon. NMFS estimated that the percent reductions of Chinook salmon in inland waters of Washington from the Southeast Alaska salmon fisheries were expected to range seasonally from 0.1 percent to 2.5 percent, with the greatest reductions occurring in July – September under the 2019 Pacific Salmon Treaty Agreement. Percent reductions in coastal waters of Washington and Oregon from the Southeast Alaska fisheries were expected to range from 0.2 percent to 12.9 percent, and similarly the greatest reductions would occur in July – September. Additional conservation measures were also implemented in 2019 to reduce impacts on Southern Resident killer whales given the whales' declining status including area closures in an area known to be important to Southern Resident killer whales, continuing implementation of a package of outreach and education programs, and continuing the promotion of adhering to voluntary "No-Go" Whale Protection Zone along the western side of San Juan Island (Warren 2019).

The whiting fishery (including at-sea, shore-based, and Tribal fisheries), which is a sector of the Pacific Coast groundfish fisheries, is estimated to have caught an average of 7,718 Chinook salmon each year from 2011 through 2015 (NMFS 2017d). Incidental capture of Chinook salmon in the bottom trawl sector of the groundfish fishery has sharply declined in recent years from an annual average over 15,000 from 2002-2003 to around 557 per year from 2011-2015 (NMFS 2017d). ESA section 7 consultations aim to limit the impact of ocean salmon fisheries on ESA-listed stocks. For example, the maximum age-3 impact rate for 2015 ocean salmon fisheries on Sacramento River winter Chinook salmon is 19 percent (PFMC 2015).

Coastal pelagic fisheries also have the potential to impact Pacific salmon through incidental capture or by removing prey biomass from the ecological system (Pacific Fishery Management Council 2014b). Pelagic fisheries primarily operate off southern and central California, but there is a large sardine fishery off Oregon and Washington, as well as California. Pacific sardine is an important source of forage for a large number of birds, marine mammals, and fish. The directed Pacific sardine fishery has been closed since July 1, 2015 because of low biomass, but small-scale directed fishing can still take place (NMFS 2019e).

Historical overfishing has been recognized as the primary cause of the decline of rockfishes in Puget Sound (Drake et al. 2010; Palsson et al. 2009). Regulatory measures have been taken by the State of Washington over the last several decades to protect all rockfish, including a commercial ban on rockfish fishing in the late 1980s and early 1990s, more recent closures of commercial fisheries with rockfish bycatch (Drake et al. 2010; Palsson et al. 2009), a moratorium initiated in 2010 on recreational rockfish catch, and a 120-ft (36.6-m) depth limit while bottom fishing (WDFW 2014). Despite these measures, ESA-listed rockfish continue to be at risk from bycatch in some of the areas of the DPSs (NMFS 2017h). Rockfish are unintentionally captured as part of fishing activities targeting other species. Although fishers may return these fish to the water, the mortality rate of these fish is extremely high (Parker et al. 2006). Although there are some methods available that could lower the mortality rates of discarded rockfish (summarized by Palsson et al. 2009), application of these methods in the

Puget Sound fishery would be difficult (Palsson et al. 2009). The Washington Department of Fish and Wildlife considers bycatch of rockfish to be a "high impact stressor" on rockfish populations (Palsson et al. 2009). Palsson et al. (2009) report that more than 3,600 pieces of abandoned fishing gear (especially gill nets) have been located in Puget Sound. About 35 percent of this derelict gear has been removed. Derelict nets continue fishing and are known to kill rockfish. While the total impact of this abandoned gear has not been fully enumerated, the Washington Department of Fish and Wildlife has concluded that derelict gear is likely to moderately affect local populations of rockfish (Palsson et al. 2009).

Sport and commercial harvest of green sturgeon has been prohibited since 2006, but past harvest of green sturgeon in California, Washington, and Oregon may have had a significant effect on Southern DPS green sturgeon abundance (Heublein et al. 2017). Southern DPS green sturgeon commercial harvest in the Columbia River Estuary alone from 1983 to 2002 averaged approximately 1,100 individuals, and the harvest of fish from this population in other commercial and sport fisheries likely equaled or exceeded harvest in the Columbia River Estuary over this period (Heublein et al. 2017).

Take of Southern DPS green sturgeon in federal fisheries was prohibited as a result of the ESA 4(d) protective regulations issued in June of 2010 (75 FR 30714). Green sturgeon are occasionally encountered as bycatch in Pacific Coast groundfish fisheries (Al-Humaidhi 2011). The estimated number of Southern DPS green sturgeon encountered in the federally-managed sectors of the groundfish fishery for 2013-2017 ranged from 1-16 per year (Richerson et al. 2019). Among state managed fisheries, bycatch was highest in the California halibut bottom trawl fishery, which encountered an estimated range of between 118-641 Southern DPS green sturgeon annually from 2013-2017 (Richerson et al. 2019). Doukakis et al. (2020) estimated post-release survival of green sturgeon captured and released from bottom trawls used to target California halibut. Of the 51 tagged green sturgeon for which data were recovered, they estimated that 11 died, for an estimated survival rate 82 percent. The California nearshore groundfish sector caught an estimated 16 Southern DPS individuals in 2017, although from 2002-2016 none were caught in this fishery.

Approximately 50 to 250 green sturgeon are encountered annually by recreational anglers in the lower Columbia River (NMFS 2015e), of which 86 percent are expected to be Southern DPS green sturgeon based on the higher range estimate of Israel et al. (2009). Green sturgeon are also caught incidentally by recreational anglers fishing in Washington outside of the Columbia River (NMFS 2015e). Southern DPS green sturgeon are also captured and released by California recreational anglers. Based on self-reported catch card data, an average of 193 green sturgeon were caught and released annually by California anglers from 2007 to 2013 (NMFS 2015e). Recreational catch and release can potentially result in indirect effects on green sturgeon, including reduced fitness and increased vulnerability to predation. However, the magnitude and impact of these effects on Southern DPS green sturgeon, including the potential for post-release mortality, are not well studied.

The main source of eulachon bycatch are the west coast shrimp fisheries (NMFS 2017g). Offshore trawl fisheries for ocean shrimp (*Pandalus jordani*) occur off the west coast of North America from the west coast of Vancouver Island to Cape Mendocino, California (Hannah and Jones 2007) and in British Columbia, Canada. *Pandalus jordani* is known as the smooth pink shrimp in British Columbia, ocean pink shrimp or smooth pink shrimp in Washington, pink shrimp in Oregon, and Pacific Ocean shrimp in California. The ocean shrimp season is open April 1 through October 31 in California, Oregon and Washington and ships deliver catch to shore-based processors. Total coast-wide ocean shrimp landings have ranged from a low of 1,888 metric tons in 1957 to a high of 46,494 metric tons in 2015 (NMFS 2017g).

Prior to 2000, bycatch in the ocean shrimp fishery ranged from 32 to 61 percent of the total catch (Hannah and Jones 2007). Eulachon occur as bycatch in shrimp trawl fisheries off the coasts of Washington, Oregon, California, and British Columbia (Gustafson et al. 2010). Ward et al. (2015) found that the coastal areas just south of Coos Bay, Oregon; between the Columbia River and Grays Harbor, Washington; and just south of La Push, Washington were consistent hotspots of eulachon bycatch across years. The previously depressed and currently increasing abundance of the Southern DPS of eulachon (James et al. 2014) are likely contributing to the increased levels of eulachon bycatch reported for 2012 to 2014. The dramatic increases in the level of eulachon bycatch in both the Washington and Oregon ocean shrimp trawl fisheries in 2012 and 2013 occurred in spite of regulations requiring the use of bycatch reduction devices. It is unclear why bycatch ratios were highest in the Washington, intermediate in the Oregon, and lowest in the California sectors of the ocean shrimp trawl fishery in 2012 and 2013. However, the bycatch ratio increased in Oregon and decreased in Washington in 2014 compared to the previous two-year period. Use of bycatch reduction devices in offshore shrimp trawl fisheries, which was mandated beginning in 2002 in California and 2003 in Washington and Oregon has substantially reduced bycatch of fin fish in these fisheries (Frinodig et al. 2009; Hannah and Jones 2007).

7.5.2 Fisheries Bycatch and Entanglement of Leatherback Sea Turtles

Fisheries bycatch mortality represents a primary threat to leatherback sea turtles. Spotila et al. (1996) and Eckert et al. (2007) noted that adult mortality rates increased significantly as a result of driftnet and longline fisheries. Spotila (2004) concluded that a conservative estimate of annual leatherback fishery-related mortalities (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s is 1,500 animals. He estimates that this represented about a 23 percent mortality rate (or 33 percent if most mortality was focused on the East Pacific population). In the Pacific Ocean, between 1,000 and 1,300 leatherback sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004).

From 1990 to 2009, there were 24 observed leatherback turtle interactions in the California drift gillnet fishery based on 15.6 percent per year observer coverage (Martin et al. 2015 as cited in NMFS 2020d). In 2001, NMFS implemented regulations (i.e., a large time/area closure in Central California) that reduced interactions by approximately 80 to 90 percent, with only two leatherback turtle interactions (both alive) observed based on 20 to 30 percent observer coverage since regulations were implemented (NMFS 2020d). Drift gillnet fishing is prohibited annually

from August 15 to November 15 within the California leatherback turtle conservation area. Currently, NMFS anticipates up to 10 interactions (or 7 mortalities) over a 5-year period (NMFS 2020d).

Shallow-set longline fisheries based out of Hawai'i are estimated to have captured and killed several hundred leatherback sea turtles per year before they were closed in 2001. They were re-opened in 2004, with substantial modifications to protect sea turtles. Since 2004, the Hawaii shallow-set pelagic longline fishery has been regulated to reduce leatherback interactions. This includes 100 percent fishery observer coverage and annual limits on the numbers of interactions that occur between fishing vessels and sea turtles (26 leatherbacks, 34 loggerheads). If either limit is reached, the shallow-set longline fishery is immediately closed for the remainder of the calendar year. Additionally, the shallow-set longline fishery targeting swordfish is required to use 18/0 circle hooks and mackerel bait, which has been shown to reduce leatherback capture rates by 83 percent. All Pacific U.S. commercial fishing vessels are required to have specific equipment on board to release incidentally captured sea turtles, and fishermen and observers are trained on safe handling and release procedures.

Leatherback sea turtles have been documented historically entangled in a large mesh drift gillnet fishery gear targeting swordfish and thresher sharks, although the number of entanglements decreased significantly when a large time/area closure was put in place on the fishery in 2001. Leatherbacks have also been documented entangled in pot and trap fisheries gear. One leatherback sea turtle was confirmed entangled in California fixed fishing gear in 2019.

7.5.3 Entanglement of Marine Mammals in Fishing Gear

Globally, 6.4 million tons of fishing gear is lost in the oceans every year (Wilcox et al. 2015). Entrapment and entanglement in fishing gear is a frequently documented source of human-caused mortality in cetaceans (see Dietrich et al. 2007); in an extensive analysis of global risks to marine mammals, incidental catch was identified as the most common threat category (Avila 2018). Materials entangled tightly around a body part may cut into tissues, enable infection, and severely compromise an individual's health (Derraik 2002). Entanglements also make animals more vulnerable to additional threats (e.g., predation and vessel strikes) by restricting agility and swimming speed. The majority of cetaceans that die from entanglement in fishing gear likely sink at sea rather than strand ashore, making it difficult to accurately determine the extent of such mortalities. Figure 82 shows the number of confirmed whale entanglements per year detected off the U.S. west coast from 2001 to 2016 (Santora et al. 2020). Most entanglements were reported off the coast of California. The number of confirmed whale entanglements, most notably humpback whales, increased markedly throughout the 2014-2016 Pacific marine heat wave event (see Section 7.2 for discussion of Pacific marine heatwave). Dungeness crab fishing gear is the most common source of entanglements in recent years (of those that could be identified to a specific fishery). A total of 26 whales were confirmed entangled off the coasts of Washington, Oregon, and California in 2019, 17 of which were humpbacks (NOAA 2020). By

comparison, there were 34 confirmed humpback whale entanglements in 2018 and 19 in 2017 (NOAA 2020).

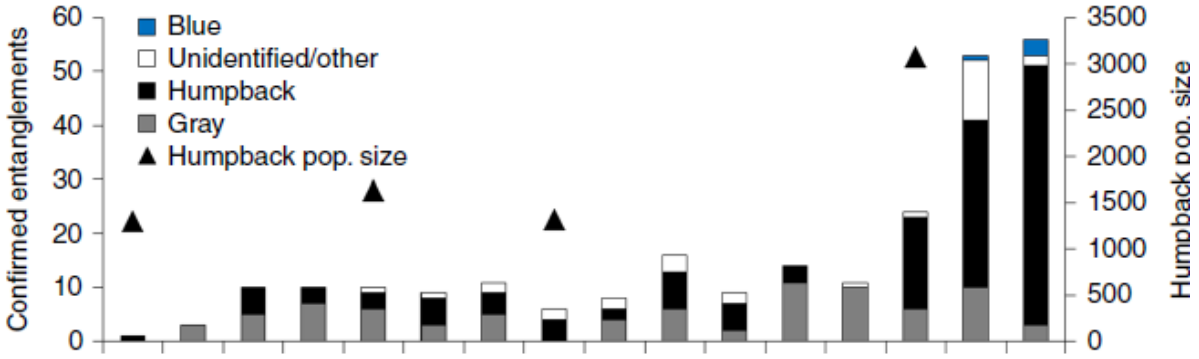


Figure 82. Trend in total confirmed whale entanglements per year detected off the U.S. west coast from 2001 to 2016, and estimated humpback whale population size (Santora et al. 2020).

Cetaceans are also known to ingest fishing gear, likely mistaking it for prey, which can lead to fitness consequences and mortality. Necropsies of stranded whales have found that ingestion of net pieces, ropes, and other fishing debris has resulted in gastric impaction and ultimately death (Jacobsen et al. 2010). As with vessel strikes, entanglement or entrapment in fishing gear likely has the greatest impact on populations of ESA-listed species with the lowest abundance (e.g., Kraus et al. 2016). Nevertheless, all species of cetaceans may face threats from derelict fishing gear.

7.6 Aquaculture

Aquaculture has the potential to impact protected species via entanglement and/or other interaction with aquaculture gear (i.e., buoys, nets, and lines), introduction or transfer of pathogens, increased vessel traffic and noise, impacts to habitat and benthic organisms, and water quality (Clement 2013; Lloyd 2003; Price et al. 2017; Price and Morris 2013). Current data suggest that interactions and entanglements of ESA-listed marine mammals with aquaculture gear are rare (Price et al. 2017). This may be because worldwide the number and density of aquaculture farms are low, and thus there is a low probability of interactions, or because they pose little risk of ESA-listed marine mammals. Nonetheless, given that in some aquaculture gear, such as that used in longline mussel farming, is similar to gear used in commercial fisheries, aquaculture may have impacts similar to fisheries and bycatch. There are very few reports of marine mammal interactions with aquaculture gear, although it is not always possible to determine if the gear animals become entangled in are from aquaculture or commercial fisheries (Price et al. 2017).

Also, some aquaculture gear has the potential for behavioral effects on marine mammals. For example, aquaculture gear may act as a "fish aggregating device" which may attract marine

mammals seeking prey for food, or depredation may occur (Callier et al. 2018). Bottlenose dolphins have been showsn to aggregate around fish cages in Italy and change their social structure by modifying hunting strategies to account for increased prey densitieis around fish farms (reviewed in Callier et al. 2018). Aquaculture gear may also block migration routes (MPI 2013) or at least cause animals to have to circumnavigate the aquaculture gear, as is the case with bottlenose and Dusky dolphins (*Lagenorhynchus obscurus*) avoiding areas with mussel culture longlines (reviewed in Callier et al. 2018; MPI 2013).

7.7 Hatcheries

There are several hundred public facilities (Federal, tribal, and state-operated) producing Pacific salmonids for release into fresh and sea water salmon habitat (Hatchery Scientific Review Group 2015). These hatcheries can be used for production or are designed to rescue wild salmon populations. Salmon hatcheries contribute to the abundance of salmon populations and to the prey base of marine mammals that feed on salmon. However, there are several concerns with how artificial propagation of salmonids may impact natural salmon populations or the habitats essential to their survival. Concerns include a decrease in water quality due to fish waste or chemical disposal, increase in predation of natural fish stocks by hatchery-raised fish, and accidental introduction of non-native species that lead to predation or increased competition with natural salmon populations.

After completing the ocean stage, hatchery-origin fish generally return to tributaries concurrently with natural-origin salmon. Unless they are harvested or collected for broodstock or removal, hatchery-origin fish spawn in natural habitat. While hatcheries can provide a temporary demographic buffer for catastrophic declines in abundance, hatchery populations could eventually be more susceptible to large-scale climate forcing than natural populations due to the absence of behavioral, physiological, and genetic adaptation in the wild (Crozier et al. 2019).

7.8 Vessel Strike

The NWTT action area is home to major shipping facilities in Puget Sound. Figure 83 displays one month of commercial ship automated identification system position data for the Washington portion of the action area.

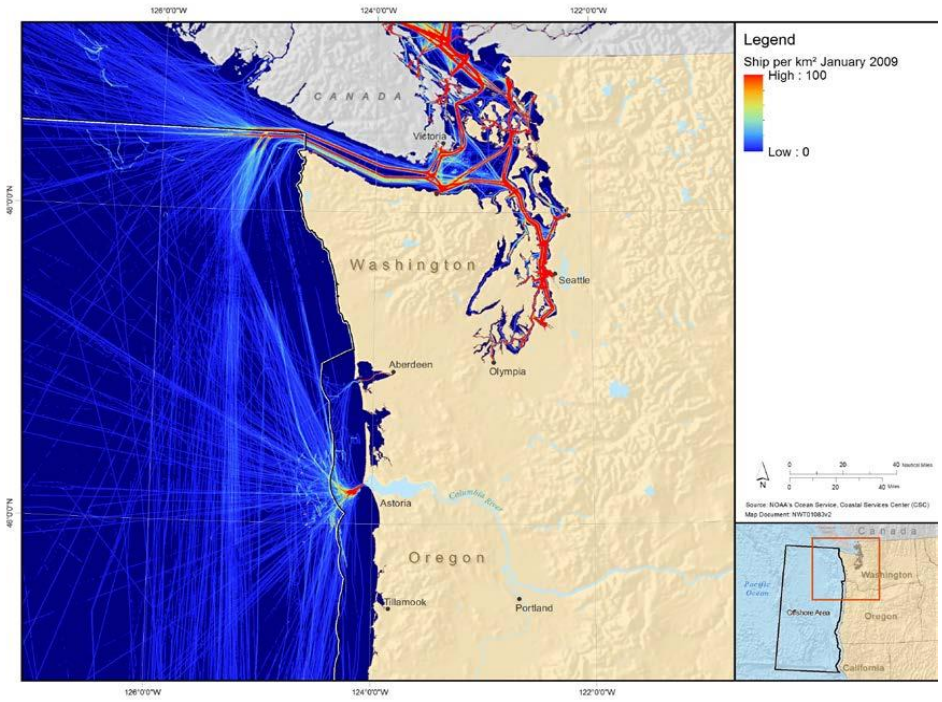


Figure 83. Commercial Ship Automated Identification System Position Data for Washington State for January 2009

Collisions with commercial ships are an increasing threat to many large whale species, particularly as shipping lanes cross important large whale breeding and feeding habitats or migratory routes. Allen et al. (2012) recorded the noises from 24 ships ranging in length from 10.4 meters to 294.1 meters at hydrophone depths of 5, 15, and 25 meters and calculated source levels to characterize the three-dimensional acoustic environment a mysticete would encounter during a whale/ship approach. Results indicated that mysticetes near the sea surface may experience greater difficulty localizing oncoming ships than in deeper waters as a combined result of lower SLs at the surface in shallow locations, bow null effect acoustic shadow zones, and masking from ambient noise. As a consequence, the range of detection for a ship may be too close for a mysticete to execute a successful avoidance maneuver.

Based on the data available from Douglas et al. (2008), Jensen and Silber (2004), and Laist et al. (2001), there have been at least 25 incidents in which marine mammals are known to have been struck by ships in the Puget Sound region and southwestern British Columbia. The marine mammals that were involved in almost half of these incidents died as a result of the strike and they suffered serious injuries in four of those strikes. Virtually all of the rorqual whale species have been documented to have been hit by vessels. This includes blue whales (Berman-Kowalewski et al. 2010; Calambokidis 2012; Van Waerebeek et al. 2007), fin whales (as recently as November 2011 in San Diego) (Douglas et al. 2008; Lammers et al. 2003; Van Waerebeek et al. 2007), sei whales (Felix and Waerebeek 2005; Van Waerebeek et al. 2007), Bryde's whales (Felix and Waerebeek 2005; Van Waerebeek et al. 2007), minke whales (Van Waerebeek et al. 2007), and humpback whales (Douglas et al. 2008; Lammers et al. 2003; Van

Waerebeek et al. 2007). Based on these data sources, fin whales were struck most frequently, accounting for almost 30 percent of the total number of incidents and two-thirds of the incidents in which the whale died as a result of the collision. Northern resident killer whales were struck slightly less frequently, although a cluster of ship strikes in 2006 accounted for four of the six ship strikes involving this population of killer whales. Humpback whales were third in frequency, followed by Southern Resident killer whales, offshore killer whales, and blue whales. About two-thirds (17 out of 25) of the incidents occurred in waters off British Columbia, although the locations were variable. An adult male Southern Resident killer whale (L98) was killed in a collision with a tug boat in 2006 off Vancouver Island.

Douglas et al. (2008) summarized humpback whale ship strike information off the Washington coast and the Strait of Juan de Fuca between 1980-2006 and found only one record of a ship-struck humpback, located on the Pacific coast north of Grays Harbor, Washington. The relatively low rate of ship strikes off the Washington coast despite the high levels of ship traffic in the area was hypothesized to be caused by underreporting of such events and the smaller concentrations of humpbacks in this area compared to locations like Hawaii and Alaska (Douglas et al. 2008). From 1996 to 2002, eight humpback whales were reported struck by vessels in Alaskan waters.

From the Marine Mammal Health and Stranding Response Program database, the number of confirmed vessel collisions with ESA-listed species in Oregon and Washington from 2000-2018 are: three sperm whale, three humpback whales, and ten fin whales. The number of confirmed vessel collisions with ESA-listed species in California from 1986-2019 is shown in Figure 84. Since only northern California is included in the NWTT action area, not all of the vessel collisions shown in Figure 84 occurred within the action area.

It is important to note that many strikes may occur and go unnoticed, while others may occur and subsequently not get reported. Carcass recovery rates have been estimated for various cetacean species including a rate of 6.5 percent for killer whales, less than five percent for grey whales, and 3.4 percent for sperm whales. In modelling ship strike mortality for three baleen whales species off the coast of California, Rockwood et al. (2017) used a high recovery rate of 17 percent based on right whales to produce minimum strike estimates and a five percent recovery (the mean of grey, killer and sperm whales) as a best estimate. The higher rate for right whales is based on them being a more buoyant species (Rockwood et al. 2017).

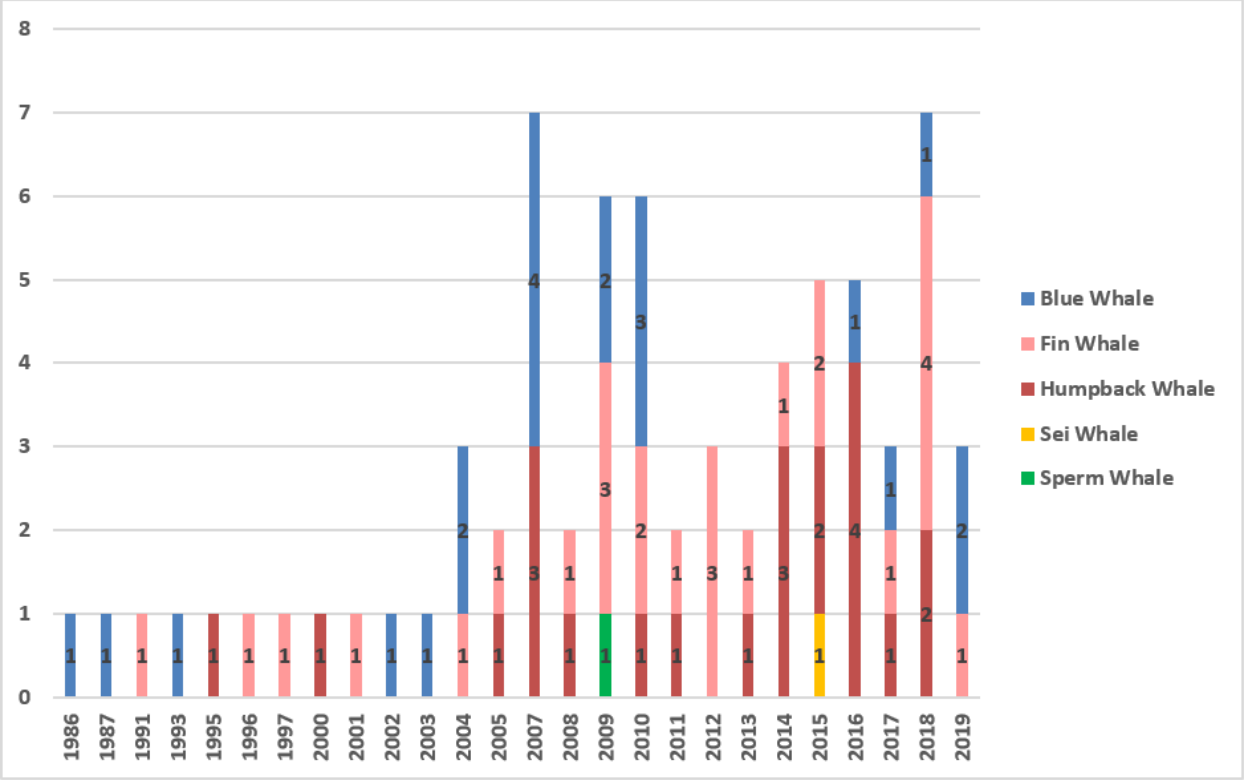


Figure 84. Number of confirmed vessel collisions with ESA-listed species in California from 1986-2019.

7.9 Water Quality Degradation

Puget Sound is the second largest estuary in the U.S. and is highly impacted by humans. Between 2000 and 2006, counties in Puget Sound increased by 315,965 people or by more than 50,000 people per year, with associated increases in the area of impervious surface and population density per square mile of impervious surface in the Puget Sound region (PSAT 2007). The current regional population (about four million) is expected to grow to 5.8 million by 2050, increasing pollution pressures on ecological health.

Between 1991 and 2001, the area of impervious surface in the Puget Sound basin increased 10.4 percent (PSAT 2007). By 2001, impervious surface covered 7.3 percent of the Puget Sound region below 1,000 feet elevation; in some counties and watersheds in the region, this area was substantially higher. Over the same time interval, about 190 mi^2 of forest (about 2.3 percent of the total forested area of the Puget Sound basin) was converted to other uses. In areas below 1,000 feet elevation, the change was more dramatic: 3.9 percent of total forest area was converted to other uses. By 2004, about 1,474 fresh and marine waters in Puget Sound were listed as "impaired waters" in Puget Sound. Fifty-nine percent of these waters tested were impaired because of toxic contamination, pathogens, low dissolved oxygen or high temperatures.

The quality of water in the Puget Sound Basin and aquatic biota those water support have been affected by a range of forestry, agricultural, and urban development practices. The chemical quality of surface water in the foothills and mountains is generally suitable for most uses. However, the physical hydrology, water temperature, and biologic integrity of streams have been influenced to varying degrees by logging (Ebbert et al. 2000).

Because of development, many streams in the Puget Lowlands have undergone changes in structure and function with a trend toward simplification of stream channels and loss of habitat (Ebbert et al. 2000). Sources of contaminants to lowland streams and lower reaches of large rivers are largely nonpoint because most major point sources discharge directly to Puget Sound. Compared with that in small streams in the Puget Lowlands, the quality of water in the lower reaches of large rivers is better because much of the flow is derived from the forested headwaters.

More than half of the agricultural acreage in the basin is located in Whatcom, Skagit, and Snohomish Counties. Agricultural land use consists of about 60 percent cropland and 40 percent pasture. Livestock produce a large amount of manure that is applied as fertilizer to cropland, some- times in excess amounts, resulting in runoff of nitrogen and phosphorus to surface water and leaching of nitrate to ground water. Runoff from agricultural areas also carries sediment, pesticides, and bacteria to streams (Ebbert et al. 2000). Pesticides and fumigant-related compounds are present, usually at low concentrations, in shallow ground water in agricultural areas.

Heavy industry is generally located on the shores of the urban bays and along the lower reaches of their influent tributaries, such as Commencement Bay and the Puyallup River in Tacoma and Elliott Bay and the Duwamish Waterway in Seattle. High-density commercial and residential development occurs primarily within and adjacent to the major cities. Development in recent years has continued around the periphery of these urban areas but has trended toward lower density. This trend has resulted in increasing urban sprawl in the central Puget Sound Basin.

Urban land-use activities have significantly reduced the quality of streams in the Puget Sound Basin (Ebbert et al. 2000). Water-quality concerns related to urban development include providing adequate sewage treatment and disposal, transport of contaminants to streams by storm runoff, and preservation of stream corridors. Water availability has been and will continue to be a major, long- term issue in the Puget Sound Basin. It is now widely recognized that ground-water withdrawals can deplete streamflows (Ebbert et al. 2000), and one of the increasing demands for surface water is the need to maintain instream flows for fish and other aquatic biota.

Water quality has large-scale effects on the marine ecosystem of the Puget Sound – Georgia Basin as evidenced by the intensity and persistence of water stratification in the basin. Historically, Puget Sound was thought to have an unlimited ability to assimilate waste from cities, farms and industries in the region and decisions about human occupation of the landscape were based on that belief. More recent data suggests that the marine ecosystems of the basin have a much more limited ability to assimilate pollution, particularly in areas such as Hood Canal, south Puget Sound, inner Whidbey basin and the central Georgia Basin. In these areas, as

strong stratification has developed and persisted, the respective water quality has steadily decreased. As waters become more stratified, through weather, climate or circulation changes, they become even more limited in their ability to assimilate pollution.

Pollutants found in Puget Sound Chinook salmon have found their way into the food chain of the Sound. Chinook salmon from Puget Sound have two to six times the concentrations of PCBs in their bodies as other Chinook salmon populations on the Pacific Coast. Because of this contamination, the Washington State Department of Health issued consumption advisories for Puget Sound Chinook salmon (PSAT 2007). Harbor seals in southern Puget Sound, which feed on Chinook salmon, have PCB levels that are seven times greater than those found in harbor seals from the Georgia Basin. Concentrations of polybrominated diphenyl ether (also known as PBDE, a product of flame retardants that are used in household products like fabrics, furniture, and electronics) in seals have increased from less than 50 parts per billion in fatty tissue to more than 1,000 ppb over the past 20 years (PSAT 2007).

Contaminants such as PCBs, chlorinated pesticides (e.g., DDT), and PBDEs also appear in rockfish collected in urban areas (West and O'Neil 1998; West et al. 2001; West et al. 2001b). While the highest levels of contamination occur in urban areas, toxins can be found in the tissues of fish in all regions of Puget Sound (PSAT 2007). Rockfish collected in rural areas of the San Juan Islands contained high levels of mercury and hydrocarbons (West et al. 2001). Rockfish pelagic prey, such as Pacific herring in Puget Sound, can have unusually high body burdens of toxins that can biomagnify in their predators. Long lifespan and residency in Puget Sound, both characteristics of the listed rockfish species, increase the risk of exposure.

Sediment contamination levels, based on 2004-2005 EPA and Washington state sediment quality data, within Southern Resident killer whale inland critical habitat areas are shown in Figure 85. The yellow symbols (map on left) indicate concentrations below the "minor adverse effect" level for different contaminants, while the red symbols indicate concentrations above the "minor adverse effect" level for different contaminants. The presence of high levels of persistent organic pollutants (POPs), including PCBs and PBDE, have been documented in Southern Resident killer whales (Herman et al. 2005b; Ross et al. 2000; Ylitalo et al. 2001). Whales become exposed to POPs through their prey as well as through nursing, when adult females offload the contaminants stored in their blubber as it is metabolized to produce milk, which then carries those contaminants to the offspring (NMFS 2019e). High contaminant levels exacerbate the effects of reduced prey abundance as the contaminants become mobilized in the blood stream when stored fat is metabolized in the absence of food. Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of such contaminants.

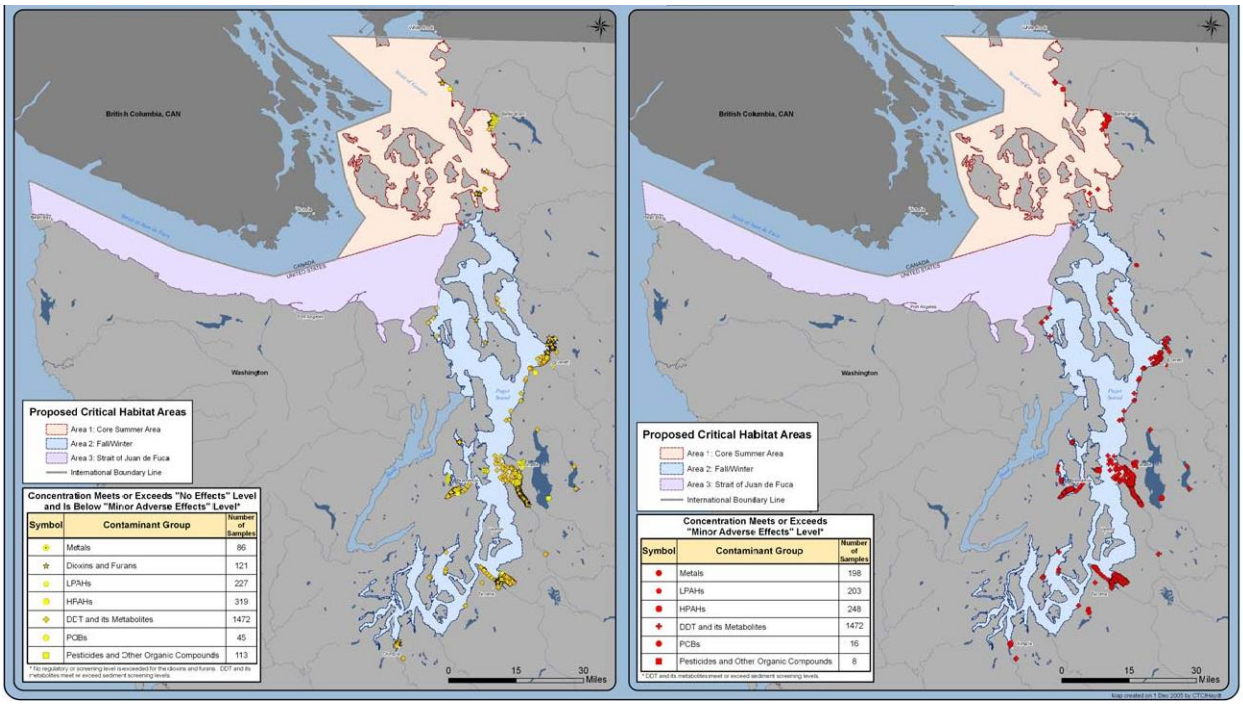


Figure 85. Sediment contamination levels within Southern Resident killer whale inland critical habitat areas (note: military sites excluded from critical habitat designation are not shown on these maps) (NMFS 2006a).

Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain largely unknown, in other species these pollutants have been reported to suppress immune responses (Kakuschke and Prange. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007). High concentrations of POPs have been linked to endocrine, metabolic, and immune disruption, cancer, decreased reproduction, and increased calf mortality (Buckman et al. 2011; de Swart et al. 1996; Gockel and Mongillo 2013; Hall et al. 2018; Lundin et al. 2016; Mongillo et al. 2016; Reijnders 1986; Schwacke et al. 2002; Ylitalo et al. 2005).

Three main contaminants of concern are PCBs, PBDEs, and DDTs, although others do exist. PCB levels have been detected in Southern Resident killer whale blubber samples at concentrations that far exceed the threshold known to have detrimental health effects on harbor seals in Puget Sound (NMFS 2019e). Recent biopsies indicate that the concentration of PCBs in male killer whales has decreased since the 1990s, likely as a result of decreased exposure due to regulations banning their production in the U.S. beginning in 1979 (Krahn et al. 2007; Ross et al. 2000). PBDEs have also been detected at relatively high levels in the whales' blubber. PBDEs have been used in many common household items such as flame retardants since the 1970s, and although banned in both the United States and Canada, they are still prevalent in many products made before 2004. However, based on declining concentrations found in other species and their discontinued production in the U.S. and Canada, the accumulation of PBDEs in Southern

Resident killer whales is expected to slow in similar fashion to PCBs (Elliott et al. 2005; Law et al. 2010; Mongillo et al. 2016; Ross et al. 2013; West et al. 2011). High levels of DDTs have also been found in the whales, especially in K and L pods, which spend more time in California in the winter where DDTs still persist in the marine ecosystem (Sericano et al. 2014). The effects of these three legacy contaminants cannot be considered in isolation, as synergistic, additive, or antagonistic effects may shape their impacts on whale health (NMFS 2019e).

Contaminants of emerging concern (CECs) have been documented in Puget Sound surface waters and organisms and pose threats to ESA-listed species including Southern Resident killer whale and salmonids (Tian et al. 2019). Tian et al. (2019) characterized the occurrence of CECs in the Puget Sound through representative sampling at 18 nearshore marine locations. In total, 87 non-polymeric CECs were identified. The major detected contaminant classes were herbicides, pharmaceuticals, vehicle-related compounds, plasticizers, and flame retardants. The authors noted the presence of perfluorooctanesulfonic acid (PFOS), a contaminant of particular concern to ESA-listed species as bioaccumulation and adverse effects in marine animals have already been documented (Tian et al. 2019). PFOS was found in about 15 percent of samples taken and concentrations exceeded EPA health advisory levels for drinking water at one site. Levels of persistent organic pollutants are known to be relatively high in marine mammals in general, though typically lower in mysticetes as compared to odontocetes due to their different trophic levels (Fleming and Jackson 2011). Southern California humpbacks were found to have the highest levels of DDT, PCBs, and PBDEs of all North Pacific humpbacks sampled on their feeding grounds (Elfes et al. 2010).

Several anthropogenic stressor have been identified that threaten the physical and biological features of Southern DPS green sturgeon critical habitat in coastal bays and estuaries. The application of pesticides is likely to adversely affect prey resources and water quality within the bays and estuaries, as well as the growth and reproductive health of green sturgeon and other ESA-listed fishes through bioaccumulation (NMFS 2018d). Other activities of concern include those that disturb bottom substrates, adversely affect fish prey resources, or degrade water quality through re-suspension of contaminated sediments. Sturgeon prey resources are affected by commercial shipping and activities generating point source pollution and non-point source pollution that discharge contaminants and result in bioaccumulation of contaminants. De Riu et al. (2014) compared the effects of selenium on growth performance, tissue burden, and histopathology between green and white sturgeon. While their results showed adverse effects on both sturgeon species at dietary selenium (selenomethionineon) concentrations within the range of those reported in the benthic macro-vertebrate community of San Francisco Bay, the exposure had a more severe pathological effect on green sturgeon (De Riu et al. 2014). In a meta-analytical review of multiple peer-reviewed papers on green sturgeon, Rodgers et al. (2019) reported that exposure to environmental contaminants (carbaryl, methylmercury and selenium) significantly reduced growth and increased the body burden and mortality rates of green sturgeon.

7.10 Oil Spills

Exposure to petroleum hydrocarbons released into the marine environment via oil spills and other discharge sources represents a serious potential health risk for ESA-listed species. Figure 86 shows the locations of the multiple sources and vectors through which crude oil can accidentally enter the marine and estuarine environment along the Pacific Coast. Figure 87 shows the locations of reported marine oil spill incidents on the Pacific Coast from 1971-2017. PAHs, a component of oil (crude and refined) and motor exhaust, are a group of compounds known to be carcinogenic and mutagenic (Pashin and Bakhitova 1979). Exposure can occur through five known pathways: contact, adhesion, inhalation, dermal contact, direct ingestion, and ingestion through contaminated prey (Rosenberger et al. 2017). Cetaceans have a thickened epidermis that reduces the likelihood of petroleum toxicity from skin contact with oiled waters (Geraci 1990; O'Shea and Aguilar 2001). Inhalation of vapors at the water's surface and ingestion of hydrocarbons during feeding are more likely pathways of exposure. While marine mammals are generally able to metabolize and excrete limited amounts of hydrocarbons, acute or chronic exposure poses greater toxicological risks (Grant and Ross 2002). Matkin et al. (2008) reported that killer whales did not attempt to avoid oil-sheened waters following the *Exxon Valdez* oil spill in Alaska. Observations of Northern Resident killer whales near a more localized spill in Robson Bight at the western end of Johnstone Strait, British Columbia, support those findings (Canadian Press 2007; Williams et al. 2009b).

In marine mammals, acute exposure to petroleum products can cause changes in behavior and reduced activity, inflammation of the mucous membranes, lung congestion, pneumonia, liver disorders, and neurological damage (Geraci and St. Aubin 1990). Previous PAH exposure estimates suggested Southern Resident killer whales can be occasionally exposed to concerning levels (Lachmuth et al. 2011). Some of these impacts can result in population-level consequences that may take decades to recover from (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Oil spills are also potentially destructive to prey populations and therefore may adversely affect Southern Resident killer whales by reducing food availability. One study evaluating the impacts of a potential oil spill on marine mammals in coastal waters of British Columbia characterized Northern and Southern Resident killer whales as being among the most vulnerable due to their small population sizes, strong site fidelity to areas with high oil spill risk, large group size, late reproductive maturity, low reproductive rate, and specialized diet, among other attributes (Jarvela-Rosenberger et al. 2017). A killer whale-specific oil spill response plan was adopted as part of the Northwest Area Contingency Plan (USCG et al. 2018).

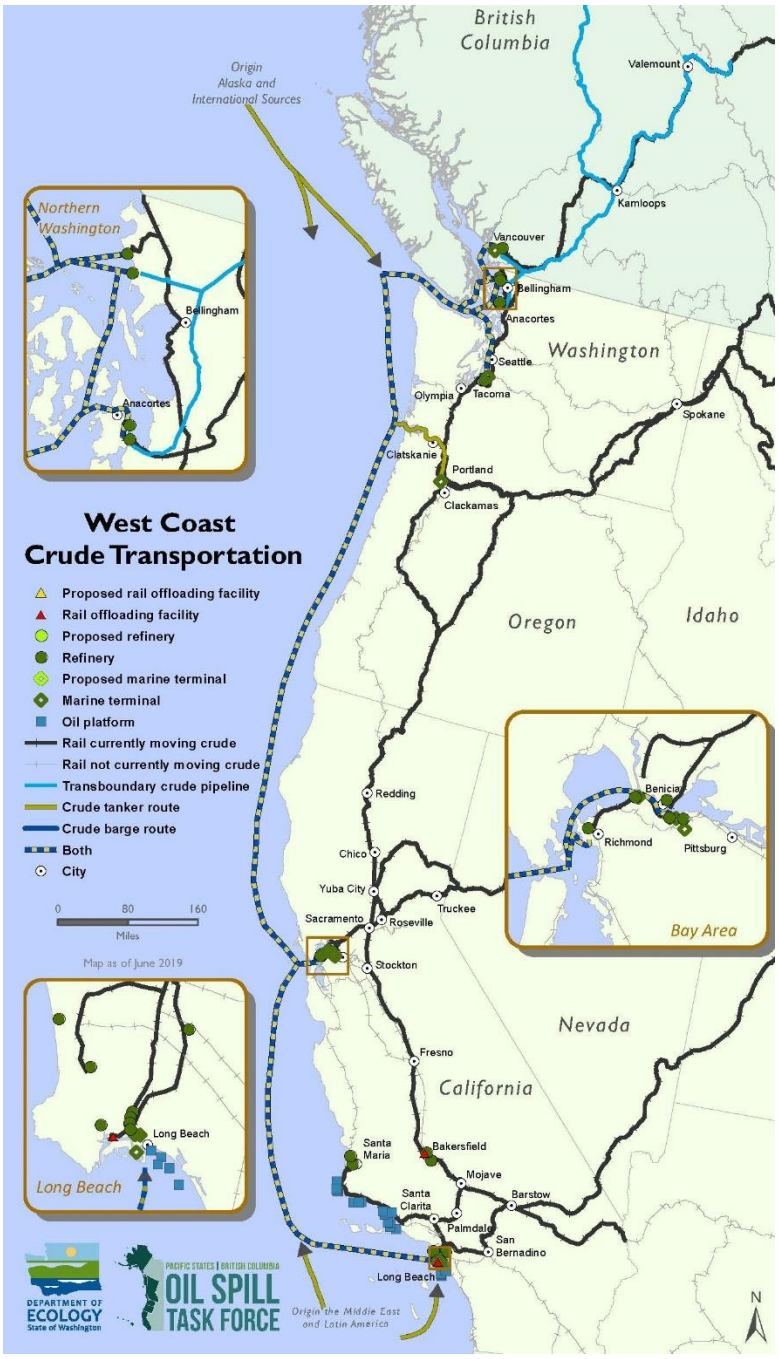


Figure 86. Map of current rail routes, interstate pipelines, and barges transporting crude oil across the West Coast (from: http://oilspilltaskforce.org/wp-content/uploads/2019/07/BCStates_crude_oil_movement_2019_rev4.pdf. Accessed 6 January 2020).

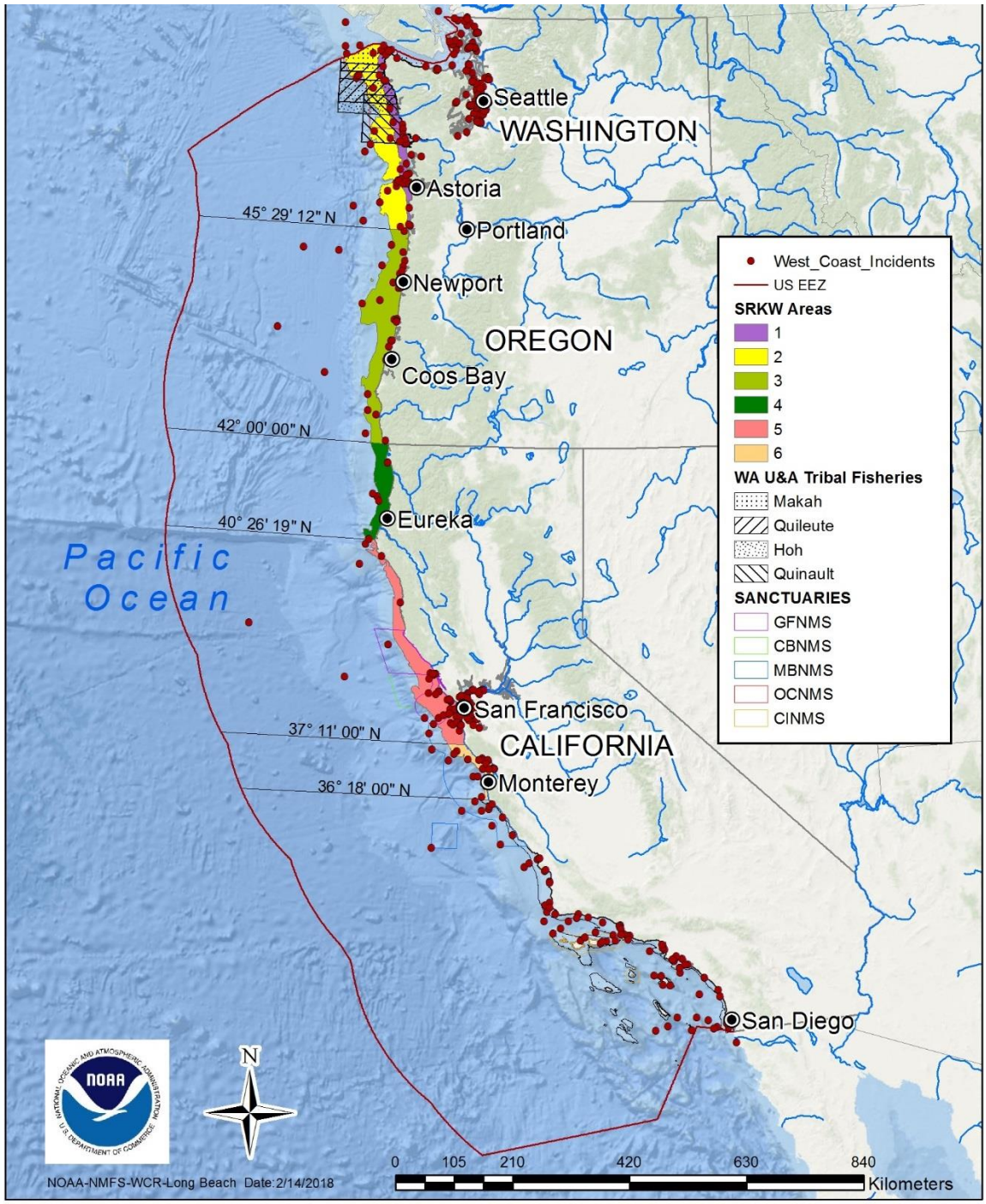


Figure 87. Marine oil spill incidents on the Pacific West Coast from 1971-2017 within areas under consideration for Southern Resident killer whale proposed critical habitat designation (NMFS 2019e).

Some of these impacts can result in population-level consequences that may take decades to recover from (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Oil spills are also potentially destructive to prey populations and therefore may adversely affect

Southern Resident killer whales by reducing food availability. One study evaluating the impacts of a potential oil spill on marine mammals in coastal waters of British Columbia characterized Northern and Southern Resident killer whales as being among the most vulnerable due to their small population sizes, strong site fidelity to areas with high oil spill risk, large group size, late reproductive maturity, low reproductive rate, and specialized diet, among other attributes (Jarvela-Rosenberger et al. 2017). A killer whale-specific oil spill response plan was adopted as part of the Northwest Area Contingency Plan (USCG et al. 2018).

In addition to marine mammals, oil spills can have significant impacts on leatherback sea turtles in the action area. For example, leatherback sea turtles are known to ingest and attempt to ingest tar balls (Atlantic Leatherback Turtle Recovery Team 2006), which can block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2010), ultimately reducing growth, reproductive success, as well as increasing mortality and predation risk (Fraser 2014). Oil exposure can also cause acute damage on direct exposure to oil, including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (NOAA 2010; Vargo et al. 1986). Further, nearshore spills or large offshore spills can release oil on beaches where leatherback sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2010).

ESA-listed fishes can also be impacted oil spills within the action area. Brocksen and Bailey (1973) conducted experiments exposing juvenile Chinook salmon to sub-lethal concentrations of the aromatic hydrocarbon benzene, a water soluble component of crude oil, for periods ranging from 1 to 96 hours. Prior to exposure, and after exposure to the benzene, respiration rates of individual fish were measured. Results showed increases in respiratory rates up to 115 percent above that of control fish after exposure periods of 48 hours for Chinook salmon. Fish exposed to benzene concentrations of 10 parts per million for periods longer than those listed exhibited a narcosis that caused a decrease in respiratory rate. The narcotic state induced by exposure to benzene was shown to be reversible when the fish were placed in fresh water and kept for periods longer than 6 days. Further, Incardona et al. (2015) found that embryonic exposure to very low, environmentally relevant levels of crude oil causes permanent structural and functional changes to the fish heart. Crude oil essentially acts as a potent teratogen that produces specific abnormalities in the compact myocardium and outflow tract following exposure during early heart development. These developmental defects initiated during organogenesis in turn led to reduced cardiorespiratory performance much later in juvenile fish. Hence, embryonic injury following crude oil exposure leads to irreversible impairment.

7.11 Marine Debris

Marine debris has become a widespread threat for a wide range of marine species that are increasingly exposed to it on a global scale. Plastic is the most abundant material type worldwide, accounting for more than 80 percent of all marine debris (Poeta et al. 2017). The

most common impacts of marine debris are associated with ingestion or entanglement. Both types of interactions can result in injury or death of many different marine species taxa. Ingestion occurs when debris items are intentionally or accidentally eaten (e.g. through predation on already contaminated organisms or by filter feeding activity, in the case of large filter feeding marine organisms, such as whales) and enter in the digestive tract. Ingested debris can damage digestive systems and plastic ingestion can also facilitate the transfer of lipophilic chemicals (especially persistent organic pollutants) into the animal's bodies. Entanglement in fishing gear also represents a major, on-going threat to many marine species. An estimated 640,000 tons of fishing gear is lost, abandoned, or discarded at sea each year throughout the world's oceans (Macfadyen et al. 2009). These "ghost nets" drift in the ocean and can fish unattended for decades (ghost fishing), killing, injuring or impairing large numbers of marine animals through entanglement.

Marine debris is a significant concern for ESA-listed species, particularly sea turtles and marine mammals. The initial developmental stages of all turtle species are spent in the open sea. During this time both juvenile turtles and their buoyant food are drawn by advection into fronts (convergences, rips, and driftlines). The same process accumulates large volumes of marine debris, such as plastics and lost fishing gear, in ocean gyres (Carr 1987). An estimated four to twelve million metric tons of plastic enter the oceans annually (Jambeck et al. 2015). It is thought that leatherback sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Laist et al. 1999; Lutcavage et al. 1997). Schuyler et al. (2016) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that up to 52 percent of sea turtles globally have ingested plastic debris and oceanic life stage turtles are at the highest risk of debris ingestion. This study also found the North Pacific gyre to be a regional hotspot for sea turtle debris ingestion. The North Pacific Subtropical gyre is a clockwise circular pattern of four prevailing ocean currents (North Pacific, California, North Equatorial, and Kuroshio currents) where debris from around the North Pacific Rim gathers and circulates (PISC 2016). In addition to ingestion risks, sea turtles can also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices (Laist et al. 1999; Lutcavage et al. 1997; NRC 1990). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Dagorn et al. 2013; Kiessling 2003).

Marine mammals are also highly susceptible to the threats associated with marine debris and many cases of ingestion and entanglement have been reported around the world (Poeta et al. 2017). Baulch and Perry (2014) found that the proportion of cetacean species ingesting debris or becoming entangled in debris is increasing. Based on stranding data, they found that recorded rates of ingestion have increased by a factor of 1.9 and rates of entanglement have increased by a factor of 6.5 over the last forty years (1970-2010). Ingestion of marine debris can also have fatal

consequences for large whales. In 2008, two male sperm whales stranded along the northern California coast with large amounts of fishing net scraps, rope, and other plastic debris in their stomachs. One animal had a ruptured stomach, the other was emaciated, and gastric impaction was suspected as the cause of both deaths (Jacobsen et al. 2010).

7.12 Anthropogenic Sound

The ESA-listed species that occur in the action area are regularly exposed to several sources of natural and anthropogenic sounds. A wide variety of anthropogenic and natural sources contribute to ocean noise throughout the world's oceans. Anthropogenic sources of noise that are most likely to contribute to increases in ocean noise are vessel noise from commercial shipping and general vessel traffic, oceanographic research, oil, gas and mineral exploration, underwater construction, geophysical (seismic) surveys, Naval and other sources of sonar, and underwater explosions (Hatch and Wright 2007b; Richardson et al. 1995b).

Noise is of particular concern to marine mammals because many species use sound as a primary sense for navigating, finding prey, avoiding predators, and communicating with other individuals. As described in greater detail later in this opinion (Sections 8.1.1.1 and 8.2.1.1), noise may cause marine mammals to leave a habitat, impair their ability to communicate, or to cause stress. Noise can cause behavioral disturbances, mask other sounds including their own vocalizations, may result in injury and, in some cases, may result in behaviors that ultimately lead to death. The severity of these impacts can vary greatly between minor impacts that have no real cost to the animal, to more severe impacts that may have lasting consequences. A comprehensive discussion of the potential impacts of anthropogenic noise on listed species is included in the *Effects of the Action* Section 8 of this opinion.

Noise in the marine environment has received a lot of attention in recent years and is likely to continue to receive attention in the foreseeable future. Any potential for cumulative impact should be put into the context of recent changes to ambient sound levels in the world's oceans as a result of anthropogenic activities. There is a large and variable natural component to the ambient noise level as a result of events such as earthquakes, rainfall, waves breaking, and lightning hitting the ocean as well as biological noises such as those from snapping shrimp, other crustaceans, fishes, and the vocalizations of marine mammals (Crawford and Huang 1999; Hildebrand 2004b; Patek 2002). However, several studies have shown that anthropogenic sources of noise have increased ambient noise levels in the ocean over the last 50 years (Jasny et al. 2005; NRC 1994; NRC 2000; NRC 2003; NRC 2005; Richardson et al. 1995b). Much of this increase is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003). Commercial fishing vessels, cruise ships, transport boats, airplanes, helicopters and recreational boats all contribute sound into the ocean (NRC 2003). The military uses sound to test the systems of Navy vessels as well as for naval operations. In some areas where oil and gas production takes place, noise originates from the drilling and production platforms, tankers, vessel and aircraft support, seismic surveys, and the explosive removal of platforms (NRC 2003).

Fisheries can also introduce explosive sounds into the marine environment. For example, at one monitoring site adjacent to Quinault Canyon (off the coast of Washington), explosions identified

as seal bombs (Wiggins et al. 2019; Wiggins et al. 2020) were present during daylight hours 68 percent of the cumulative hours per week during the monitoring period (Wiggins et al. 2017a). The prevalent and continued use of seal bombs in the action area indicated that marine mammals and their prey have been repeatedly exposed to this explosive stressor that is purposefully directed at deterring marine mammals.

Andrew et al. (2002) compared ocean ambient sound from the 1960s to the 1990s from a receiver off the California coast. The data showed an increase in ambient noise of approximately 10 dB in the frequency ranges of 20 to 80 Hz and 200 to 300 Hz, and about 3 dB at 100 Hz over a 33-year period. Each 3 dB increase is noticeable to the human ear as a doubling in sound level. A possible explanation for the rise in ambient noise is the increase in shipping noise. There are approximately 11,000 supertankers worldwide, each operating approximately 300 days per year, each producing constant broadband noise at typical source levels of 198 dB (Hildebrand 2004b). Generally the most energetic regularly operated sound sources are seismic airgun arrays from approximately 90 vessels with typically 12 to 48 individual guns per array, firing about every 10 seconds (Hildebrand 2004b).

Many researchers have described behavioral responses of marine mammals to the sounds produced by helicopters and fixed-wing aircraft, boats and ships, as well as dredging, construction, geological explorations, etc. (Richardson et al. 1995b). Most observations have been limited to short-term behavioral responses, which included cessation of feeding, resting, or social interactions. Sections 7.12.1, 7.12.2, 7.12.3, and 7.12.4 discuss these sound sources in more detail.

7.12.1 Seismic Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor. Numerous seismic surveys have been conducted off the west coast over the past several decades. Unlike other regions (e.g., Gulf of Mexico) where the large majority of seismic activity is associated with oil and gas development, seismic surveys conducted in the NWTT action area are primarily for scientific research, to identify possible seafloor or shallow-depth geologic hazards, and to locate potential archaeological resources and benthic habitats that should be avoided. Seismic surveys are typically conducted by towing a sound source behind a research vessel, such as an airgun array that emits acoustic energy in timed intervals. The transmitted acoustic energy is reflected and received by an array of hydrophones. This acoustic information is processed to provide information about geological structure below the seafloor. Research geologists conduct seismic surveys to study plate tectonics as well as other topics in marine geology. The underwater sound produced by seismic surveys could affect marine life, including ESA-listed species.

There are two major categories of seismic surveys: (1) deep seismic surveys which include ocean bottom, vertical seismic profile or borehole, 2-dimensional, 3-dimensional, 4-dimensional and wide azimuth surveys, and (2) high resolution surveys. Deep seismic survey acoustic sources consist of airgun arrays while receiver arrays consist of hydrophones or geophones encased in plastic tubing called streamers. High-resolution surveys collect data on surface and near-surface geology used to identify archaeological sites, potential shallow geologic and manmade hazards

for engineering, and site planning for bottom-founded structures. High-resolution surveys may use airguns but also use other sound sources such as sub-bottom profilers (at 2.5-7 kilohertz), echosounders (single-beam at 12-240 kilohertz; multibeam at 50-400 kilohertz), boomers (at 300-3,000 Hz), sparkers (at 50-4,000 Hz), compressed high intensity radar pulse sub-bottom profiler (at 4-24 kilohertz), pingers (at 2 kilohertz), and side-scan sonars (16-1,500 kilohertz).

Exposure of cetaceans to very strong impulsive sound sources from airgun arrays can result in auditory damage, such as changes to sensory hairs in the inner ear, which may temporarily or permanently impair hearing by decreasing the range of sound an animal can detect within its normal hearing ranges (reviewed in Finneran 2015a). A TTS results in a temporary change to hearing sensitivity, and the impairment can last minutes to days, but full recovery of hearing sensitivity is expected. At higher received levels, particularly in frequency ranges where animals are more sensitive, a PTS can occur, meaning lost auditory sensitivity is unrecoverable. Either of these conditions can result from exposure to a single pulse or from the accumulation of multiple pulses, in which case each pulse need not be as loud as a single pulse to have the same accumulated effect. Since there is frequency overlap between airgun array sounds and vocalizations of ESA-listed cetaceans, particularly baleen whales and to some extent sperm whales, seismic surveys could mask these calls at some of the lower frequencies for these species.

ESA-listed cetaceans are expected to exhibit a wide range of behavioral responses as a consequence of being exposed to seismic airgun sound fields and echosounders. Baleen whales are expected to mostly exhibit avoidance behavior, and may also alter their vocalizations. Sperm whales are expected to exhibit less overt behavioral changes, but may alter foraging behavior, including vocalizations. These responses are expected to be temporary with behavior returning to a baseline state shortly after the seismic source becomes inactive or leaves the area. Individual whales exposed to sound fields generated by seismic airguns could also exhibit responses not readily observable, such as stress (Romano et al. 2002), that may have adverse effects. Other possible responses to impulsive sound sources like seismic airguns include neurological effects, bubble formation, resonance effects, and other types of organ or tissue damage (Cox et al. 2006; Southall et al. 2007b; Tal et al. 2015; Zimmer and Tyack 2007), but similar to stress, these effects are not readily observable.

As with cetaceans, ESA-listed sea turtles may exhibit a variety of different responses to sound fields associated with seismic airguns and echosounders. Avoidance behavior and physiological responses from airgun exposure may affect the natural behaviors of sea turtles (Mccauley et al. 2000a). Mccauley et al. (2000a) conducted trials with caged sea turtles and an approaching-departing single air gun to gauge behavioral responses of green and loggerhead sea turtles. Their findings showed behavioral responses to an approaching airgun array at 166 dB re: one micro Pascal rms and avoidance around 175 dB re: 1 micro Pascal rms. From measurements of a seismic vessel operating 3-dimensional airgun arrays in 100 to 120 meters water depth this corresponds to behavioral changes at around two kilometers and avoidance around one kilometer.

NMFS issues permits for seismic activity conducted near marine mammals and ESA-listed sea turtles. MMPA and ESA permits specify the conditions under which researchers can operate seismic sound sources, such as airguns, including mitigation measure to minimize adverse effects to protected species. One such mitigation measure is the suspension of seismic activities whenever marine mammals are observed within the designated safety zone, which differs by species and sound source, as specified in the permit.

7.12.2 Active Sonar

Active sonar emits high-intensity acoustic energy and receives reflected and/or scattered energy. A wide range of sonar systems are in use for both civilian and military applications. The primary sonar characteristics that vary with application are the frequency band, signal type (pulsed or continuous), rate of repetition, and source level. Sonar systems can be divided into categories, depending on their primary frequency of operation; low frequency for one kilohertz and less, mid frequency for one to 10 kilohertz; high frequency for 10 to 100 kilohertz; and very high frequency for greater than 100 kilohertz (Hildebrand 2004a). Low frequency systems are designed for long-range detection (Popper et al. 2014a). The effective source level of an low-frequency active array, when viewed in the horizontal direction, can be 235 dB re 1µPa-m or higher (Hildebrand 2004a). Signal transmissions are emitted in patterned sequences that may last for days or weeks. Mid-frequency military sonars include tactical anti-submarine warfare sonars, designed to detect submarines over several tens of kilometers, depth sounders and communication sonars. High-frequency military sonars includes those incorporated into weapons (torpedoes and mines) or weapon countermeasures (mine countermeasures or anti-torpedo devices), as well as side-scan sonar for seafloor mapping. Commercial sonars are designed for fish finding, depth sounding, and sub-bottom profiling. They typically generate sound at frequencies of 3 to 200 kilohertz, with source levels ranging from 150-235 dB re 1µPa-m (Hildebrand 2004a). Depth sounders and sub-bottom profilers are operated primarily in nearshore and shallow environments, however, fish finders are operated in both deep and shallow areas.

7.12.3 Vessel Sound and Commercial Shipping

Individual vessels produce unique acoustic signatures, although these signatures may change with vessel speed, vessel load, and activities that may be taking place on the vessel. Sound levels are typically higher for the larger and faster vessels. Peak spectral levels for individual commercial vessels are in the frequency band of ten to 50 Hz and range from 195 dB re: μPa^2-s at 1 meter for fast-moving (greater than 20 knots) supertankers to 140 dB re: μPa^2-s at 1 meter for smaller vessels (NRC 2003). Although large vessels emit predominantly low frequency sound, studies report broadband sound from large cargo vessels above two kilohertz, which may interfere with important biological functions of cetaceans (Holt 2008). At frequencies below 300 Hz, ambient sound levels are elevated by 15 to 20 dB when exposed to sounds from vessels at a distance (McKenna et al. 2013).

Much of the increase in sound in the ocean environment over the past several decades is due to increased shipping, as vessels become more numerous and of larger tonnage (Hildebrand 2009; Mckenna et al. 2012; NRC 2003). Shipping constitutes a major source of low-frequency (five to 500 Hz) sound in the ocean (Hildebrand 2004a), particularly in the Northern Hemisphere where the majority of vessel traffic occurs. While commercial shipping contributes a large portion of oceanic anthropogenic noise, other sources of maritime traffic can also impact the marine environment. These include recreational boats, whale-watching boats, research vessels, and ships associated with oil and gas activities.

Vessel noise can result from several sources including propeller cavitation, vibration of machinery, flow noise, structural radiation, and auxiliary sources such as pumps, fans and other mechanical power sources. Kipple and Gabriele (2007) measured sounds emitted from 38 vessels ranging in size from 14 to 962 feet at speeds of 10 knots and at a distance of 500 yards (457.2 meters) from the hydrophone. Sound levels ranged from a minimum of 157 to a maximum of 182 dB re 1 µPa-m, with sound levels showing an increasing trend with both increasing vessel size and with increasing vessel speed. Vessel sound levels also showed dependence on propulsion type and horsepower. Mckenna et al. (2012) measured radiated noise from several types of commercial ships, combining acoustic measurements with ship passage information from Automatic Identification System (AIS). On average, container ships and bulk carriers had the highest estimated broadband source levels (186 dB re 1 lPa2 20 to 1000 Hz), despite major differences in size and speed. Differences in the dominant frequency of radiated noise were found to be related to ship type, with bulk carrier noise predominantly near 100 Hz while container ship and tanker noise was predominantly below 40 Hz. The tanker had less acoustic energy in frequencies above 300 Hz, unlike the container and bulk carrier.

Sound emitted from large vessels, such as shipping and cruise ships, is the principal source of low frequency noise in the ocean today, and marine mammals are known to react to or be affected by that noise (Anderwald et al. 2013; Erbe et al. 2014; Foote et al. 2004; Guerra et al. 2014; Hatch and Wright 2007a; Hildebrand 2005; Holt et al. 2008; Kerosky et al. 2013; May-Collado and Quinones-Lebron 2014; Melcon et al. 2012; Richardson et al. 1995a; Williams et al. 2014b). As noted previously, in the inland waters of Puget Sound, Erbe et al. (2012) estimated the maximum annual underwater sound exposure level from vessel traffic near Seattle was 215 dB re 1 µPa2-s and Bassett et al. (2010) measured mean sound pressure levels at Admiralty Inlet from commercial shipping at 117 dB re 1 µPa with a maximum exceeded 135 dB re 1 µPa on some occasions. In contrast, Navy combatant vessels have been designed to generate minimal noise and use ship quieting technology to elude detection by enemy passive acoustic devices (Mintz and Filadelfo 2011a).

Bassett et al. (2012) recorded vessel traffic over a period of just under a year as large vessels passed within 20 km of a hydrophone site located at Admiralty Inlet in Puget Sound. During this period there were 1,363 unique AIS transmitting vessels recorded. In 2014, there were over 5,300 cargo, cruise, or fishing vessels docking at one of the major ports in Puget Sound. In addition to these port calls resulting in approximately 10,600 annual vessel transits, there is the

routine ferry, recreational, and other vessel traffic from commercial activities such as whale watching in the Inland Waters portion of the action area. Because Navy vessels are much fewer in number, they are a small component of overall vessel traffic and vessel noise in most areas where they operate; this is especially the case in the Study Area (see (Mintz and Filadelfo 2011a) concerning a general summary for the U.S. EEZ).

Several studies have demonstrated short-term effects of disturbance on humpback whale behavior (Baker et al. 1983; Bauer and Herman 1986; Hall 1982; Krieger and Wing 1984), but the long-term effects, if any, are unclear or not detectable. Carretta et al. (2001) and Jasny et al. (2005) identified the increasing levels of anthropogenic noise as a habitat concern for whales and other cetaceans because of its potential effect on their ability to communicate. Significant changes in odontocete behavior attributed to vessel noise have been documented up to at least 5.2 kilometers away from the vessel (Pirotta et al. 2012).

Erbé (2002) recorded underwater noise of whale-watching boats in the popular killer whale-watching region of southern British Columbia and northwestern Washington State. Source levels ranged from 145 to 169 dB re 1 µPa-m and increased as the vessel's speed increased. Based on sound propagation models, she concluded that the noise of fast boats would be audible to killer whales over 16 km, would mask killer whale calls over 14 km, would elicit behavioral response over 200 m, and would cause a temporary threshold shifts of 5 dB within 450 meters after 30 to 50 minutes of exposure. She concluded that boats cruising at slow speeds would be audible and would cause masking at 1 km, would elicit behavioral responses at 50 m, and would result in temporary threshold shifts at 20 m.

(SMRU Consulting North America 2017) used computer simulations to calculate noise exposures to Southern Resident killer whales from whale watch boats and commercial vessels. Ranges of prey detection by killer whales were found to decrease by 12 to 37 percent due to cumulative noise impacts from whale watch and commercial vessels. Click masking and killer whale behavioral responses from vessel noise resulted in a total of 4.9 to 5.5 hours of lost foraging time per day.

Galli et al. (2003) measured ambient noise levels and source levels of whale-watch boats in Haro Strait. They measured ambient noise levels of 91 dB (at frequencies between 50 and 20,000 Hz) on extremely calm days (corresponding to sea states of zero) and 116 dB on the roughest day on which they took measures (corresponding to a sea state of ~5). Mean sound spectra from acoustic moorings set off Cape Flattery, Washington, showed that close ships dominated the sound field below 10 kilohertz while rain and drizzle were the dominant sound sources above 20 kilohertz. At these sites, shipping noise dominated the sound field about 10 to 30 percent of the time but the amount of shipping noise declined as weather conditions deteriorated. The large ships they measured produced source levels that averaged 184 dB-m ± 4 dB, which was similar to the 187 dB at 1 meter reported by Greene (1995). The engines associated with the boats in their study produced sounds in the 0.5 to 8.0 kilohertz range at source levels comparable to those of killer whale vocalizations. They concluded that those boats in their study that travelled at their highest

speeds proximate to killer whales could make enough noise to make hearing difficult for the whales.

In addition to the disturbance associated with the presence of vessel, the vessel traffic affects the acoustic ecology of Southern Resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of Southern Resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15 percent during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2009) reported that Southern Resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales needs to increase the duration of their vocalization to avoid masking by the boat noise.

Commercial shipping traffic is a major source of low frequency (5 to 500 Hz) human generated sound in the world's oceans (NRC 2003; Simmonds and Hutchinson 1996). The radiated noise spectrum of merchant ships ranges from 20 to 500 Hz and peaks at approximately 60 Hz. Ross (1976) estimated that between 1950 and 1975 shipping had caused a rise in ambient ocean noise levels of 10 dB; based on his estimates, Ross predicted a continuously increasing trend in ocean ambient noise of 0.55 dB per year. Chapman and Price (2011) recorded low frequency deep ocean ambient noise in the Northeast Pacific Ocean from 1976 to 1986 and reported that the trend of 0.55 dB per year predicted by Ross (1976) persisted until at least around 1980; afterward, the increase per year was significantly less, about 0.2 dB per year. Within the action area identified in this Opinion, the vessel sound inside the western half of the Strait of Juan de Fuca and off the Washington coast comes from cargo ships (86 percent), tankers (6 percent), and tugs (5 percent) (NMFS 2008e citing Mintz and Filadelfo 2004a, 2004b)). Williams et al. (2014a) measured ocean noise levels at 12 sites in the Canadian Pacific Ocean, including Haro Strait, and reported that noise levels were high enough to reduce the communication spaces for fin, humpback and killer whales under typical (median) conditions by 1, 52 and 62 percent, respectively, and 30, 94 and 97 percent under noisy conditions.

Bassett et al. (2012) paired one year of AIS data with hydrophone recordings in Puget Sound's Admiralty Inlet to assess ambient noise levels and the contribution of vessel noise to these levels. Results suggested ambient noise levels between 20 Hz and 30 kilohertz were largely driven by vessel activity and that the increases associated with vessel traffic were biologically significant. Throughout the year, at least one AIS-transmitting vessel was within the study area 90 percent of the time and multiple vessels were present 68 percent of the time. A vessel noise budget showed cargo vessels accounted for 79 percent of acoustic energy, while passenger ferries and tugs had lower source levels but spent substantially more time in the study site and contributed 18 percent of the energy in the budget. All vessels generated acoustic energy at frequencies relevant to all marine mammal functional hearing groups.

In the Salish Sea, Southern Resident killer whales are highly exposed to noise from vessels such as ferries, tugboats, vehicle carriers, recreational vessels, containers, and bulkers (Cominelli et al. 2018). MacGillivray et al. (2019) found that containerships, cruise vessels, vehicle carriers, tankers, and bulkers slowing to 11 knots in the Haro Strait, British Columbia was an effective way to reduce the mean broadband source levels of the noise they produced. Implementing such measures in the Salish Sea has been shown to reduce the amount of time lost to Southern Resident killer whales that could have been spent foraging (Joy et al. 2019a).

Humpback whales and killer whales also interact with a large and growing number of small commercial and recreational vessels that partake in whale watching in the Salish Sea. These vessels often approach close to cetaceans and thus pose risk via collision, marine noise and pollution. The primary management tool for mitigating impacts from whale watch vessels is minimum distance regulations. Mandatory marine mammal distance regulations) imposed in the Salish Sea have varied from 100 to 400 m, depending on year, species, jurisdiction and vessel type (Fraser et al. 2020). Fraser et al. (2020) examined commercial and recreational small vessel compliance with viewing distances across two seasons (June–September 2018 and 2019) during over 400 hours of on-water observation. They found that overall vessel compliance was nearly 80 percent, recreational boats were significantly more likely to violate distance regulations, and boaters were more likely to be non-compliant around killer whales.

Urick (1983a) provided a discussion of the ambient noise spectrum expected in the deep ocean. Shipping, seismic activity, and weather are primary causes of deep-water ambient noise. Noise levels between 20 and 500 Hz appear to be dominated by distant shipping noise that usually exceeds wind-related noise. Above 300 Hz, the level of wind-related noise might exceed shipping noise. Wind, wave, and precipitation noise originating close to the point of measurement dominate frequencies from 500 to 50,000 Hz. The ambient noise frequency spectrum and level can be predicted fairly accurately for most deep-water areas based primarily on known shipping traffic density and wind state (wind speed, Beaufort wind force, or sea state) (Urick 1983a). For frequencies between 100 and 500 Hz, Urick (1983a) has estimated the average deep water ambient noise spectra to be 73 to 80 dB for areas of heavy shipping traffic and high sea states, and 46 to 58 dB for light shipping and calm seas.

In contrast to deep water, ambient noise levels in shallow waters (i.e., coastal areas, bays, harbors, etc.) are subject to wide variations in level and frequency depending on time and location. The primary sources of noise include distant shipping and industrial activities, wind and waves, and marine animals (Urick 1983a). At any given time and place, the ambient noise level is a mixture of these noise types. In addition, sound propagation is also affected by the variable shallow water conditions, including the depth, bottom slope, and type of bottom. Where the bottom is reflective, the sound levels tend to be higher than when the bottom is absorptive.

Although the long-term effects of behavioral responses from vessel interactions are not well known, it is well documented that resident killer whales respond to vessels engaged in close proximity with short-term behavioral changes (Bain et al. 2006b; Foote et al. 2004; Kriete 2002; Kruse 1991; Lusseau et al. 2009; Noren et al. 2009; Senigaglia et al. 2016; Wieland et al. 2010;

Williams et al. 2002b; Williams et al. 2009a; Williams et al. 2014b; Williams et al. 2006a; Williams et al. 2002c). Previous research results indicate that short-term behavioral changes observed in killer whales can occur with high boat densities and at varying distances of the vessels, ranging from vessel approaches within 100 meters (109.4 yards) to vessels at 400 meters (437.4 yards) or greater distances (Bain et al. 2006b; Giles and Cendak 2010; Lusseau et al. 2009; Noren et al. 2009; Williams et al. 2009a; Williams et al. 2002c). These observed behavioral changes have included faster swimming speeds (Williams et al. 2002c), less directed swimming paths (Bain et al. 2006b; Williams et al. 2009a; Williams et al. 2002c), and less time foraging (Bain et al. 2006b; Giles and Cendak 2010; Lusseau et al. 2009; Senigaglia et al. 2016; Williams et al. 2006a). Vessels in the path of the whales can also interfere with important social behaviors such as prey sharing (Ford and Ellis 2006) or nursing (Kriete 2007). With the disruption of feeding behavior that has been observed, it is estimated that the presence of vessels could result in an 18 percent decrease in energy intake, a consequence that could have a significant negative effect on an already prey-limited species (Lusseau et al. 2009; Williams et al. 2006a). In addition to the behavioral changes resulting from the presence of vessels, noise from vessels may mask sounds produced by marine mammals, including echolocation used to locate prey and other signals the whales rely on for communication and navigation.

7.12.4 Aircraft Noise

Sound generated in air is transmitted to water primarily in a narrow area directly below the source. A sound wave propagating from any source must enter the water at an angle of incidence of about 13 degrees or less from the vertical for the wave to continue propagating under the water's surface. At greater angles of incidence, the water surface acts as an effective reflector of the sound wave and allows very little penetration of the wave below the water (Urick 1983b). Water depth and bottom conditions strongly influence how the sound from airborne sources propagate underwater. At lower altitudes, sound levels reaching the water surface would be higher, but the transmission area would be smaller. As the sound source gains altitude, sound reaching the water surface diminishes, but the possible transmission area increases.

Noise generated by fixed-wing aircraft is transient in nature and extremely variable in intensity. Most fixed-wing aircraft sorties (a flight mission made by an individual aircraft) would occur above 3,000 feet Air combat maneuver altitudes generally range from 5,000 to 30,000 ft., and typical airspeeds range from very low (less than 100 knots) to high subsonic (less than 600 knots). Sound exposure levels at the sea surface from most air combat maneuver overflights are expected to be less than 85 A-weighted decibels based on an F/A-18 aircraft flying at an altitude of 5,000 feet and at a subsonic airspeed (400 knots) (U.S. Department of the Navy 2009b).

An intense but infrequent type of aircraft noise is the sonic boom, produced when an aircraft exceeds the speed of sound. Several factors that influence sonic booms include weight, size, and shape of aircraft or vehicle; altitude; flight paths; and atmospheric conditions. A larger and heavier aircraft must displace more air and create more lift to sustain flight, compared with small, light aircraft. Therefore, larger aircraft create sonic booms that are stronger than those of smaller, lighter aircraft. Consequently, the larger and heavier the aircraft, the stronger the shock

waves (U.S. Department of the Navy and Department of Defense 2007). Aircraft maneuvers that result in changes to acceleration, flight path angle, or heading can also affect the strength of a boom. In general, an increase in flight path angle (lifting the aircraft's noise) will diffuse a boom while a decrease (lowering the aircraft's nose) will focus it. In addition, acceleration will focus a boom while deceleration will weaken it. Any change in horizontal direction will focus a boom, causing two or more wave fronts that originated from the aircraft at different times to coincide exactly (U.S. Department of the Navy 2001). Atmospheric conditions such as wind speed and direction and air temperature and pressure can also influence the sound propagation of a sonic boom.

Of all the factors influencing sonic booms, increasing altitude is the most effective method of reducing sonic boom intensity. The width of the boom "carpet" or area exposed to sonic boom beneath an aircraft is about 1 mi. for each 1,000 feet of altitude. For example, an aircraft flying supersonic, straight and level at 50,000 feet can produce a sonic boom carpet about 50 mi. wide. The sonic boom, however, would not be uniform, and its intensity at the water surface would decrease with greater aircraft altitude. Maximum intensity is directly beneath the aircraft and decreases as the lateral distance from the flight path increases until shock waves refract away from the ground or water surface and the sonic boom attenuates. The lateral spreading of the sonic boom depends only on altitude, speed, and the atmosphere and is independent of the vehicle's shape, size, and weight. The ratio of the aircraft length to maximum cross-sectional area also influences the intensity of the sonic boom. The longer and more slender the aircraft, the weaker the shock waves. The wider and more blunt the aircraft, the stronger the shock waves can be (U.S. Department of the Navy and Department of Defense 2007).

In air, the energy from a sonic boom is concentrated in the frequency range from 0.1 to 100 Hz. The underwater sound field due to transmitted sonic boom waveforms is primarily composed of low-frequency components (Sparrow 2002a), and frequencies greater than 20 Hz have been found to be difficult to observe at depths greater than 33 feet (10 meters) (Sohn et al. 2000b).

Smultea et al. (2008) documented a recognized "stress behavioral reaction" by a group of sperm whales in response to small aircraft fly-bys. The group ceased forward movement, moved closer together in a parallel flank-to-flank formation, and formed a fan-shaped semi-circle with the lone calf remaining near the middle of the group. Section 8 discusses additional documented reactions of marine mammals to aircraft noise, as well as those of sea turtles and fishes.

7.13 Commercial and Private Marine Mammal Watching

Vessels (both commercial and private) engaged in marine mammal watching also have the potential to impact whales in the action area. A recent study of whale watch activities worldwide has found that the business of viewing whales and dolphins in their natural habitat has grown rapidly over the past decade into a billion dollar ($US) industry involving over 80 countries and territories and over 9 million participants (Hoyt 2001). In 1988, the Center for Marine Conservation and the NMFS sponsored a workshop to review and evaluate whale watching programs and management needs (CMC and NMFS 1988). That workshop produced several

recommendations for addressing potential harassment of marine mammals during wildlife viewing activities that include developing regulations to restrict operating thrill craft near cetaceans, swimming and diving with the animals, and feeding cetaceans in the wild.

Since then, NMFS has promulgated regulations at 50 CFR §224.103 that specifically prohibit: (1) the negligent or intentional operation of an aircraft or vessel, or the doing of any other negligent or intentional act which results in disturbing or molesting a marine mammal; (2) feeding or attempting to feed a marine mammal in the wild; and (3) approaching humpback whales in Hawaii and Alaska waters closer than 100 yards (91.4 meters). In addition, NMFS launched an education and outreach campaign to provide commercial operators and the general public with responsible marine mammal viewing guidelines which in part state that viewers should: (1) remain at least 50 yards (46 meters) from dolphins, porpoise, seals, sea lions and sea turtles and 100 yards (91 meters) from large whales; (2) limit observation time to 30 minutes; (3) never encircle, chase or entrap animals with boats; (4) place boat engine in neutral if approached by a wild marine mammal; (5) leave the water if approached while swimming; and (6) never feed wild marine mammals. In January 2002, NMFS also published an official policy on human interactions with wild marine mammals which states that: "NOAA Fisheries cannot support, condone, approve or authorize activities that involve closely approaching, interacting or attempting to interact with whales, dolphins, porpoises, seals or sea lions in the wild. This includes attempting to swim with, pet, touch or elicit a reaction from the animals."

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, marine mammal watching is not without potential negative impacts. One concern is that animals may become more vulnerable to vessel strikes once they habituate to vessel traffic. Another concern is that preferred habitats may be abandoned if disturbance levels are too high.

The number and proximity of vessels, particularly whale-watch vessels in the areas occupied by Southern Resident killer whales, represents a source of chronic disturbance for this population. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Cotton 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Several investigators have studied the effects of whale watch vessels on marine mammals (Amaral and Carlson 2005a; Au and Green 2000; Christiansen et al. 2013; Christiansen et al. 2011; Corkeron 1995; Erbe 2002; Felix 2001; Magalhaes et al. 2002; May-Collado and Quinones-Lebron 2014; Richter et al. 2006; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002a). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel

noise, and the number of vessels. Responses changed with these different variables and, in some circumstances, the whales or dolphins did not respond to the vessels, but in other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

To reduce behavioral and acoustic disturbance and risk of vessel strikes, NMFS implemented regulations in 2011 to prohibit vessels from approaching killer whales within 200 yards (182.9 meters), and from parking in the path of the whales within 400 yards (365.8 meters) in inland waters of Washington State (76 FR 20870, April, 14, 2011). In 2015, NMFS adopted the voluntary best practices program, Whale SENSE, in Alaska as a tool to work with whale watch operators to reduce disturbance to whales and minimize the risk of vessel strike. Holt et al. (2017) found that noise levels decreased over two of the three years examined following the implementation of these regulations, but the authors acknowledged that differences in noise levels were not explained by regulation implementation nor by average vessel distance to the animals. An assessment comparing education and outreach efforts, enforcement, vessel compliance, biological effectiveness, and economic impacts for five years pre- and post-implementation found that compliance with the regulations is improving among vessel operators but instances of non-compliance continue to occur (Mongillo et al. 2018). Vessels continue to approach Southern Resident killer whales too closely, impacting their foraging activities. Mongillo et al. (2018) recommended continued and expanded enforcement, determining what vessels have the highest impact on Southern Resident killer whales, and improving estimates of the energetic cost of reduced foraging opportunities.

7.14 Ongoing Military Training and Testing Activities

Ongoing U.S. Navy training and testing activities in the action area are discussed here as part of the environmental baseline. The Navy has conducted training and testing activities and other military readiness activities in the Northwest Training Range Complex in the past, and these activities are ongoing and are expected to continue into the future. During training, existing and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Activities include routine gunnery, missile, surface fire support, amphibious landing, bombing, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. The Navy performs testing activities to ensure that its military forces have the latest technologies and techniques available to them. The majority of the training and testing activities the Navy conducts in the action area are similar, if not identical, to activities that have been occurring in the same locations for decades.

The Navy categorizes training exercises and testing activities into functional warfare areas called primary mission areas. Training exercises fall into the following eight primary mission areas: Anti-air warfare; Strike warfare; Anti-submarine warfare; Mine warfare; Anti-surface warfare; Electronic warfare; and Naval special warfare. Details regarding each warfare area can be found above in the *Description of the Proposed Action* (Section 3) and in the Final Environmental

Impact Statement/Overseas Environmental Impact Statement (FEIS/OEIS) for Northwest Training and Testing (Navy 2020f).

Navy activities produce sound and visual disturbances to marine mammals and sea turtles throughout the action area. Impacts from harassment due to Navy activities include changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures. Sound produced during Navy training and testing activities also results in instances of TTS and PTS to marine mammals and sea turtles. The Navy training and testing activities constitute a federal action and take of ESA-listed marine mammals and sea turtles considered for these Navy activities have previously undergone section 7 consultations. Through these consultations with NMFS, the Navy has implemented monitoring and conservation measures to reduce the potential effects of underwater sound from military training and testing activities on ESA- protected resources in the NWTT action area. Conservation measures include employing visual observers and implementing mitigation zones when training and testing using active sonar or explosives.

7.14.1 Summary of Northwest Training and Testing Phase II Biological Opinion

On November 9, 2015, we completed a section 7 consultation on the Navy's NWTT activities occurring from November 2015 through November 2020 (Phase II). The activities considered in this, and other prior Navy consultations (Phase I) were similar to those proposed for NWTT Phase III that are the subject of this consultation and included the use of active sonar, explosives, and vessels. Each of these opinions concluded that the Navy's proposed actions would not jeopardize the continued existence of threatened or endangered species or destroy or adversely modify designated critical habitat. We also consulted with the NMFS Permits Division, pursuant to section 7 of the ESA, on the issuance of the proposed rule and draft LOAs under section 101(a)(5)(A) of the MMPA (NMFS 2020e) for NWTT activities. The following levels of marine mammal incidental take from acoustic stressors (i.e., sonar and explosives) in the form of behavioral harassment and/or TTS were authorized for NWTT Phase II training exercises, by species:

- Blue whale: up to 5 per year;
- Fin whale: up to 25 per year;
- Humpback whale: up to 12 per year;
- Southern Resident killer whale: up to 2 per year;
- Sperm whale: up to 81 per year;
- Guadalupe fur seal: up to 7 per year;
- Leatherback sea turtle; up to 15 per year (NMFS 2015c).

The following levels of marine mammal incidental take from acoustic stressors in the form of behavioral harassment and/or TTS were authorized for NWTT Phase II testing activities, by species:

- Blue whale: up to 6 per year;

- Fin whale: up to 36 per year;
- Humpback whale: up to 45 per year;
- Sei whale: up to 2 per year;
- Sperm whale: up to 78 per year;
- Guadalupe fur seal: up to 3 per year;
- Leatherback sea turtle; up to 19 per year (NMFS 2015c).

No marine mammal or sea turtle take in the form of harm (i.e., PTS or injury) from acoustic stressors was authorized for NWTT Phase II activities. Similarly, no marine mammal or sea turtle take (i.e., injury or mortality) from vessel strike was authorized for NWTT Phase II.

The ITS for the Phase II opinion also included anticipated take (in the form of injury and mortality) of ESA-listed salmonids, eulachon and rockfish resulting from the Navy's use of explosives in the NWTT action area (NMFS 2015c).

7.15 Invasive Species

Introduction of invasive species is considered one of primary threats to ESA-listed species (Anttila et al. 1998; Pimentel et al. 2004; Wilcove and Chen 1998). Clavero and Garcia-Bertro (2005) found that invasive species were a contributing cause to over half of the extinct species in the International Union for Conservation of Nature (IUCN) database for which an extinction cause could be determined (for 75 percent of extinct species a specific cause could not be determined); invasive species were the only cited cause in 20 percent of those cases. Invasive species consistently rank as one of the top threats to the world's oceans (Pughiuc 2010; Raaymakers 2003; Raaymakers and Hilliard 2002; Terdalkar et al. 2005; Wambiji et al. 2007). A variety of vectors are thought to be responsible for introducing aquatic non-native species including, but not limited to, aquarium and pet trades, recreation, ballast water discharges from ocean-going ships, and hull fouling. Common impacts of invasive species are alteration of habitat and nutrient availability, as well as altering species composition and diversity within an ecosystem (Strayer 2010).

Invasive or nonindigenous species are an emerging threat to biogenic habitat in Puget Sound. Invasive species that have established in Puget Sound include the following: cordgrass, Japanese eelgrass, *Sargassum*, purple varnish/mahogany clam, gallo mussel, Atlantic/Eastern oyster drill, Japanese oyster drill, Asian mudsnail, Bamboo worm, seven species of tunicate, red swamp crayfish, and Eurasian water milfoil. However, the degree to which invasives influence ESA-listed fish species in Puget Sound is not understood (Drake et al. 2010).

7.16 Parasites and/or Disease

Cetaceans have evolved with a group of parasites belonging to the genus *Crassicauda* (order Spirurida) (Lambertsen 1992a). Infections with these nematodes are endemic in both the toothed and baleen whales. Such infections are a major cause of disease of the urinary, respiratory and digestive systems. Of several known crassicaudid infections, those caused by *Crassicauda boopis* are especially pathogenic. This giant worm infects blue whales, humpback whales, and

fin whales (Lambertsen 1992a). Jauniaux et al. (2000) reported evidence for morbillivirus infection in the two fin whales stranded on the Belgian and French coastlines.

Mass mortality events of marine mammals, including cetaceans, have been reported more frequently since 1978, with virus, bacteria, and parasites commonly listed as the cause (Gulland and Hall 2007). Morbillivirus was reported in a neonate female sperm whale that stranded and died in Oahu; the individual was also infected with the bacterial genus *Brucella* (West et al. 2015). In 1987, 14 humpback whales died in Cape Cod Bay, Massachusetts, after consuming mackerel containing a dinoflagellate toxin (Geraci et al. 1989). Acute toxicity events may result in marine mammal mass mortalities; repeated exposure to lower level contaminants may result in immune suppression and/or endocrine disruption (Atkinson et al. 2008). Pinnipeds may become exposed to infectious diseases (e.g., Chlamydia and leptospirosis) through polluted waterways (Aguirre et al. 2007). Infectious diseases are recognized as a significant threat to Hawaiian monk seals. In addition to polluted runoff water, other avenues for exposure include contact with other animals such as marine mammals, and domestic and feral animals (NMFS 2016i).

Fibropapillomatosis (FP) is a neoplastic disease that can negatively impact ESA-listed sea turtle populations. FP has long been present in sea turtle populations with the earliest recorded mention from the late 1800s in the Florida Keys (Hargrove et al. 2016). FP has been reported in every species of marine turtle but is of greatest concern in green turtles, the only known species where this disease has reached a panzootic status (Williams Jr et al. 1994). Historical data indicate that the disease rose in prevalence most noticeably in the 1980s. Prevalence rates as high as 45 to 50 percent have been reported within some local green turtle populations (Hargrove et al. 2016; Jones et al. 2015). FP primarily affects medium-sized immature turtles in coastal foraging pastures.

Most young fish are highly susceptible to disease during the first two months of life. The cumulative mortality in young animals can reach 90 to 95 percent. Although fish disease organisms occur naturally in the water, native fish have co-evolved with them. Fish can carry these diseases at less than lethal levels (Foott et al. 2003; Kier Associates 1991; Walker and Foott 1993). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Guillen 2003; Spence et al. 1996). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (Spence et al. 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). Examples of parasites and disease for salmonids include whirling disease, infectious hematopoietic necrosis (IHN), sea-lice (e.g. *Lepeophtheirus salmonis*, various *Caligus* species *Henneguya salminicola*, or Ich (*Ichthyopthirius multifiliis)* and Columnaris *(Flavobacterium columnare*)).

Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The

disease occurs in the wild and in hatcheries and results in losses to fry and fingerling salmonids, especially rainbow trout. The disease is transmitted by infected fish, fish parts and birds.

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. This disease affects rainbow/steelhead, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook salmon, sockeye salmon, chum salmon, and coho salmon. The virus is triggered by low water temperatures and is shed in the feces, urine, sexual fluids, and external mucus of salmonids. Transmission is mainly from fish to fish, primarily by direct contact and through the water.

Sea lice is a marine ectoparasite found in coastal waters that can also cause deadly infestations of farm-grown salmon and may affect wild salmon. *Henneguya salminicola,* a protozoan parasite, is commonly found in the flesh of salmonids, particularly in British Columbia. The fish responds by walling off the parasitic infection into a number of cysts that contain milky fluid. This fluid is an accumulation of a large number of parasites. Fish with the longest freshwater residence time as juveniles have the most noticeable infection. The order of prevalence for infection is coho salmon followed by sockeye salmon, Chinook salmon, chum salmon, and pink salmon. The *Henneguya* infestation does not appear to cause disease in the host salmon – even heavily infected fish tend to return to spawn successful.

Additionally, ich (a protozoan) and Columnaris (a bacterium) are two common fish diseases that were implicated in the massive kill of adult salmon in the Lower Klamath River in September 2002 (CDFG 2003; Guillen 2003).

7.17 Scientific Research and Permits

Many of the ESA-listed species in this opinion are the subject of scientific research and monitoring activities. The impacts of these research activities pose both benefits and risks. In the short term, take may occur in the course of scientific research. However, these activities have a great potential to benefit ESA-listed species in the long-term. Most importantly, the information gained during research and monitoring activities will assist in planning for the recovery of listed species. Information obtained from scientific research is essential for understanding the status of ESA-listed species, obtaining specified critical biological information, and achieving species recovery goals.

Research on ESA-listed species is granted an exemption to the ESA take prohibitions of section 9 through the issuance of section 10(a)(1)(A) permits. Research activities authorized through scientific research permits can produce various stressors on wild and captive animals resulting from capture, handling, and research procedures. The ESA requires that research conducted under a section 10(a)(1)(A) research permit cannot operate to the disadvantage of the species. Scientific research permits issued by NMFS are conditioned with mitigation measures to ensure that the impacts of research activities on target and non-target ESA-listed species are as minimal as possible.

Over time, NMFS has issued dozens of permits on an annual basis for various forms of "take" of marine mammals, sea turtles, and ESA-listed fish species in the action area from a variety of research activities. Authorized research on ESA-listed marine mammals includes close vessel and aerial approaches, photographic identification, photogrammetry, biopsy sampling, tagging, ultrasound, exposure to acoustic activities, breath sampling, behavioral observations, passive acoustic recording, and underwater observation. Only non-lethal "takes" of marine mammals are authorized for research activities. In 2019, NMFS concluded section 7 consultation on a Program for the Issuance of Permits for Research and Enhancement Activities on Threatened and Endangered Cetaceans pursuant to Section 10(a) of the ESA and Section 104 of the MMPA (NMFS 2019b). According to this programmatic, 466 cetacean (all species) takes due to biopsies and tagging were reported within the research program from 2009 to 2017. In addition, 6,192 takes were reported for all cetacean species due to harassment from vessel surveys. The number of researchers and their proposed research and enhancement activities remain relatively consistent over time; and thus, the frequency of these research and enhancement activities are not expected to significantly change in the foreseeable future (NMFS 2019b). Between 2009 and 2020, 26 takes of Guadalupe fur seals were reported to the Permits Division. These included takes from vessel survey disturbance, sample collection, capture, morphometrics, and tagging. There are currently 20 active permits authorizing research using the aforementioned methods on all or a subset of the ESA-listed marine mammals in the NWTT action area.

ESA-listed sea turtle research includes approach, capture, handling, restraint, tagging, biopsy, blood or tissue sampling, lavage, ultrasound, imaging, antibiotic (tetracycline) injections, laparoscopy, captive experiments, and mortality. Most authorized take is sub-lethal as mortality is rarely authorized by NMFS in sea turtle research permits. On average, from 2007 to 2017 approximately 988 sea turtle (all species) takes were reported within the research program throughout the U.S. in any given year. In 2017, NMFS concluded section 7 consultation on a Program for the Issuance of Permits for Research and Enhancement Activities on Threatened and Endangered Sea Turtles Pursuant to Section 10(a) of the ESA. This programmatic consultation allows for the authorization of up to two leatherback sea turtle mortalities within the Pacific Ocean basin every ten years (NMFS 2017c). This programmatic also includes an ITS that allows for one green sturgeon Southern DPS lethal take every ten years.

Although not identified as a factor for decline or a threat preventing recovery, scientific research and monitoring activities have the potential to affect the species' survival and recovery by killing listed salmonids—whether intentionally or not (National Marine Fisheries Service 2019). For the year 2019, NMFS has issued numerous research section 10(a)(1)(A) scientific research permits allowing lethal and non-lethal take of ESA-listed fish species, along with the state scientific research programs under ESA section 4(d) and tribal 4(d) research. Table 100 shows the total take for the ongoing research authorized under ESA sections 4(d) and 10(a)(1)(A). In general, actual take levels associated with research activities are almost always considerably lower than the permitted levels.

Table 100. Total expected take of west coast ESA-listed fish species for scientific research and monitoring authorized for 2019.

Species	Life Stage	Origin[a]	Total Take	Percent of ESU/DPS taken	Lethal Take	Percent of ESU/DPS killed
CC Chinook salmon	Adult	Natural	1,028	14.6147%	36	0.5118%
	Juvenile	Natural	307,003	24.0207%	3,932	0.3076%
CVS Chinook salmon	Adult	LHAC	740	32.5561%	256	11.2626%
		Natural	701	18.8087%	26	0.6976%
	Juvenile	LHAC	18,542	0.8547%	3,033	0.1398%
		Natural	873,548	112.6470%	16,830	2.1703%
LCR Chinook salmon[b]	Adult	LHAC	88	0.2280%	13	0.0337%
		LHIA	0		0	
		Natural	289	0.9807%	10	0.0339%
	Juvenile	LHAC	62,096	0.1890%	1,326	0.0040%
		LHIA	410	0.0383%	34	0.0032%
		Natural	819,245	6.9095%	11,323	0.0955%
PS Chinook salmon[b]	Adult	LHAC	1,620	16.3997%	125	0.8814%
		LHIA	929		12	
		Natural	958	4.2772%	39	0.1741%
	Juvenile	LHAC	138,699	0.3821%	10,580	0.0291%
		LHIA	40,527	0.5574%	2,513	0.0346%
		Natural	475,765	15.6745%	9,981	0.3288%
SacR winter-run Chinook salmon	Adult	LHAC	197	8.8262%	53	2.3746%
		Natural	275	130.9524%	15	7.1429%
	Juvenile	LHAC	11,552	5.7760%	1,471	0.7355%
		Natural	175,523	89.8487%	5,075	2.5978%
SnkR fall-run Chinook salmon	Adult	LHAC	241	2.1085%	15	0.1312%
		LHIA	202	1.6313%	3	0.0242%
		Natural	240	2.5408%	12	0.1270%
	Juvenile	LHAC	951	0.0361%	46	0.0017%
		LHIA	565	0.0198%	23	0.0008%
		Natural	2,303	0.3519%	89	0.0136%
SnkR spr/sum-run Chinook salmon	Adult	LHAC	1,029	43.1085%	19	0.7960%
		LHIA	618	146.7933%	7	1.6627%
		Natural	1,404	10.9705%	27	0.2110%
	Juvenile	LHAC	23,573	0.4952%	384	0.0081%
		LHIA	12,756	1.4684%	156	0.0180%
		Natural	519,711	40.0813%	6,149	0.4742%
UCR spring-run Chinook salmon	Adult	LHAC	12	0.1927%	4	0.0642%
		LHIA	0	0.0000%	0	0.0000%
		Natural	110	3.8301%	4	0.1393%
	Juvenile	LHAC	352	0.0577%	10	0.0016%
		LHIA	107	0.0336%	19	0.0060%
		Natural	2,057	0.4325%	57	0.0120%
UWR Chinook salmon[b]	Adult	LHAC	133	0.4225%	18	0.0572%
		LHIA	0		0	
		Natural	167	1.6368%	11	0.1078%
	Juvenile	LHAC	8,551	0.1641%	151	0.0029%
		LHIA	34	21.6561%	4	2.5478%
		Natural	42,666	3.3446%	822	0.0644%

Species	Life Stage	Origin[a]	Total Take	Percent of ESU/DPS taken	Lethal Take	Percent of ESU/DPS killed
CR chum salmon	Adult	LHIA	0	0.0000%	0	0.0000%
		Natural	14	0.1315%	4	0.0376%
	Juvenile	LHIA	550	0.0849%	6	0.0009%
		Natural	38,194	0.6553%	464	0.0080%
HCS chum salmon	Adult	LHIA	0	0.0000%	0	0.0000%
		Natural	1,999	5.1658%	32	0.0827%
	Juvenile	LHIA	135	0.0900%	3	0.0020%
		Natural	729,353	12.3059%	2,560	0.0432%
CCC coho salmon	Adult	LHAC	0	0.0000%	0	0.0000%
		Natural	2,126	110.0414%	38	1.9669%
	Juvenile	LHAC	25,486	15.3641%	858	0.5172%
		Natural	168,286	106.4226%	3,233	2.0445%
LCR coho salmon[b]	Adult	LHAC	339	3.8562%	38	0.4323%
		LHIA	0		0	
		Natural	601	2.0123%	12	0.0402%
	Juvenile	LHAC	54,058	0.7662%	990	0.0140%
		LHIA	627	0.2184%	11	0.0038%
		Natural	175,566	26.9530%	2,448	0.3758%
OC coho salmon	Adult	LHAC	10	1.7889%	4	0.7156%
		Natural	9,110	9.6586%	105	0.1113%
	Juvenile	LHAC	99	0.1650%	3	0.0050%
		Natural	556,873	8.3847%	12,228	0.1841%
SONCC coho salmon[b]	Adult	LHAC	590	19.2702%	14	0.2744%
		LHIA	1,517		16	
		Natural	1,510	16.6575%	28	0.3089%
	Juvenile	LHAC	8,551	4.2755%	108	0.0540%
		LHIA	266	0.0463%	266	0.0463%
		Natural	171,792	8.5316%	2,377	0.1180%
OL sockeye salmon[c]	Adult	LHAC	5	0.4567%	0	0.0794%
		LHIA	0		0	
		Natural	18		4	
	Juvenile	LHAC	31	0.0678%	3	0.0066%
		LHIA	0	0.0000%	0	0.0000%
		Natural	86	0.0083%	4	0.0004%
SnkR sockeye salmon	Adult	LHAC	3	0.0749%	0	0.0000%
		Natural	89	16.3004%	6	1.0989%
	Juvenile	LHAC	397	0.1934%	255	0.1242%
		Natural	7,539	38.0661%	401	2.0247%
LCR steelhead[b]	Adult	LHAC	71	0.3184%	4	0.0179%
		LHIA	0		0	
		Natural	2,147	16.6176%	22	0.1703%
	Juvenile	LHAC	40,731	3.2908%	584	0.0472%
		LHIA	0	0.0000%	0	0.0000%
		Natural	67,070	20.0867%	1,138	0.3408%
MCR steelhead	Adult	LHAC	920	155.4054%	12	2.0270%
		LHIA	39	26.3514%	1	0.6757%
		Natural	1,067	16.0066%	13	0.1950%
	Juvenile	LHAC	3,381	0.8456%	48	0.0120%

Species	Life Stage	Origin[a]	Total Take	Percent of ESU/DPS taken	Lethal Take	Percent of ESU/DPS killed
		LHIA	4,269	4.1933%	60	0.0589%
		Natural	119,023	28.6555%	2,408	0.5797%
PS steelhead[c]	Adult	LHAC	21	7.6788%	6	0.1916%
		LHIA	17		0	
		Natural	1,445		31	
	Juvenile	LHAC	4,247	3.8609%	86	0.0782%
		LHIA	835	0.7422%	16	0.0142%
		Natural	65,211	2.9683%	1,205	0.0548%
SnkR basin steelhead	Adult	LHAC	2,951	2.1247%	49	0.0353%
		LHIA	2,201	7.8086%	37	0.1313%
		Natural	8,381	45.4920%	109	0.5917%
	Juvenile	LHAC	28,003	0.8399%	358	0.0107%
		LHIA	22,323	3.4518%	247	0.0382%
		Natural	217,540	26.6142%	2,834	0.3467%
UCR steelhead	Adult	LHAC	9	0.0821%	2	0.0182%
		LHIA	0	0.0000%	0	0.0000%
		Natural	105	2.6329%	2	0.0502%
	Juvenile	LHAC	9,474	1.4293%	203	0.0306%
		LHIA	2,204	1.5298%	54	0.0375%
		Natural	17,727	10.4819%	361	0.2135%
UWR steelhead	Adult	Natural	220	7.5549%	6	0.2060%
	Juvenile	Natural	8,602	5.9778%	192	0.1334%
S eulachon[d]	Adult	Natural	34,070	0.1831%	30,909	0.1663%
	Juvenile	Natural	350		350	
S green sturgeon	Adult	Natural	181	4.1258%	5	0.1140%
	Subadult	Natural	39	0.3528%	1	0.0090%
	Juvenile	Natural	1,611	76.4957%	111	5.2707%
PS/GB bocaccio[d]	Adult	Natural	25	1.3678%	14	0.5862%
	Juvenile	Natural	38		13	
PS/GB yelloweye rockfish[d]	Adult	Natural	75	0.1119%	34	0.0731%
	Juvenile	Natural	64		15	

[a] LHAC=Listed Hatchery Adipose Clipped, LHIA = Listed Hatchery Intact Adipose.
[b] Abundances for adult hatchery salmonids are LHAC and LHIA combined.
[c] Abundances for all adult components are combined.
[d] Abundance for these species are only known for the adult life stage which is used to represent the entire DPS.

7.18 Impact of the Baseline on ESA-listed Resources

Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed resources considered in this consultation. Some of these stressors result in mortality or serious injury to individual animals (e.g., vessel strike, whaling, entanglement in fishing gear), whereas others result in more indirect (e.g., a fishery that impacts prey availability) or non-lethal impacts (e.g., whale watching). Assessing the aggregate impacts of these stressors on species is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this opinion are wide ranging and subject to stressors in locations throughout the action area and outside the NWTT action area.

We consider the best indicator of the aggregate impact of the *Environmental Baseline* on ESA-listed resources to be the status and trends of those species. As noted in Section 6, some of the

species considered in this consultation are experiencing increases in population abundance, some are declining, and for others, their status remains unknown. Taken together, this indicates that the *Environmental Baseline* is impacting species in different ways. The species experiencing increasing population abundances are doing so despite the potential negative impacts of the *Environmental Baseline*. Therefore, while the *Environmental Baseline* may slow their recovery, recovery is not being prevented. For the species that may be declining in abundance, it is possible that the suite of conditions described in the *Environmental Baseline* is preventing their recovery. However, it is also possible that their populations are at such low levels (e.g., due to historic commercial whaling) that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, species may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each species is discussed in Section 6.2 above (*Status of Species Likely Affected by the Proposed Action*).

8 EFFECTS OF THE ACTION

Section 7 regulations define "effects of the action" as all consequences to ESA-listed species or designated critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 C.F.R. §402.02). Section 7 regulations (50 C.F.R. §402.17) elaborate on this definition as follows:

- *Activities that are reasonably certain to occur* - A conclusion of reasonably certain to occur must be based on clear and substantial information, using the best scientific and commercial data available. Factors to consider when evaluating whether activities caused by the proposed action (but not part of the proposed action) or activities reviewed under cumulative effects are reasonably certain to occur include, but are not limited to: (1) Past experiences with activities that have resulted from actions that are similar in scope, nature, and magnitude to the proposed action; (2) Existing plans for the activity; and (3) Any remaining economic, administrative, and legal requirements necessary for the activity to go forward.

- *Consequences caused by the proposed action* - To be considered an effect of a proposed action, a consequence must be caused by the proposed action (i.e., the consequence would not occur but for the proposed action and is reasonably certain to occur). A conclusion of reasonably certain to occur must be based on clear and substantial information, using the best scientific and commercial data available. Considerations for determining that a consequence to the species or critical habitat is not caused by the proposed action include, but are not limited to: (1) The consequence is so remote in time from the action under consultation that it is not reasonably certain to occur; or (2) The consequence is so geographically remote from the immediate area involved in the action that it is not reasonably certain to occur; or (3) The consequence is only reached through a lengthy causal chain that involves so many steps as to make the consequence not reasonably certain to occur.

The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 C.F.R. §402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

The destruction and adverse modification analysis considers whether the action produces "a direct or indirect alteration that appreciably diminished the value of critical habitat as a whole for the conservation of a listed species" (50 C.F.R. §402.02).

Previously in Section 5, we identified the potential stressors created by the Navy's testing and training activities. This section begins with a summary table of our effects determinations by stressor category for each ESA-listed species considered during this consultation (Table 101). This serves as a cross reference for the sections to follow that provide the analyses supporting these effects determinations. This table also lists the overall effects determination for each species. Recall that at the start of Section 6, we provided a complete list of ESA-listed species and designated critical habitat that may be affected by the proposed action. Further, in Section 6.1 we explained that some ESA-listed species were not likely to be adversely affected by any of the stressors associated with the proposed action (i.e., overall determination labeled as "NLAA" in Table 101). This is because any effects on these species were extremely unlikely to occur such that they were discountable, or the size or severity of the impact was so low as to be insignificant, including those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. The ESA-listed species addressed in Section 6.1 are included in the summary table below because this table reflects all species considered during consultation. However, ESA-listed species determined in Section 6.1 to not likely be adversely affected by any of the stressors associated with the proposed action are not discussed further in this section of the opinion. There is no meaningful potential for the proposed action to affect the survival or recovery of these species (i.e., no meaningful potential to jeopardize the species).

In this section, we focus on those species that are *likely to be adversely affected* by one or more stressors created by the proposed action. This section is organized by taxa (i.e., marine mammals, sea turtles, and fish) since the species within each taxa often respond to stressors in similar ways.

In Section 8.1, we discuss the stressors associated with the proposed action that we determined are *not likely to adversely affect* all species from a particular taxa (i.e., stressors labeled as "NLAA" in Table 101). We do not carry these stressors forward in our effects analysis since there is no meaningful potential for these stressors to affect the survival or recovery of species within the particular taxa. Finally, in Section 8.2, we summarize the analysis for the stressor and taxa (i.e., marine mammals, sea turtles, and fish) combinations that are likely to result in adverse effects to species within the taxa (labeled as "LAA" in Table 101). Cells marked as 'NE' in Table 101 indicate that we anticipate the stressor would have "no effect" on the species; these stressors are not included in our effects analysis for those particular species as there is no meaningful potential for these stressors to affect the survival or recovery of any species in the action area.

In Section 8.3 we discuss potential impacts of the proposed action on the designated (or proposed) critical habitat within the action area.

Table 101. NMFS ESA effects determinations by stressor and overall effects determination for each ESA-listed species (LAA = likely to adversely affect; NLAA = not likely to adversely affect; NE = no effect).

		Acoustic Stressors				*Explosive Stressors*	*Energy Stressors*		*Physical Disturbance and Strike Stressors*			*Entanglement Stressors*		*Ingestion Stressors*		
ESA-Listed Species	***Overall Determination***	***Sonar & Other Transducers***	***Vessel Noise***	***Aircraft Noise***	***Weapons Noise***	***Explosions***	***In-Water Electromagnetic Devices***	***High Energy Lasers***	***Vessels & In-Water Devices***	***Military Expended Material***	***Seafloor Devices***	***Wires & Cables***	***Decelerators/Parachutes***	***Military Expended Materials - Munitions***	***Military Expended Materials – Other than Munitions***	***Secondary Stressors***
Marine Mammals																
Blue whale	LAA	LAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Fin whale	LAA	LAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Gray whale – Western North Pacific DPS	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale – Central America DPS	LAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale – Mexico DPS	LAA	LAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	*Overall Determination*	*Acoustic Stressors*				*Explosive Stressors*	*Energy Stressors*		*Physical Disturbance and Strike Stressors*			*Entanglement Stressors*		*Ingestion Stressors*		*Secondary Stressors*
		Sonar & Other Transducers	*Vessel Noise*	*Aircraft Noise*	*Weapons Noise*	*Explosions*	*In-Water Electromagnetic Devices*	*High Energy Lasers*	*Vessels & In-Water Devices*	*Military Expended Material*	*Seafloor Devices*	*Wires & Cables*	*Decelerators/Parachutes*	*Military Expended Materials - Munitions*	*Military Expended Materials – Other than Munitions*	
Killer whale – Southern Resident killer whales DPS	LAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	LAA
North Pacific Right Whale	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sei whale	LAA	LAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sperm whale	LAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Guadalupe fur seal	LAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steller sea lion – Western DPS	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Marine Reptiles																
Leatherback sea turtle	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	*Overall Determination*	*Acoustic Stressors*				*Explosive Stressors*	*Energy Stressors*		*Physical Disturbance and Strike Stressors*			*Entanglement Stressors*		*Ingestion Stressors*		*Secondary Stressors*
		Sonar & Other Transducers	*Vessel Noise*	*Aircraft Noise*	*Weapons Noise*	*Explosions*	*In-Water Electromagnetic Devices*	*High Energy Lasers*	*Vessels & In-Water Devices*	*Military Expended Material*	*Seafloor Devices*	*Wires & Cables*	*Decelerators/Parachutes*	*Military Expended Materials - Munitions*	*Military Expended Materials – Other than Munitions*	
Fish																
Bocaccio – Puget Sound/Georgia Basin DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – California Coastal ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Central Valley Spring-Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Lower Columbia River ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Puget Sound ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	Overall Determination	Acoustic Stressors				Explosive Stressors	Energy Stressors		Physical Disturbance and Strike Stressors			Entanglement Stressors		Ingestion Stressors		Secondary Stressors
		Sonar & Other Transducers	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	High Energy Lasers	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators/Parachutes	Military Expended Materials - Munitions	Military Expended Materials – Other than Munitions	
Chinook salmon – Sacramento River Winter-Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Snake River Fall-Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Snake River Spring/Summer Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Upper Columbia River Spring-Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Upper Willamette River ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

		Acoustic Stressors				*Explosive Stressors*	*Energy Stressors*		*Physical Disturbance and Strike Stressors*			*Entanglement Stressors*		*Ingestion Stressors*		
ESA-Listed Species	***Overall Determination***	***Sonar & Other Transducers***	***Vessel Noise***	***Aircraft Noise***	***Weapons Noise***	***Explosions***	***In-Water Electromagnetic Devices***	***High Energy Lasers***	***Vessels & In-Water Devices***	***Military Expended Material***	***Seafloor Devices***	***Wires & Cables***	***Decelerators/Parachutes***	***Military Expended Materials - Munitions***	***Military Expended Materials – Other than Munitions***	***Secondary Stressors***
Chum salmon – Columbia River ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chum salmon – Hood Canal Summer-Run ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Central California Coast ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Lower Columbia River ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Oregon Coast ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	Overall Determination	Acoustic Stressors				Explosive Stressors	Energy Stressors		Physical Disturbance and Strike Stressors			Entanglement Stressors		Ingestion Stressors		Secondary Stressors
		Sonar & Other Transducers	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	High Energy Lasers	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators/Parachutes	Military Expended Materials - Munitions	Military Expended Materials – Other than Munitions	
Coho salmon – Southern Oregon and Northern California Coasts ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Eulachon – Southern DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Green Sturgeon – Southern DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sockeye salmon – Ozette Lake ESU	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sockeye salmon – Snake River ESU	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – California Central Valley DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

		Acoustic Stressors				*Explosive Stressors*	*Energy Stressors*		*Physical Disturbance and Strike Stressors*			*Entanglement Stressors*		*Ingestion Stressors*		
ESA-Listed Species	***Overall Determination***	***Sonar & Other Transducers***	***Vessel Noise***	***Aircraft Noise***	***Weapons Noise***	***Explosions***	***In-Water Electromagnetic Devices***	***High Energy Lasers***	***Vessels & In-Water Devices***	***Military Expended Material***	***Seafloor Devices***	***Wires & Cables***	***Decelerators/Parachutes***	***Military Expended Materials - Munitions***	***Military Expended Materials – Other than Munitions***	***Secondary Stressors***
Steelhead – Central California Coast DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Lower Columbia River DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Middle Columbia River DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Northern California DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Puget Sound DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	Overall Determination	Acoustic Stressors				Explosive Stressors	Energy Stressors		Physical Disturbance and Strike Stressors			Entanglement Stressors		Ingestion Stressors		Secondary Stressors
		Sonar & Other Transducers	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	High Energy Lasers	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators/Parachutes	Military Expended Materials - Munitions	Military Expended Materials – Other than Munitions	
Steelhead – Snake River Basin DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – South-Central California Coast DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Southern California DPS	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Upper Columbia River DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Upper Willamette River DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Species	Overall Determination	Acoustic Stressors				Explosive Stressors	Energy Stressors		Physical Disturbance and Strike Stressors			Entanglement Stressors		Ingestion Stressors		Secondary Stressors
		Sonar & Other Transducers	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	High Energy Lasers	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators/Parachutes	Military Expended Materials - Munitions	Military Expended Materials – Other than Munitions	
Yelloweye Rockfish – Puget Sound/Georgia Basin DPS	LAA	NLAA	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

8.1 Stressors Not Likely to Adversely Affect ESA-listed Species

Our analysis of the stressors associated with the proposed action led to the determination that some stressors are not likely to adversely affect some or all ESA-listed species because the effect of that stressor would be insignificant or discountable. The following section discusses stressors that are not likely to adversely affect some or all ESA-listed species considered in this opinion. For analysis of effects to ESA-listed species, note that discussion in this section is organized by taxa (i.e., marine mammals, sea turtles, fishes) because the pathways for effects for these stressors is generally the same by taxa and, in most cases, we would not expect different effects at the species level. While there is variation among species within each taxa, the species within each taxa share many similar life history patterns and other factors (e.g., morphology) which make them similarly vulnerable (or not) to the stressors created by the proposed action.

8.1.1 Marine Mammals

We determined that several of the acoustic stressors, all of the energy stressors, entanglement stressors, ingestion stressors, and potential secondary stressors are not likely to adversely affect ESA-listed blue whales, fin whales, Central America DPS humpback whales, Mexico DPS humpback whales, Southern Resident killer whales DPS killer whales, sei whales, sperm whales, or Guadalupe fur seals. Our analysis for these stressors and marine mammals is summarized below.

8.1.1.1 Acoustic Stressors – Marine Mammals

The discussion below focuses on a subset of the acoustic stressors associated with the proposed action. We determined that these acoustic stressors are not likely to adversely affect ESA-listed marine mammals. Additional discussion of the acoustic stressors associated with the proposed action is included in Section 5.1 above. The effects of additional acoustic stressors, which we determined are likely to adversely affect marine mammals, are discussed in Sections 8.2.1.

8.1.1.1.1 Effects of Vessel Noise on Marine Mammals

Numerous studies of interactions between surface vessels and cetaceans have demonstrated that cetaceans engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two. However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of cetaceans to surface vessels are similar to their behavioral responses to predators. Based on studies of cetacean behavior to vessel approaches, several variables determine whether cetaceans are likely to be disturbed by surface vessels. The behavioral repertoire cetaceans have used to avoid interactions with surface vessels appears to depend on the number of vessels in their perceptual field (the area within which animals detect acoustic, visual, or other cues) and the animal's assessment of the risks associated with those vessels (the primary index of risk is

probably vessel proximity relative to the animal's flight initiation distance) (Sims et al. 2012). Below a threshold number of vessels (which probably varies from one species to another, although groups of cetaceans probably share sets of patterns), studies have shown that whales will attempt to avoid an interaction using horizontal avoidance behavior. Above that threshold, studies have shown that cetaceans will tend to avoid interactions using vertical avoidance behavior, although some marine mammals will combine horizontal avoidance behavior with vertical avoidance behavior (Bryant et al. 1984; David 2002; Kruse 1991; Lusseau 2003; Nowacek et al. 2001; Stensland and Berggren 2007; Williams and Ashe 2007);

The distance between vessels and cetaceans when the animal perceives that an approach has started and during the course of the interaction can affect whether cetaceans are likely to be disturbed by surface vessels (Au and Perryman 1982; David 2002; Hewitt 1985; Kruse 1991; Lundquist et al. 2012; Lusseau 2003; Tseng et al. 2011). Cetaceans are also more likely to respond to an approaching vessel when the vessel stays on a single or predictable path (Acevedo 1991; Angradi et al. 1993; Browning and Harland. 1999; Lusseau 2003; Lusseau 2006; Williams et al. 2002a) than when it engages in frequent course changes (Evans et al. 1994; Lusseau 2006; Williams et al. 2002a). Other studies have shown that noise associated with the vessel (particularly engine noise) and the rate at which the engine noise increases (which the animal may treat as evidence of the vessel's speed) affect how cetaceans react to vessels (David 2002; Lusseau 2003; Lusseau 2006; Polagye et al. 2011). The behavioral state of individual cetaceans at the time of a vessel approach can also determine whether or how the animal will react (David 2002; Lusseau 2003; Lusseau 2006; Wursig et al. 1998). For example, Würsig et al. (1998) concluded that whales were more likely to engage in avoidance responses when the whales were milling or resting than during other behavioral states.

Most of the investigations reported that animals tended to reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies (Corkeron 1995; Lundquist et al. 2012; Lusseau 2003; Lusseau 2004; Nowacek et al. 2001; Van Parijs and Corkeron 2001; Williams et al. 2002a; Williams et al. 2002d). In the process, their dive times increased, vocalizations and jumping were reduced (with the exception of beaked whales), individuals in groups move closer together, swimming speeds increased, and their direction of travel took them away from the source of disturbance (Baker and Herman 1989; Edds and Macfarlane 1987; Evans et al. 1992; Kruse 1991). Some individuals also dove and remained motionless, waiting until the vessel moved past their location. Most animals finding themselves in confined spaces, such as shallow bays, during vessel approaches tended to move towards more open, deeper waters (Kruse 1991). We assume that this movement would give them greater opportunities to avoid or evade vessels as conditions warranted. Williams et al. (2014b) found that there was a 50 percent chance of killer whales, exhibiting, at minimum, minor changes in respiration following exposure to broadband ship noise at received levels of around 130 dB re 1 µPa root-mean-square.

Although most of these studies focused on small cetaceans (for example, bottlenose dolphins, spinner dolphins, spotted dolphins, harbor porpoises, beluga whales, and killer whales) and less information is available for pelagic and deep-diving marine mammals (reviewed in Erbe et al. 2019), studies of large whales have reported similar results for fin and sperm whales (David 2002). Baker et al. (1983) reported that humpbacks in Hawaii responded to vessels at distances of two to four kilometers. Richardson et al. (1985a) reported that bowhead whales (*Balaena mysticetus)* swam in the opposite direction of approaching seismic vessels at distances between one and four kilometers and engage in evasive behavior at distances under one kilometer. Fin whales also responded to vessels at a distance of about one kilometer (Edds and Macfarlane 1987).

In short-term studies, researchers have noted changes in resting and surface behavior states of cetaceans to whale watching vessels (Aguilar Soto et al. 2006; Arcangeli and Crosti 2009; Au and Green 2000; Christiansen et al. 2010; Erbe 2002; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams and Noren 2009). Noren et al. (2009) conducted research in the San Juan Islands in 2005 and 2006 and their findings suggested that close approaches by vessels impacted the whales' behavior and that the whale-watching guideline minimum approach distance of 100 meters may be insufficient in preventing behavioral responses. Most studies of this type are opportunistic and have only examined the short-term response to vessel sound and vessel traffic (Magalhaes et al. 2002; Noren et al. 2009; Richardson and Wursig 1995; Watkins 1981c). Fin whales may alter their swimming patterns by increasing speed and heading away from a vessel, as well as changing their breathing patterns in response to a vessel approach (Jahoda et al. 2003). Vessels that remained 328 feet (100 meters) or farther from fin and humpback whales were largely ignored in one study where whale watching activities are common (Watkins 1981a). Only when vessels approached more closely did the fin whales in this study alter their behavior by increasing time at the surface and exhibiting avoidance behaviors. Other studies have shown when vessels are near, some but not all fin whales change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au and Green 2000; Castellote et al. 2012; Williams et al. 2002d).

In the Watkins (1981a) study, humpback whales did not exhibit any avoidance behavior but did exhibit minor behavioral reactions to vessel presence. In a study of regional vessel traffic, Baker et al. (1983) found that when vessels were in the area, the respiration patterns of the humpback whales changed. The whales also exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 1.24 and 2.48 miles (2,000 and 4,000 meters) away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were within approximately 1.2 miles (2,000 m; Baker and Herman 1983). Similar findings were documented for humpback whales when approached by whale watch vessels in Hawaii (Au and Green 2000).

Mother-calf humpback whale pairs in Vava'u, Kingdom of Tonga showed significant vertical avoidance responses to approaching tourism vessels, with mothers diving for significantly longer periods of time in the presence of vessels and human swimmers (Fiori et al. 2019). The greatest number of avoidance responses were observed when vessels employed "J approaches," where the vessel runs parallel to the animal(s), overtakes the animal(s), and then turns in front of the animal(s).

Gende et al. (2011) reported on observations of humpback whales in inland waters of Southeast Alaska subjected to frequent cruise ship transits (i.e., in excess of 400 transits in a 4-month season in 2009). The study was focused on determining if close encounter distance was a function of vessel speed. The reported observations, however, seem in conflict with other reports of avoidance at much greater distance so it may be that humpback whales in those waters are more tolerant of vessels (given their frequency) or are engaged in behaviors, such as feeding, that they are less willing to abandon. This example again highlights that context is critical for predicting and understanding behavioral reactions as concluded by Southall et al. (2007a) and Ellison et al. (2012).

In the presence of approaching vessels, blue whales perform shallower dives accompanied by more frequent surfacing, but otherwise do not exhibit strong reactions (Calambokidis et al. 2009a). Minke whales in the Antarctic did not show any apparent response to a survey vessel moving at normal cruising speeds (about 12 knots) at a distance of 5.5 nm; however, when the vessel drifted or moved at very slow speeds (about one knot), many whales approached it (Leatherwood et al. 1982a).

Although not expected to be in the NWTT action area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004), and therefore might provide insight to behavioral responses of other baleen whales with the same hearing frequencies. North Atlantic right whales continue to use habitats in high vessel traffic areas (Nowacek et al. 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Nowacek et al. 2004; Terhune and Verboom 1999). Although this may minimize potential disturbance from passing ships, it could increase the whales' vulnerability to ship strike.

Using historical records, Watkins (1986) showed that the reactions of four species of mysticetes to vessel traffic and whale watching activities in Cape Cod had changed over the 25-year period examined (1957 through 1982). Reactions of minke whales changed from initially more positive reactions, such as coming towards the boat or research equipment to investigate, to more 'uninterested' reactions towards the end of the study. Fin whales, the most numerous species in the area, showed a trend from initially more negative reactions, such as swimming away from the boat with limited surfacing, to more uninterested (ignoring) reactions allowing boats to approach within 98.4 feet (30 meters). Right whales showed little change over the study period, with a roughly equal number of reactions judged to be negative and uninterested; no right whales were

noted as having positive reactions to vessels. Humpback whales showed a trend from negative to positive reactions with vessels during the study period. The author concluded that the whales had habituated to the human activities over time (Watkins 1986).

Sperm whales generally react only to vessels approaching within several hundred meters; however, some individuals may display avoidance behavior, such as quick diving (Magalhaes et al. 2002; Wursig et al. 1998). One study showed that after diving, sperm whales showed a reduced timeframe from when they emitted the first click than before vessel interaction (Richter et al. 2006). Small whale-watching and research vessels generate more noise in higher frequency bands and are more likely to approach odontocetes directly, and to spend more time near the individual whale. Reactions to Navy vessels are not well documented, but smaller whale-watching and research boats have been shown to cause these species to alter their breathing intervals and echolocation patterns.

Masking occurs when one sound (i.e., noise), interferes with the detection, discrimination, or recognition of another sound (i.e., signal). The quantitative definition of masking is the amount in decibels an auditory detection or discrimination threshold is raised in the presence of a masker (Erbe et al. 2015). Masking can effectively limit the distance over which a marine mammal can communicate, detect biologically relevant sounds, and echolocate (odontocetes) (Navy 2019c). Masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Masking can lead to vocal changes (e.g., Lombard effect, increasing amplitude, or changing frequency) and behavior changes (e.g., cessation of foraging, leaving an area) to both signalers and receivers, in an attempt to compensate for noise levels (Erbe et al. 2016). For example, (Guazzo et al. 2020) found evidence for the Lombard effect in singing humpback whales, which increased source levels as ambient ocean noise levels increased. Masking is more likely to occur in the presence of broadband, relatively continuous noise sources such as vessels.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity (Holt et al. 2008) as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado and Wartzok 2008). Likewise, modification of multiple vocalization parameters has been shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al. 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al. 2005). Killer whales off the northwestern coast of the U.S. have been observed to increase the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which has been suggested as a response to increased masking noise produced by the vessels (Foote et al. 2004). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. For example, the source level of killer whale vocalizations has been shown to increase with higher background noise levels associated with vessel traffic (Hotchkin and Parks 2013). In addition, calls with a

high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al. 2008).

Mysticetes have been shown to both increase and decrease calling behavior in the presence of vessel noise. An increase in feeding call rates and repetition by humpback whales in Alaskan waters was associated with vessel noise (Doyle et al. 2008). Melcon et al. (2012) documented that blue whales increased the proportion of time spent producing certain types of calls when vessels were present. Conversely, decreases in singing activity by humpback whales have been noted near Brazil due to boat traffic (Sousa-Lima and Clark 2008). Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012) demonstrated that fin whales' songs had shortened duration and decreased bandwidth, center frequency, and peak frequency in the presence of high shipping noise levels. It is not known if these changes in vocal behavior corresponded to other behaviors. Right whales were observed to shift the frequency content of their calls upward while reducing the rate of calling in areas of increased anthropogenic noise (Parks et al. 2007) as well as increasing the amplitude (intensity) of their calls (Parks 2009; Parks et al. 2011). However, Cholewiak et al. (2018) found that right whale gunshot calls had the lowest loss of communication space in Stellwagen National Sanctuary (5 percent), while fin and humpback whales lost up to 99 percent of their communication space with increased ambient noise and shipping noise combined. Although humpback whales off Australia did not change the frequency or duration of their vocalizations in the presence of ship noise, their source levels were lower than expected based on source level changes to wind noise, potentially indicating some signal masking (Dunlop 2016). Increased vessel noise decreased the modeled acoustic communication area of humpback whales off Australia and group social interactions were significantly reduced in number (Dunlop 2019). Clark et al. (2009) estimated the noise from the passage of two vessels could reduce the optimal communication space for North Atlantic right whales by 84 percent (see also Hatch et al. 2012).

The available information, as discussed above, suggests that ESA-listed cetaceans are either not likely to respond to vessel noise or are expected to respond only briefly if exposed to noise from Navy vessels. Expected behavioral responses include brief avoidance behavior, changes in respiration rate, or changes in vocal patterns. Most avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior.

We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. Long-term and cumulative impacts of vessel sound on cetaceans remains largely unknown. For behavioral responses to result in energetic costs that result in long-term harm, such disturbances would likely need to be sustained

for a significant duration or extent where individuals exposed would not be able to select alternate habitat to recover and feed. Given the typical Navy training and testing activities involving vessels are not continuous year round in the action area, we do not expect prolonged vessel noise exposures and preclusion of individuals from feeding, breeding, or sheltering habitat. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect cetacean reactions to vessel noise to have any measurable effects on an individual's fitness and behavioral responses to vessel noise which would result in adverse effects.

Exposure of marine species to vessel noise would be greatest in nearshore areas of highest vessel traffic. Of the marine mammals considered in this opinion, only Southern Resident killer whales are expected to spend large amounts of time in the inland portion of the action area. Humpback whales have recently returned to the Salish Sea (Strait of Juan De Fuca, Puget Sound, Strait of Georgia) where a large increase in sightings was reported for 2014 (Calambokidis et al. 2017b). Although chronic vessel disturbance and noise is considered one of the primary threats to the Southern Resident killer whale population, Navy vessels account for an extremely small proportion (less than 0.1 percent) of the vessel activity in areas where Southern Resident killer whales are found. Adverse effects of stressors resulting from vessel activity on Southern Resident killer whales or humpback whales are more likely associated with the large and growing volume of commercial shipping, whale watching, ferry operations, and recreational vessels in the Puget Sound region. There are a total of 44 Navy vessels (6 destroyers, 14 submarines, 2 aircraft carriers, and 22 security vessels) homeported in the action area that at one time or another will be engaged in training and testing activities. Activities involving Navy vessel movements occur intermittently and are variable in duration. Chronic vessel noise disturbance is extremely unlikely to occur as a result of the proposed action given the small level of Navy vessel activity and the short duration and intermittent nature of this stressor. In addition, many Navy ships incorporate quieting technology (Mintz and Filadelfo 2011b).

Pinniped reactions to vessels are variable and reports include a wide spectrum of possibilities from avoidance and alert, to cases where animals in the water are attracted, and cases on land where there is lack of significant reaction suggesting habituation to or tolerance of vessels (Richardson et al. 1995b). Specific case reports in Richardson et al. (1995b) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007a), pinniped responses to vessels are affected by the context of the situation and by the animal's experience.

Anderwald et al. (2013) investigated grey seal reactions to an increase in vessel traffic off Ireland's coast in association with construction activities, and their data suggests the number of vessels had an indeterminate effect on the seals' presence. Harbor seals haul out on tidewater glaciers in Alaska, and most haulouts occur during pupping season. Blundell & Pendleton (2015) found that the presence of any vessel reduces haulout time, but cruise ships and other large

vessels in particular shorten haulout times. Another study of reactions of harbor seals hauled out on ice to cruise ship approaches in Disenchantment Bay, Alaska, revealed that animals are more likely to flush and enter the water when cruise ships approach within 500 meters and four times more likely when the cruise ship approaches within 100 meters (Jansen et al. 2010). Karpovich et al. (2015) also found that harbor seal heart rates increased when vessels were present during haulout periods, and increased further when vessels approached and animals re-entered the water. Harbor seals responded more to vessels passing by haulout sites in areas with less overall vessel activity, and the model best predicting their flushing behavior included the number of boats, type of boats, and distance to boats. More flushing occurred to non-motorized vessels (e.g., kayaks), likely because they tended to occur in groups rather than as single vessels, and tended to pass closer (25–184 meters) to the haulout sites than motorized vessels (55–591 meters) (Cates and Acevedo-Gutiérrez 2017). Jones et al. (2017) also modeled the spatial overlap of vessel traffic and grey and harbor seals in the United Kingdom, and found most overlap to occur within 50 km of the coast, and high overlap occurring within 5 of 13 grey seal Special Areas of Conservation and within 6 of 12 harbor seal Special Areas of Conservation. They also estimated received levels of shipping noise and found maximum daily M-weighted cumulative SEL values from 170 to 189 dB, with the upper confidence intervals of those estimates sometimes exceeding TTS values. However, there was no evidence of reduced population size in these high overlap areas.

In summary, ESA-listed marine mammals are either not likely to respond to Navy vessel noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns that include, but are not limited to, breeding, feeding or sheltering. Additionally, the effects of any temporary masking specifically from Navy vessels are expected to be of a short duration and not result in meaningful changes to an animals ability to communicate or detect biologically relevant cues given the background noise levels in the action area independent of Navy vessels and the small percentage of vessel traffic Navy vessels represent in the action area (as discussed in Section 5.1.3 above). Therefore, the effects of vessel noise on ESA-listed marine mammals from Navy vessels are considered insignificant, and thus are not likely to cause adverse effects.

8.1.1.1.2 Effects of Aircraft Noise on Marine Mammals

The following paragraphs summarize what is known about the reaction of various marine mammal species to overhead flights of many types of fixed-wing aircraft and helicopters, as well as unmanned aerial vehicles. Additional discussion of aircraft overflight noise as a potential stressor is included in Section 5.1.4. Thorough reviews of the subject and available information are presented in Richardson et al. (1995b) and elsewhere (e.g., Efroymson et al. 2001; Holst et al. 2011; Luksenburg and Parsons 2009; Smith et al. 2016).

Transmission of sound from a moving airborne source to a receptor underwater is influenced by numerous factors, but significant acoustic energy is primarily transmitted into the water directly

below the craft in a narrow cone (Navy 2017b). Underwater sounds from aircraft are strongest just below the surface and directly under the aircraft. Marine mammals may respond to both the physical presence and to the noise generated by aircraft, making it difficult to attribute causation to one or the other stimulus. In addition to noise produced, all low-flying aircraft make shadows, which can cause animals at the surface to react. Helicopters may also produce strong downdrafts, a vertical flow of air that becomes a surface wind, which can also affect an animal's behavior at or near the surface.

The most common responses of cetaceans to overflights were short surfacing durations, abrupt dives, and percussive behavior (breaching and tail slapping; Nowacek et al. 2007). Other behavioral responses such as flushing and fleeing the area of the source of the noise have also been observed (Holst et al. 2011; Manci et al. 1988). Richardson et al. (1995b) noted that marine mammal reactions to aircraft overflights have largely consisted of opportunistic and anecdotal observations lacking clear distinction between reactions potentially caused by the noise of the aircraft and the visual cue an aircraft presents. In addition, it was suggested that variations in the responses noted were generally due to other undocumented factors associated with overflights (Richardson et al. 1995b). These factors could include aircraft type (single engine, multi-engine, jet turbine), flight path (altitude, centered on the animal, off to one side, circling, level and slow), environmental factors (e.g., wind speed, sea state, cloud cover) and locations where native subsistence hunting continues and animals are more sensitive to anthropogenic impacts, including the noise from aircraft. Christiansen et al. (2016) measured the in air and underwater noise levels of two unmanned aerial vehicles. The researchers found that in air the broadband source levels were around 80 dB re 20 µPa, while at a meter underwater received levels were 95 to 100 dB re 1 µPa when the vehicle was only five to ten meters above the surface, and were not quantifiable above ambient noise levels when the vehicle was higher. Therefore, if an animal is near the surface and the unmanned aerial vehicle is flying at a low altitude, it may be detected, but in most cases these vehicles are operated at much higher altitudes (e.g. well over 30 meters) and are not likely to be heard.

The impact of aircraft overflights is one of the least understood sources of potential behavioral response by any species or taxonomic group, and so many generalities must be made based on the little data available. There is some data for each taxonomic group; taken together it appears that in general, cetaceans have varying levels of sensitivity to overflights depending on the species and context. Mysticetes either ignore or occasionally dive in response to aircraft overflights (Koski et al. 1998). Richardson et al. (1985b) and Richardson et al. (1995b) found no evidence that single or occasional aircraft flying above mysticetes causes long-term displacement of these mammals.

Bowhead whales in the Beaufort Sea exhibited a transient behavioral response to fixed-wing aircraft and vessels. Reactions were frequently observed at less than 1,000 feet (304.8 meters) above sea level, infrequently observed at 1,500 feet (457.2 meters), and not observed at all at 2,000 feet (609.6 meters) (Richardson et al. 1985b). Bowhead whales reacted to helicopter

overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 150 meters or higher. The bowheads exhibited fewer behavioral changes than did beluga whales in the same area (Patenaude et al. 2002). It should be noted that bowhead whales in this study may have more acute responses to anthropogenic activity than many other cetaceans because these animals were presented with restricted egress due to limited open water between ice floes. Additionally, these animals are hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

A pilot study was conducted on the use of unmanned aerial vehicles to observe bowhead whales. Flying at altitudes between 120 to 210 meters above the surface, no behavioral responses were observed in any animals (Koski et al. 2015; Koski et al. 1998). Similarly, Christiansen et al. (2016) did not observe any responses to an unmanned aerial vehicle flown 30 to 120 meters above the water when taking photos of humpback whales to conduct photogrammetry and assess fitness. Acevedo-Whitehouse et al. (2010) successfully maneuvered a remote-controlled helicopter over large baleen whales to collect samples of their blows, with no more avoidance behavior than noted for typical photo-identification vessel approaches. Unmanned aircraft are much smaller and quieter than typical aircraft and so are less likely to cause a behavioral response, although they may fly at much lower altitudes (Smith et al. 2016).

During standard marine mammal surveys at an altitude of 750 feet, some sperm whales remained on or near the surface the entire time the aircraft was in the vicinity, while others dove immediately or a few minutes after being sighted. Other authors have corroborated the variability in sperm whales' reactions to fixed-wing aircraft or helicopters (Richter et al. 2006; Smultea et al. 2008; Wursig et al. 1998). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al. 1995b). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 feet) by moving closer together and forming a defensive fan-shaped semicircle, with their heads facing outward. Several individuals in the group turned on their sides, apparently to look up toward the aircraft (Smultea et al. 2008). Whale-watching aircraft (fixed-wing airplanes and helicopters) apparently caused sperm whales to turn more sharply but did not affect blow interval, surface time, time to first click, or the frequency of aerial behavior (Richter et al. 2003).

Low flight altitudes of helicopters during some anti-submarine warfare and mine warfare activities, often under 100 feet, may elicit a somewhat stronger behavioral response due to the proximity to cetaceans, the slower airspeed and therefore potentially longer exposure duration, and the downdraft created by the helicopter's rotor. Cetaceans would likely avoid the area under the helicopter due to the downdraft, noise, and presence of the helicopter. It is unlikely that an individual would be exposed repeatedly for long periods because Navy aircraft typically transit open ocean areas within the action area. The literature cited above indicates that aircraft noise would cause only small temporary, short-term behavioral changes. Specifically, cetaceans at or

near the surface when an aircraft flies overhead at low altitude may startle, divert their attention to the aircraft, or avoid the immediate area by swimming away or diving.

Richardson et al. (1995b) noted that responsiveness to aircraft overflights generally was dependent on the altitude of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). In general pinnipeds are unresponsive to overflights, and may startle, orient towards the sound source or increase vigilance, or may briefly re-enter the water, but typically remain hauled out or immediately return to their haulout location (Blackwell et al. 2004; Gjertz and Børset 1992). Adult females, calves, and juveniles are more likely to enter the water than males, and stampedes resulting in mortality to pups (by separation or crushing) can occur when disturbance is severe, although this is rare (Holst et al. 2011). Responses may also be dependent on the distance of the aircraft. For example, reactions of walruses on land varied in severity and included minor head raising at a distance of 2.5 km, orienting toward or entering the water at less than 150 meters and 1.3 km in altitude, to full flight reactions at horizontal ranges of less than 1 km at altitudes as high as 1,000–1,500 meters (Richardson et al. 1995b).

Helicopters are used in studies of several species of seals hauled out and are considered an effective means of observation (Bester et al. 2002; Gjertz and Børset 1992), although they have been known to elicit behavioral reactions such as fleeing (Hoover 1988). For California sea lions and Steller sea lions at a rocky haulout off Crescent City in northern California, helicopter approaches to landing sites typically caused the most severe response of diving into the water (National Oceanic and Atmospheric Administration 2010). Responses were also dependent on the species, with Steller sea lions being more sensitive and California sea lions more tolerant. Depending on the time between subsequent approaches, animals hauled out in between and fewer animals reacted upon subsequent exposures (National Oceanic and Atmospheric Administration 2010).

Pinniped reactions to rocket launches and overflight at San Nicolas Island were studied from August 2001 to October 2008 (Holst et al. 2011). California sea lions startled and increased vigilance for up to two minutes after a rocket overflight, with some individuals moving down the beach or returning to the water. Northern elephant seals showed little reaction to any overflight. Harbor seals had the most pronounced reactions of the three species observed with most animals within approximately 4 km of the rocket trajectory leaving their haulout sites for the water and not returning for several hours. The authors concluded that the effects of the rocket launches were minor with no effects on local populations evidenced by the growing populations of pinnipeds on San Nicolas Island (Holst et al. 2011).

Pinnipeds may be more sensitive to unmanned aerial systems, especially those flying at low altitudes, due to their possible resemblance to predatorial birds (Smith et al. 2016), which could lead to flushing behavior (Olson 2013). Responses may also vary by species, age class, behavior, and habituation to other anthropogenic noise, as well as by the type, size, and configuration of unmanned aerial vehicle used (Pomeroy et al. 2015). However, in general pinnipeds have

demonstrated little to no response to unmanned aerial systems, with some orienting towards the vehicle, other alerting behavior, or short-term flushing possible (Moreland et al. 2015; Sweeney et al. 2015).

Many of the observations cited in this section are of marine mammal reactions to aircraft flown for whale-watching and cetacean research purposes. Marine mammal survey aircraft are typically used to locate, photograph, track, and sometimes follow animals for long distances or for long periods of time, all of which results in the animal being much more frequently located directly beneath the aircraft (in the cone of the loudest noise and potentially in the shadow of the aircraft) for extended periods. In contrast, Navy aircraft would not follow marine mammals and would not result in prolonged exposure of cetaceans to overhead noise or encroachment.

In most instances, exposure of a marine mammal to fixed-wing aircraft, helicopters, and unmanned aircraft presence and noise would last for only seconds as the aircraft quickly passes overhead. Animals would have to be at or near the surface at the time of an overflight to be exposed to appreciable sound levels. Takeoffs and landings occur at established airfields as well as on vessels at sea at unspecified locations across the action area. Takeoffs and landings from Navy vessels could startle marine mammals. However, these events only produce in-water noise at any given location for a brief period as the aircraft climbs to cruising altitude. Some sonic booms from aircraft could also startle marine mammals, but these events are transient and happen infrequently at any given location within the action area. Repeated exposure to most individuals over short periods (days) is extremely unlikely, except for animals that are resident in inshore areas around Navy ports, on Navy fixed ranges, or during major training exercises. Resident animals could be subjected to multiple overflights per day, though most of the ESA-listed cetaceans considered in this opinion have wide ranging life histories. Additionally, aircraft would pass quickly overhead, typically at altitudes above 3,000 feet, which would make marine mammals unlikely to respond.

In summary, ESA-listed marine mammals are either not likely to respond to Navy aircraft noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns that include, but are not limited to, breeding, feeding or sheltering. Due to the short term and infrequent nature of any exposures, and the brief and inconsequential behavioral responses of animals that could follow such exposure, the effects of aircraft overflight noise from Navy activities on ESA-listed marine mammals is considered insignificant, and thus not likely to cause adverse effects.

8.1.1.1.3 Effects of Weapons Firing, Launch, and Impact Noise on Marine Mammals

Activities using weapons would be conducted as described in Section 3 of this opinion. Additional discussion on weapons noise as a potential stressor is included in Section 5.1.5. The use of weapons during training could occur almost anywhere within the action area. Noise associated with large caliber weapons firing and the impact of non-explosive practice munitions or kinetic weapons would typically occur at locations greater than 12 nautical miles from shore

for safety reasons. Small- and medium-caliber weapons firing could occur throughout the action area.

A gun fired from a ship on the surface of the water propagates a blast wave away from the gun muzzle into the water. Yagla and Stiegler (2003b) found that the average peak sound pressure in the water measured directly below the muzzle of the gun and under the flight path of the shell (assuming it maintains an altitude of only a few meters above the water's surface) was approximately 200 dB re 1 µPa. Animals at the surface of the water, in a narrow footprint under a weapons trajectory, could be exposed to naval gunfire noise and may exhibit brief startle reactions, avoidance, diving, or no reaction at all. Sound due to missile and target launches is typically at a maximum at initiation of the booster rocket and rapidly fades as the missile or target travels downrange. These sounds would be transient and of short duration, lasting no more than a few seconds at any given location. Additionally, due to the short-duration, transient nature of launch noise, cetaceans are unlikely to be exposed multiple times within a short period.

Although missiles are launched from aircraft, they are expected to produce minimal noise in the water due to the altitude of the aircraft at launch. Missiles and targets launched by ships or near the water's surface may expose cetaceans to levels of sound that could produce brief startle reactions, avoidance, or diving. Some objects, such as hyperkinetic projectiles and non-explosive practice munitions, could impact the water with great force and produce a relatively large impulse. Marine mammals within the area may hear the impact of non-explosive ordnance on the surface of the water and would likely alert, startle, dive, or avoid the immediate area.

In summary, ESA-listed marine mammals are either not likely to respond to Navy weapons noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns that include, but are not limited to, breeding, feeding or sheltering. If they do occur, behavioral reactions would likely be short-term (seconds to minutes) and multiple exposures of the same animal over a short duration are not anticipated. For these reasons, the effects of weapons noise from Navy activities on ESA-listed cetaceans are considered insignificant, and thus not likely to cause adverse effects.

8.1.1.1.4 Effects of Underwater Explosives from De Minimis Sources

In addition to the explosives quantitatively analyzed for impacts to ESA-listed species (shown in Table 20 above), the Navy uses some very small impulsive sources (less than 0.1 pounds NEW), categorized in bin E0, that were not quantitatively analyzed by the Navy for potential exposure to ESA-listed species. Quantitative modeling in multiple locations of the area impacted by these explosives has indicated that these sources have a very small zone of influence. As such, it is extremely unlikely that ESA-listed cetaceans would be exposed to explosives in bin E0. Therefore, potential effects from explosives in bin E0 on ESA-listed marine mammals are discountable.

8.1.1.1.5 Effects of Explosions in Air on Marine Mammals

Explosions that occur during air warfare would typically be at sufficient altitude that a large portion of the sound refracts upward due to cooling temperatures with increased altitude and would not reach the water's surface where ESA-listed species could occur. There may also be sound that is audible near the surface, although sound transmission would also be limited by reflection at the air-water boundary, depending on the angle of incidence. Marine mammals within the audible range of sound from explosions in air may exhibit a behavioral startle response but are expected to quickly return to normal behavior. Due to the short duration and sporadic nature of explosions in the air, and the extremely low likelihood of an ESA-listed marine mammal being within close enough proximity to detect sounds from such explosions, we do not expect any temporary behavioral responses to result in a significant disruption of breeding, feeding, or sheltering of individual animals. Therefore, the effects of sound from explosions in air during Navy activities on ESA-listed marine mammals are considered insignificant, and thus not likely to cause adverse effects.

8.1.1.1.6 Effects of Nitrogen Decompression and Acoustically-induced Bubble Formation due to Sonar Exposures

In this section we discuss two potential effects resulting from exposure to Navy sonar in the action area that we determined are not likely to adversely affect ESA-listed marine mammals. These are nitrogen decompression and bubble formation that may occur in blood and other tissue of an animal exposed to this stressor. In Section 8.2, we discuss all other effects resulting from Navy sonar exposure that are likely to adversely affect ESA-listed marine mammals in the action area.

Nitrogen Decompression

Marine mammals are thought to deal with nitrogen loads in their blood and other tissues, caused by gas exchange from the lungs under conditions of high ambient pressure during diving, through anatomical, behavioral, and physiological adaptations (Hooker et al. 2012). Although not a direct injury, variations in marine mammal diving behavior or avoidance responses could result in nitrogen off-gassing in super-saturated tissues, possibly to the point of deleterious vascular and tissue bubble formation (Hooker et al. 2012; Jepson et al. 2003; Saunders et al. 2008) with resulting symptoms similar to decompression sickness (also known as "the bends" in humans).

The process has been under debate in the scientific community (Hooker et al. 2012; Saunders et al. 2008), although analyses of by-caught and drowned animals has demonstrated that nitrogen bubble formation can occur once animals are brought to the surface and tissues are supersaturated with nitrogen (Bernaldo De Quiros et al. 2013; Moore et al. 2009). Deep diving whales, such as beaked whales (not listed under the ESA), normally have higher nitrogen loads in body tissues, which may make them more susceptible to decompression for certain modeled

changes in dive behavior (Fahlman et al. 2014b; Fernandez et al. 2005a; Hooker et al. 2012; Jepson et al. 2003).

Researchers have examined how dive behavior affects tissue supersaturation conditions that could put an animal at risk of gas bubble embolism. An early hypothesis was that if exposure to a startling sound elicits a rapid ascent to the surface, tissue gas saturation sufficient for the evolution of nitrogen bubbles might result (Fernandez et al. 2005a; Jepson et al. 2003). However, modeling suggested that even unrealistically rapid rates of ascent from normal dive behaviors are unlikely to result in supersaturation to the extent that bubble formation would be expected in beaked whales (Zimmer and Tyack 2007). Instead, emboli observed in animals exposed to mid-frequency active sonar (Fernandez et al. 2005a; Jepson et al. 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth of lung collapse (Hooker et al. 2012; Tyack et al. 2006; Zimmer and Tyack 2007). Longer times spent diving at mid-depths above lung collapse would allow gas exchange from the lungs to continue under high hydrostatic pressure conditions, increasing potential for supersaturation; below the depth of lung collapse, gas exchange from the lungs to the blood would likely not occur (Fahlman et al. 2014b). However, Costidis and Rommel (2016) suggest that gas exchange may continue to occur across the tissues of air-filled sinuses in deep-diving odontocetes (e.g., sperm whales) below the depth of lung collapse if hydrostatic pressures are high enough to drive gas exchange across into non-capillary veins, contributing to tissue gas loads. To examine the potential for gas bubble formation, a bottlenose dolphin was trained to dive repetitively to depths shallower than lung collapse to elevate nitrogen saturation to the point that asymptomatic nitrogen bubble formation was predicted to occur. However, inspection of the vascular system of the dolphin via ultrasound did not demonstrate the formation of any nitrogen gas bubbles (Houser et al. 2009). To estimate risk of decompression sickness, Kvadsheim (2012) modeled gas exchange in the tissues of sperm, pilot, killer, and beaked whales based on actual dive behavior during exposure to sonar in the wild. Results indicated that venous supersaturation was within the normal range for these species, which have naturally high levels of nitrogen loading. Researchers have also considered the role of accumulation of carbon dioxide produced during periods of high activity by an animal, theorizing that accumulating carbon dioxide, which cannot be removed by gas exchange below the depth of lung collapse, may facilitate the formation of bubbles in nitrogen saturated tissues (Bernaldo De Quiros et al. 2012; Fahlman et al. 2014b). Garcia Parraga et al. (2018) suggest that diving marine mammals have physiological and anatomical adaptations to control gas uptake above the depth of lung collapse, favoring oxygen uptake while minimizing nitrogen uptake. Under the hypothesis of Garcia Parraga et al. (2018), elevated activity due to a strong evasive response could lead to increased uptake of nitrogen, resulting in an increased risk of nitrogen decompression.

Modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of long-halftime tissues (i.e., tissues that take longer to give

off nitrogen, e.g., fat and bone lipid) to the point that they are supersaturated when the animals are at the surface (Fahlman et al. 2014b; Hooker et al. 2009; Saunders et al. 2008). The presence of osteonecrosis (bone death due to reduced blood flow) in deep diving sperm whales has been offered as evidence of chronic supersaturation (Moore and Early 2004). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al. 2006; Hooker et al. 2009), while the condition of supersaturation required for bubble formation in these tissues has been demonstrated in marine mammals drowned at depth as fisheries bycatch and brought to the surface (Moore et al. 2009). For beaked whale strandings associated with sonar use, one theory is that observed bubble formation may be caused by long periods of compromised blood flow caused by the stranding itself (which reduces ability to remove nitrogen from tissues) following rapid ascent dive behavior that does not allow for typical management of nitrogen in supersaturated, long-halftime tissues (Houser et al. 2009).

A fat embolic syndrome (out of place fat particles, typically in the bloodstream) was identified by Fernandez et al. (2005b) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream.

Dennison et al. (2011) reported on investigations of dolphins stranded in 2009 to 2010 and, using ultrasound, identified gas bubbles in kidneys of 21 of the 22 live-stranded dolphins and in the liver of two of the 22. The authors postulated that stranded animals are unable to recompress by diving, and thus may retain bubbles that are otherwise re-absorbed in animals that can continue to dive. The researchers concluded that the minor bubble formation observed could be tolerated since the majority of stranded dolphins released did not re-strand.

The appearance of extensive bubble and fat emboli in beaked whales (not listed under the ESA) is unique to strandings associated with certain high intensity sonar events. The phenomenon has not been observed in other stranded cetaceans, nor has it been observed in beaked whale strandings not associated with sonar use. It is not clear whether there is some mechanism for this phenomenon specific to beaked whales or whether the phenomenon occurs only following rapidly occurring stranding events (i.e., when whales are not capable of sufficiently decompressing). Because of the lack of evidence for extensive nitrogen bubble formation while diving, NMFS believes that the likelihood of ESA-listed cetaceans getting "the bends" following sonar acoustic exposure is extremely low. Therefore, the likelihood of effects from nitrogen decompression in ESA-listed marine mammals from Navy sonar as a result of the proposed action is extremely unlikely and thus discountable.

Acoustically-induced Bubble Formation Due to Sonars

A suggested cause of injury to cetaceans is rectified diffusion (Crum and Mao 1996), the process of increasing the size of a microscopic gas bubble by exposing it to a sound field. The process is

dependent upon a number of factors including the SPL and duration of exposure. Under this hypothesis, microscopic bubbles assumed to exist in the tissues of marine mammals may experience one of three things: (1) bubbles grow to the extent that tissue hemorrhage (injury) occurs, (2) bubbles develop to the extent that an immune response is triggered or the nervous tissue is subjected to enough localized pressure that pain or dysfunction occurs (a stress response without injury), or (3) the bubbles are cleared by the lungs without negative consequence to the animal.

Rectified diffusion is facilitated if the environment in which the ensonified bubbles exist is supersaturated with gas. As discussed above, repetitive diving by cetaceans can cause the blood and some tissues to become supersaturated (Ridgway and Howard 1979). The dive patterns of some cetaceans (e.g., non-ESA listed beaked whales) are predicted to induce greater supersaturation (Houser et al. 2001). If rectified diffusion were possible in cetaceans exposed to high-level sound, conditions of tissue supersaturation could theoretically speed the rate and increase the size of bubble growth. Subsequent effects due to tissue trauma and emboli would presumably mirror those observed in humans suffering from decompression sickness.

It is unlikely that the short duration of sonar pulses would be long enough to drive bubble growth to any substantial size, if such a phenomenon occurs. However, an alternative but related hypothesis, suggested by Crum et al. (2005), is that stable microbubbles could be destabilized by high-level sound exposures such that bubble growth then occurs through static diffusion of gas out of supersaturated tissues. In such a scenario, the cetacean would need to be in a gas-supersaturated state for a long enough time for bubbles to reach a problematic size. The phenomena of bubble growth due to a destabilizing exposure was shown by Crum et al. (2005) by exposing highly supersaturated ex vivo bovine tissues to a 37 kilohertz source at 214 dB re 1 µPa. Although bubble growth occurred under the extreme conditions created for the study, these conditions would not likely exist in the wild because the levels of tissue supersaturation in the study (as high as 400–700 percent) are substantially higher than model predictions for cetaceans (Fahlman et al. 2009; Fahlman et al. 2014b; Houser et al. 2001; Saunders et al. 2008). In addition, such high exposure levels would only occur in very close proximity to the most powerful Navy sonars. With the proposed Navy mitigation measures in place, it is improbable that this mechanism is responsible for stranding events or traumas associated with beaked whale strandings. For these reasons we believe that ESA-listed cetacean injury resulting from acoustically induced bubble formation during Navy NWTT activities to be extremely unlikely, and therefore discountable.

8.1.1.1.7 Effects of Other Acoustic Sources Not Quantitatively Analyzed

Several of the acoustic sources associated with NWTT activities were not quantitatively analyzed in terms of their effects on ESA-listed species. These include the following: broadband sound sources; Doppler sonar; fathometers; hand-held sonar; imaging sonar; high-frequency acoustic modems; tracking pingers; acoustic releases; and side-scan sonars (see Table 19) above for

details). When used during routine training and testing activities, and in a typical environment, these sources fall into one or more of the following categories:

- Transmit primarily above 200 kilohertz: Sources above 200 kilohertz are above the hearing range of the ESA-listed species in the action area.
- Source levels of 160 dB re 1 µPa or less: Low-powered sources with source levels less than 160 dB re 1 µPa are typically hand-held sonars, range pingers, transponders, and acoustic communication devices. Assuming spherical spreading for a 160 dB re 1 µPa source, the sound will attenuate to less than 140 dB re 1 µPa within ten meters and less than 120 dB re 1 µPa within 100 meters of the source. Ranges would be even shorter for a source less than 160 dB re 1 µPa source level. As discussed above (Section 2.2.2), we assume that marine mammals would not exhibit a behavioral response when exposed to such low source levels.
- Acoustic source classes listed in Table 19: Sources with operational characteristics, such as short pulse length, narrow beam width, downward-directed beam, and low energy release, or manner of system operation, which exclude the possibility of any significant impact on an ESA-listed species (actual source parameters are classified). Even if there is a possibility that some species may be exposed to and detect some of these sources, any response is expected to be short term and insignificant.

Therefore, these acoustic sources associated with NWTT activities (as described in Table 19) would either have no effect on ESA-listed marine mammals, or the effects would be insignificant and thus unlikely to cause an adverse effect, depending on the particular source considered.

8.1.1.2 Energy Stressors – Marine Mammals

This section summarizes the effects of energy stressors used during Navy training and testing activities on marine mammals within the action area. Additional discussion on energy stressors is included in Section 5.2. This section includes analysis of the potential impacts of: 1) in-water electromagnetic devices and 2) high-energy laser weapons.

8.1.1.2.1 Effects of In-water Electromagnetic Devices on Cetaceans

In-water electromagnetic energy devices used as part of NWTT are unmanned mine warfare systems that simply mimic the electromagnetic signature of a vessel passing through the water. The sound and electromagnetic signature of these devices cause nearby mines to detonate. Normandeau et al. (2011) concluded there was behavioral, anatomical, and theoretical evidence indicating cetaceans sense magnetic fields. Fin, humpback, and sperm whale studies have shown positive correlations between whale behavior and geomagnetic field differences. Although none of the studies have determined the mechanism for magneto-sensitivity, the suggestion from these studies is that whales can sense the Earth's magnetic field and may use it to migrate long distances. Cetaceans appear to use the Earth's magnetic field for migration in two ways: as a map by moving parallel to the contours of the local field topography, and as a timer based on the

regular fluctuations in the magnetic field allowing animals to monitor their progress on this map (Klinowska 1990).

Most of the evidence of cetaceans sensing magnetic fields is indirect evidence from correlation of sighting and stranding locations suggesting that cetaceans may be influenced by local variation in the earth's magnetic field (Kirschvink 1990; Klinowska 1985; Walker et al. 1992). Results from one study in particular showed that long-finned and short-finned pilot whales, striped dolphin, Atlantic spotted dolphin, Atlantic white-sided dolphin, fin whale, common dolphin, harbor porpoise, sperm whale, and pygmy sperm whale were found to strand in areas where the earth's magnetic field was locally weaker than surrounding areas. Results also indicated that certain species may be able to detect total intensity changes of only 0.05 microteslas (Kirschvink et al. 1986). This gives insight into what changes in intensity levels some species are capable of detecting, but does not provide experimental evidence of levels to which animals may physiologically or behaviorally respond.

Impacts to marine mammals associated with electromagnetic fields are dependent on the animal's proximity to the source and the strength of the magnetic field. Electromagnetic fields associated with NWTT activities are relatively weak (only ten percent of the earth's magnetic field at 24 meters), temporary, and localized. Once the source is turned off or moves from the location, the electromagnetic field is gone. A cetacean would have to be present within the electromagnetic field (approximately 200 meters from the source) during the activity in order to detect it, though detection does not necessarily signify a significant biological response which would have an adverse effect on an individual animal. Given the small area associated with mine fields, the infrequency and short duration of magnetic energy use, the low intensity of electromagnetic energy sources (essentially mimicking the magnetic field of a steel vessel), the density of ESA-listed marine mammals in these areas, and the Navy's procedural mitigation measures regarding ESA-listed marine mammal approaches (see Table 36), NMFS considers it extremely unlikely that ESA-listed marine mammals would be exposed to electromagnetic energy at sufficient intensities to create an adverse effect through behavioral disruption or otherwise. Therefore, potential adverse effects from electromagnetic devices used during Navy activities are considered discountable.

8.1.1.2.2 Effects of Lasers on Cetaceans

High-energy laser weapons activities involve evaluating the effectiveness of an approximately 30-kilowatt high-energy lasers deployed from a surface ship or a helicopter to create small but critical failures in potential targets from short ranges. A marine mammal could be exposed to the laser beam at or near the water's surface, which could result in injury or death. However, marine mammals could only be exposed if the laser beam missed the target (i.e., if the laser hit the target, it would not be expected to penetrate the water and potentially impact an animal underwater), which is not a common occurrence. The following safeguards are in place to reduce

the probability of a high-energy laser weapon striking the water: 1) the high-energy laser platform has provisions that prevent misfiring (i.e., firing when not intended) that all but eliminate the possibility of that event, 2) the high-energy laser platforms have built-in constraints that only permit firing when it is locked onto a target, and 3) the operators are trained to stop firing when the laser aim point moves off of the selected target.

Additionally, ESA-listed marine mammal densities in the action area are relatively low, which further reduces the likelihood of a laser strike. The potential for marine mammals to be directly hit by a high-energy laser beam was evaluated by the Navy using statistical probability modeling to estimate the probability of direct strike exposures in a worst-case scenario. We adopt the Navy's quantitative analysis as the best available information for analyzing the effects of high-energy laser weapons on marine mammals. Model input values included high-energy laser use data (e.g., number of high-energy laser exercises and laser beam footprint), size of the training or testing area, marine mammal density data, and animal size. To estimate the probability of hitting a marine mammal in a worst-case scenario (based on assumptions listed below), the impact area for all laser training and testing events was summed over one year in each respective training or testing area. The marine mammal species with the highest average seasonal density within the action area was used in the analysis. All other species with a lower density would be expected to have a lower probability of being struck by the laser. Other conservative assumptions incorporated into the model are as follows:

- The model is two-dimensional and assumes that all animals would be at or near the surface 100 percent of the time.
- The model assumes the animal is stationary and does not account for any movement of the marine mammal or any potential avoidance of the training or testing activity.

The Navy's model estimates 0.0000020 exposures for blue whales, 0.0000075 fin whales, 0.0000002 sei whales, 0.0000159 humpback whales, 0.0000004 Southern Resident killer whales, 0.0000018 sperm whales and 0.0000082 exposures every year for Guadalupe fur seals. Based on the very low number of annual exposures, the characteristics of activities that would use high-energy laser weapons (e.g., short range distance from source to target, high-precision targeting, short duration of the energized beam), and likely animal avoidance behavior of laser targets, NMFS adopts the Navy's analysis and considers it extremely unlikely that ESA-listed marine mammals would be exposed to high energy lasers. Therefore, potential effects from lasers during Navy activities are considered to be discountable.

8.1.1.3 Physical Disturbance and Strike Stressors – Marine Mammals

This section summarizes the analyses of the potential effects of physical disturbance and strike of ESA-listed marine mammals during NWTT activities resulting from in-water devices, military expended materials (including non-explosive practice munitions and fragments from high-

explosive munitions), and seafloor devices. The effects of vessel strike on marine mammals are analyzed in Section 8.2.1.5 below.

8.1.1.3.1 Effects of In-water Devices on Marine Mammals

In-water devices include unmanned vehicles, such as remotely operated vehicles, unmanned surface vehicles and unmanned undersea vehicles and towed devices that are used throughout the action area. Despite thousands of Navy exercises in which in-water devices have been used, there have been no recorded instances of marine species strikes from these devices. The Navy will implement mitigation to avoid potential impacts from in-water device strikes on cetaceans throughout the action area. Mitigation includes Lookouts and watch personnel that have been trained to identify marine mammals and requiring underway vessels and in-water devices that are towed from manned surface platforms to maintain a specified safe distance from marine mammals (See Section 3.5.1). For these reasons, NMFS considers it extremely unlikely for any ESA-listed marine mammal to be struck by an in-water device. It is possible that cetacean species that occur in areas that overlap with in-water devices use may experience some level of physical disturbance, but it is not expected to result in more than a momentary behavioral response. Any avoidance behavior would be of short duration and intensity such that it would be insignificant to the animal. Therefore, potential effects on ESA-listed marine mammals from in-water devices are discountable (in the case of strike) or insignificant and thus not likely to cause adverse effects (in the case of behavioral response).

8.1.1.3.2 Effects of Military Expended Materials on Marine Mammals

While no strike of marine mammals from military expended materials has ever been reported or recorded, the possibility of a strike still exists. We considered the potential for ESA-listed marine mammal strike resulting from NWTT activities involving the following types of military expended materials: 1) all sizes of non-explosive practice munitions, 2) fragments from high-explosive munitions, 3) expendable targets and target fragments; and 4) expended materials other than munitions, such as sonobuoys, expended bathythermographs, and torpedo accessories.

Given the large geographic area involved and the relatively low densities of ESA-listed marine mammals in the action area, we do not believe such interactions are likely. Additionally, while disturbance or strike from any expended material as it falls through the water column is possible, it is extremely unlikely because the objects will slow in velocity as they sink toward the bottom (e.g., guidance wires sink at an estimated rate of 0.2 meters per second; heavier items such as non-explosive munitions would likely sink faster, but would still be slowed as they sink to the bottom), and can be avoided by highly mobile cetacean and pinniped species. In addition, the Navy has proposed procedural mitigation for vessel movement and towed-in water devices to limit the potential for strikes of marine mammals where military expended materials are used in offshore environments (see Section 3.5.1 for details).

In summary, NMFS considers it extremely unlikely for any ESA-listed marine mammal to be struck by military expended materials. Any individuals encountering military expended materials as they fall through the water column are likely to move to avoid them. Given the effort expended by individuals to avoid them will be minimal (i.e., a few meters distance) and temporary, behavioral avoidance of military expended materials sinking through the water column is considered minor with no lasting or meaningful effects expected for an individual animal. For these reasons, potential effects on ESA-listed marine mammals from physical disturbance and strike with military expended materials are discountable (in the case of strikes), or insignificant and thus not likely to cause adverse effects (in the case of behavioral response).

8.1.1.3.3 Effects of Seafloor Devices on Marine Mammals

Activities that use seafloor devices include items placed on, dropped on, or moved along the seafloor such as mine shapes, anchor blocks, anchors, bottom-placed devices, and bottom-crawling unmanned underwater vehicles. Seafloor devices are either stationary or move very slowly along the bottom and do not pose a threat to highly mobile organisms. Objects falling through the water column will slow in velocity as they sink toward the bottom and would be avoided by ESA-listed marine mammals. Given their mobility and the low densities in areas where seafloor devices would likely be used, it is extremely unlikely that ESA-listed marine mammals would be struck by a seafloor device. Therefore, potential effects on ESA-listed marine mammals from seafloor device strike are discountable.

Any individuals encountering seafloor devices on the ocean bottom are likely to behaviorally avoid them. Given the slow movement and relatively small size of seafloor devices, the effort expended by individuals to avoid them is expected to be minimal and temporary, and will not have fitness consequences. Therefore, behavioral avoidance of seafloor devices by ESA-listed marine mammals is insignificant, and thus not likely to cause adverse effects.

8.1.1.4 Entanglement Stressors – Marine Mammals

Expended materials from Navy activities that may pose an entanglement risk include wires, cables, decelerators, and parachutes. Interactions with these materials could occur at the sea surface, in the water column, or on the seafloor. Though there is a potential for ESA-listed marine mammals to encounter military expended material, for the reasons described below, we believe such interactions are extremely unlikely to occur. Additional discussion of entanglement stressors, in general, is included in Section 5.4.

8.1.1.4.1 Effects of Entanglement from Wires and Cables on Marine Mammals

There has never been a reported or recorded instance of a marine mammal entangled in military expended materials despite the Navy expending materials in the action area (and other range complexes) for decades. NOAA (2014a) conducted a review of entanglement of marine species in marine debris with an emphasis on species in the U.S. The review did not document any known instances where military expended materials had entangled a marine mammal. Instead,

the vast majority of entanglements have been from actively fished or derelict fishing gear. For example, Knowlton et al. (2012) conducted a 30-year comprehensive review of entanglement rates of North Atlantic right whales using photographs. Much of the habitat occupied by North Atlantic right whale is coextensive with Navy training and testing activities using military expended materials in the western Atlantic (Navy 2018a). Knowlton et al. (2012) reported that of the 626 individuals whales observed the vast majority showed evidence of entanglement involving non-mobile pot gear and nets used for fishing. Baulch and Perry (2014) reported that nearly 98 percent of documented cetacean entanglements worldwide were from abandoned, lost, or derelict fishing gear.

If encountered, it is extremely unlikely that an animal would get entangled in a fiber optic cable, sonobuoy wires, or guide wire while these were sinking or settling on the seafloor. An animal would have to swim through loops or become twisted within the cable or wire to become entangled, and given the properties of the expended cables and wires (low breaking strength and a design to resist coiling or the forming of loops) the likelihood of marine mammal entanglement from cables and wires is extremely low. Specifically, fiber optic cable is brittle and would be expected to break if kinked, twisted or sharply bent. Thus, the physical properties of the fiber optic cable would not allow the cable to loop, greatly reducing the likelihood of entanglement. Based on degradation times, guidance wires would break down within one to two years and no longer pose an entanglement risk.

For these reasons cited above, it is extremely unlikely that ESA-listed marine mammals will become entangled in military expended wires and cables in the action area. The effects from entanglement of ESA-listed cetaceans in wires and cables are, therefore, discountable.

8.1.1.4.2 Effects of Entanglement from Decelerators and Parachutes on Marine Mammals

The majority of the decelerators and parachutes used for NWTT activities are in the small size category (see Table 62 above) and are associated with sonobuoys. Both small- and medium-sized decelerators and parachutes are made of cloth and nylon and have weights attached to their short attachment lines (i.e., from 0.3 to 5.8 meters). The majority of parachutes/decelerators would not remain suspended in the water column for more than a few minutes as the attached weights speed the sinking of materials to the seafloor. Small and medium decelerators/parachutes with weights are expected to remain at the surface for 5 to 15 seconds before the housing sinks to the seafloor where it becomes flattened (Navy 2019c).

Some large or extra-large decelerators/parachutes are also proposed for use in the action area. In contrast to small and medium parachutes, large parachutes do not have weights attached and may remain at the surface or are suspended in the water column for some time prior to eventual settlement onto the seafloor. However, a limited number of these items are proposed for use (i.e., ten large parachutes annually) in the NWTT action area. The small number of large or extra-large parachutes proposed for use annually, and generally low species densities, reduces the

potential for ESA-listed marine mammals to encounter and become entangled in these items. In addition, during activities that involve recoverable targets (e.g., aerial drones), the Navy recovers the target and any associated decelerators/parachutes to the maximum extent practicable consistent with personnel and equipment safety. This standard operating procedure would further reduce the potential entanglement of cetaceans in decelerators/parachutes.

As noted above, the vast majority of large whale entanglements have been associated with fishing gear. In contrast, as noted previously, there has never been a documented instance where a large whale was observed entangled in military expended material, including decelerators and parachutes. There are a number of key differences between decelerators/parachutes and fishing gear that result in the likelihood of entanglement in these materials being significantly lower than the likelihood of entanglement in fishing gear. First, as noted above, except for a small number of large decelerators/parachutes, most decelerators/parachutes used by the Navy sink quickly to the seafloor and do not remain suspended in the water column for extended periods of time. This is in contrast to fishing gear which can remain in the water column for days or weeks at a time. Additionally, parachutes would be highly visible in the water column, likely alerting a nearby animal to the presence of the obstacle. By contrast, fishing gear may consist of some buoys and traps that are visible, but often contains hundreds of feet of rope or line in between these items that is often not visible by design. Finally, the cords associated with parachutes are, at most, 80 feet long. In contrast, typical gear associated with some fisheries has hundreds of feet of rope suspended in the water column.

There is the potential for a bottom feeding cetacean (e.g., sperm whale) to become entangled when they are foraging in areas where parachutes have settled onto the seafloor. For example, if bottom currents are present, the canopy may temporarily billow and pose a greater entanglement threat. However, the likelihood of bottom currents causing a billowing of a parachute and being encountered by an ESA-listed cetacean is considered extremely unlikely.

In summary, based on their deep-water location of use, their sinking rate, their degradation rate, and the low density of ESA-listed marine mammals, it is extremely unlikely that these species would become entangled in small or medium decelerators or parachutes. Based on the limited number deployed, the standard operating procedure to recover decelerators/parachutes to the maximum extent practicable, and the low density of ESA-listed marine mammals, and it is extremely unlikely that these species would become entangled in large or extra-large decelerators or parachutes. Therefore, potential effects on ESA-listed marine mammals from entanglement in decelerators or parachutes are discountable.

8.1.1.4.3 Effects of Entanglement from Biodegradable Polymers

The Navy has proposed to test the use of biodegradable polymers during four events per year at the Dabob and Keyport range sites. Of the marine mammals in the action area, only Southern Resident killer whales could overlap with areas proposed for biodegradable polymer testing.

Unlike other entanglement stressors, biodegradable polymers only retain their strength for a relatively short period of time; the longer the biodegradable polymer remains in the water, the weaker it becomes, making it more brittle and likely to break (Navy 2020e). Therefore, the potential for entanglement by a killer whale would be limited. A killer whale would have to encounter the biodegradable polymer immediately after it was expended for it to be a potential entanglement risk (Navy 2020e). Given the small number of events proposed, the short entanglement risk time frame after deployment, and the low likelihood of Southern Resident killer whales being in the proposed areas during this activity, we find that potential effects on Southern Resident killer whales from entanglement in biodegradable polymers are extremely unlikely and thus discountable.

8.1.1.5 Ingestion Stressors – Marine Mammals

Additional discussion on ingestion stressors is included in Section 5.5. The munitions and other materials small enough to be ingested by ESA-listed cetaceans are small- and medium-caliber projectiles, broken pieces of firing targets, chaff, flare caps, and shrapnel fragments from explosive ordnance. Other military expended materials (e.g., non-explosive bombs or surface targets) are considered too large for ESA-listed marine mammals to consume and are made of metal a marine mammal would not be able to break-apart to ingest.

Most expendable materials would be used over deep water portions of the action area and most items are expected to sink quickly and settle onto the seafloor, with the exception of chaff and some firing target materials. Given the limited time most items will spend in the water column, it is not likely that these items would be accidentally ingested by ESA-listed marine mammals that do not typically forage on the sea floor. Of the marine mammals in the action area, the only species potentially exposed to expended munitions and shrapnel fragments while foraging on the sea floor in deep water is sperm whales. Sperm whales are recorded as having ingested fishing net scraps, rope, wood, and plastic debris such as plastic bags and items from the seafloor (Walker and Coe 1990; Whitehead 2003b). However, the relatively low density of both sperm whales and expended materials along the vast sea floor suggests ingestion would be rare. Humpback whales also feed at the seafloor but do so in relatively shallow water and soft sediment areas where ingestion stressors are less likely to be present (fewer activities take place in shallow water and expended materials are more likely to bury in soft sediment and be less accessible). If a large whale were to accidentally ingest expended materials small enough to be eaten, it is likely the item will pass through the digestive tract and neither result in an injury (e.g., Wells et al. 2008) nor an increased likelihood of injury from significant disruption of normal behavioral patterns such as breeding, feeding, or sheltering.

ESA-listed marine mammals may also encounter military expended material that remains suspended in the water column for extended periods of time. Since baleen whales feed by filtering large amounts of water, they could encounter and consume debris at higher rates than other marine animals (NOAA 2014a). For example, baleen whales are believed to routinely

encounter microplastics (from numerous anthropogenic sources) within the marine environment based on concentrations of these items and baleen whale feeding behaviors (Andrady 2011). Laist (1997) reported on two species of mysticetes (bowhead and minke whale) with records of having ingested debris items that included plastic sheeting and a polythene bag. Bergmann et al. (2015) documented records of marine debris ingestion in seven mysticetes, including right whales, pygmy right whales, gray whales, and four rorqual species. Information compiled by Williams et al. (2011a) listed humpback whale, fin whale, and minke whale as three species of mysticetes known to have ingested debris including items the authors characterized as fishing gear, polyethylene bag, plastic sheeting, plastic bags, rope, and general debris. Military expended materials were not documented as having been consumed in any of these studies.

Some Styrofoam, plastic endcaps, and other small military expended materials (e.g., chaff, flare pads, pistons) may float for some time before sinking. However, these items are likely too small to pose a risk of intestinal blockage to any marine mammal that happened to encounter it. Chaff is composed of fine fibers of silicon dioxide coated with aluminum alloy. Due to its light weight and small size this floating material can be carried great distances in both air and water currents. Their dispersal in wind and water results in chaff fibers likely occurring in low densities on the ocean surface. Several literature reviews and controlled experiments have indicated that chaff poses little risk, except at concentrations substantially higher than those that could reasonably occur from military training (Arfsten et al. 2002; Force 1997; Hullar et al. 1999). Similar to chaff, flare pads and pistons are also relatively small and float in sea water. Given the small size, low densities, and low toxicity of chaff or flare expended materials, any accidental ingestion by ESA-listed marine mammals feeding at the ocean surface is not expected to result in an injury or an increased likelihood of injury from significant disruption of normal behavioral patterns such as breeding, feeding, or sheltering. Chaff cartridge plastic end caps and pistons and flare pads would also be released into the marine environment during Navy activities, where they may persist for long periods, and therefore could be ingested by cetaceans while initially floating on the surface and sinking through the water column. However, these materials would eventually sink to the seafloor where they would be less likely to be ingested by marine mammals. Firing target materials are normally retrieved before sinking so it is not reasonable to expect ingestion of these items to occur. Chaff recovery studies conducted within chaff release areas within several U.S. military operating areas have shown that the concentrations of chaff dipoles (fibers) in the environment are very small (Farrell and Siciliano 2004). Consequently, even though chaff dipoles contain aluminum and other trace metals that can ultimately be leached from the chaff, the amount of chaff needed to raise environmental concentrations of these metals above background levels far exceeds the number than can be realistically deposited in the open ocean (Farrell and Siciliano 2004). As such, chaff releases are not expected to have any significant effect on marine ecosystem functioning (Farrell and Siciliano 2004).

In conclusion, since we expect smaller military expended materials would likely pass through marine mammals with no adverse effects, the effects of this stressor (i.e., ingestion of small expended materials) are insignificant. Since ingestion of military expended material of sufficient

size to result in adverse effects on ESA-listed marine mammals is extremely unlikely, the effects of this stressor (i.e., ingestion of large expended materials) are discountable. While baleen whales could accidentally ingest chaff or flare remains, if this occurs the effects of these stressors on those individuals exposed are expected to be so minor as to be insignificant and thus not likely to cause adverse effects.

8.1.1.6 Stressors Resulting in Effects to Marine Mammal Habitat or Prey

This section analyzes potential impacts to ESA-listed marine mammals exposed to stressors through impacts to their habitat or prey or through the introduction of parasites or disease. The stressors evaluated in this section include: 1) explosives, 2) explosive byproducts and unexploded munitions, 3) metals, 4) chemicals, and 5) transmission of disease and parasites.

8.1.1.6.1 Explosives

In this section, we discuss the anticipated effects of NWTT explosives on the prey of blue whales, fin whales, Central America DPS and Mexico DPS humpback whales, sei whales, sperm whales, and Guadalupe fur seals. We address the effects of NWTT explosives on the prey of Southern Resident killer whales in Section 8.2.1.4.

Underwater explosions could impact other species in the food web, including prey species that marine mammals feed upon. The impacts of explosions would differ depending on the type of prey species in the area of the blast. In addition to physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that might include swimming to the surface or scattering away from the source. This startle and flight response is the most common secondary defense among animals (Hanlon and Messenger 1996; Mather 2004). The abundances of prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Alternatively, any prey species that would be directly injured or killed by the blast could draw in scavengers from the surrounding waters that would feed on those organisms, and in turn could be susceptible to becoming directly injured or killed by subsequent explosions. Any of these scenarios would be temporary, only occurring during activities involving explosives, and no lasting effect on prey availability or the pelagic food web would be expected. For this reason, the effects of NWTT explosives on blue whales, fin whales, humpback whales, sei whales, and Guadalupe fur seals through impacts to their prey are insignificant and thus not likely to cause adverse effects.

8.1.1.6.2 Explosive Byproducts and Unexploded Munitions

High-order explosions (i.e., a successful explosion or an explosion that produces the intended result) consume almost all of the explosive material in the ordnance, leaving little to no material in the environment that could affect marine mammal species or their habitats. By contrast, low order detonations and unexploded munitions leave more explosive material in the environment. Lotufo et al. (2010) studied the potential toxicity of Royal Demolition Explosive byproducts to marine organisms. The authors concluded that degradation products of these explosives are not

toxic at realistic exposure levels. Furthermore, while explosives and their degradation products were detectable in marine sediment approximately 6 to 12 inches away from degrading munitions, the concentrations of these compounds were not statistically distinguishable from baseline levels beyond 3 to 6 feet from the degrading munitions. Based on these results, while it is possible that ESA-listed marine mammals could be exposed to degrading explosives, such exposure would likely only occur within a very small radius of the explosive, and exposure to degrading explosives at toxic levels is extremely unlikely.

Research efforts focused on World War II underwater munitions disposal sites in Hawaii (Briggs et al. 2016; Edwards and coauthors. 2016; Kelley et al. 2016; Koide et al. 2016) and an intensively used live fire range in the Mariana Islands (Smith and Marx Jr. 2016) provide information in regard to the impacts of undetonated materials and unexploded munitions on marine life. Findings from these studies indicate that there were no adverse impacts on the local ecology from the presence of degrading munitions and there was no bioaccumulation of munitions-related chemicals in local marine species.

The island of Farallon de Medinilla in the Commonwealth of the Northern Mariana Islands has been used as a target area since 1971. Although outside the action area for this consultation, we use this island as an example of the anticipated effects from this stressor. Between 1997 and 2012, there were 14 underwater scientific survey investigations around the island providing a long term look at potential impacts on the marine life from training and testing involving the use of munitions (Smith and Marx Jr. 2016). Munitions use has included high-explosive rounds from gunfire, high-explosives bombs by Navy aircraft and U.S. Air Force B-52s, in addition to the expenditure of inert rounds and non-explosive practice bombs. Marine life assessed during these surveys included algae, corals, benthic invertebrates, sharks, rays, and bony fishes, and sea turtles. The investigators found no evidence over the 16-year period, that the condition of the biological resources had been adversely impacted to a significant degree by the Navy training activities (Smith and Marx Jr. 2016). Furthermore, they found that the health, abundance, and biomass of fishes, corals and other marine resources were comparable to or superior to those in similar habitats at other locations within the Mariana Archipelago.

The concentration of munitions, explosives, expended material, or devices in any one location in the action area are expected to be a small fraction of that from the sites described above. As a result, explosion by-products and unexploded munitions are not anticipated to have adverse effects (i.e., no measureable effects are anticipated) on water quality or cetacean prey abundance in the action area. For this reason, the effects of explosive byproducts and unexploded munitions on ESA-listed marine mammals through impacts on prey and water quality are considered insignificant and thus not likely to cause adverse effects.

8.1.1.6.3 Metals

Metals are introduced into seawater and sediments as a result of training and testing activities involving ship hulks, targets, munitions, and other military expended materials (Environmental

Sciences Group, 2005). Some metals bioaccumulate and physiological impacts begin to occur only after several trophic transfers concentrate the toxic metals. Evidence from a number of studies (Briggs et al. 2016; Edwards and coauthors. 2016; Kelley et al. 2016; Koide et al. 2016; Navy 2013c) indicate metal contamination is highly localized and that bioaccumulation resulting from munitions cannot be demonstrated. Specifically, in sampled marine life living on or around munitions on the seafloor, metal concentrations could not be definitively linked to the munitions since comparison of metals in sediment next to munitions show relatively little difference in comparison to other baseline marine sediments used as a control (Koide et al. 2016). Research has demonstrated that some smaller marine organisms are attracted to metal munitions as a hard substrate for colonization or as shelter (Kelley et al. 2016; Smith and Marx Jr. 2016), but this would not have an effect on the availability of marine mammal prey. The research cited above indicates that metals introduced into the action area are unlikely to have measureable impacts on ESA-listed marine mammal prey or habitat. Thus, the effects of metals introduced into seawater and sediments as a result of NWTT activities on ESA-listed marine mammals through impacts to their prey or habitat are insignificant and thus not likely to cause adverse effects.

8.1.1.6.4 Chemicals

Several Navy training and testing activities introduce chemicals into the marine environment that are potentially harmful in higher concentrations. However, rapid dilution would be expected and toxic concentrations are unlikely to be encountered by ESA-listed marine mammals or their prey. Chemicals introduced are principally from flares and propellants for missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment. Flares and missiles that operationally fail may release perchlorate, which is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals if in sufficient concentration. However, such concentrations would be localized and are not likely to persist in the ocean. Research has demonstrated that perchlorate did not bioconcentrate or bioaccumulate, which was consistent with the expectations for a water-soluble compound (Furin et al. 2013). Given the dynamic nature of the environment (currents, tides, etc.), long-term impacts from perchlorate in the environment near the expended item are not expected. It is extremely unlikely that perchlorate from failed expendable items would compromise water quality to the point that it would result in adverse effects on ESA-listed marine mammal prey or habitat. In summary, the effects of chemicals used during Navy training and testing on ESA-listed marine mammals in the NWTT action area via water quality and prey are considered discountable.

8.1.1.6.5 Disease and Parasites

The primary vector through which parasites of disease would be transferred to new locations and the ESA-listed species there would be through the deployment of marine mammals used by the Navy's Marine Mammal Systems. Pinnipeds and dolphins used in the Marine Mammal Program

are under exceptional care and control. Navy animals receive regular veterinarian care, including predeployment exams, regular deworming, and regional screening for specific pathogens of interest (Navy 2013b). The animals are fed high-quality, suitable for human consumption, fish to minimize the likelihood of parasite ingestion and animal waste is collected and managed to control the potential spread of parasites. Prior to animal deployment Navy personnel observe the surrounding area and if wild marine mammals are spotted animal deployment is delayed. Contact between Navy animals and wild animals is minimized to the greatest extent possible. In the 40 years the Marine Mammal Program has been operating there has been no known disease or parasite transmissions from Navy animals to wild animals (Navy 2013b). Given the exceptional care Navy animals receive, the waste disposal protocols, the minimal time Navy animals are in contact with wild animals, and the 40 years of their use without incident, it is unlikely parasites or diseases will be transferred to ESA-listed species.

In summary, we find it extremely unlikely that ESA-listed species would be exposed to toxic levels of explosives, explosive byproducts, metals, other chemicals, or parasites/disease resulting from NWTT activities. This is based on the information provided above regarding the potential for explosives and byproducts, metals, chemicals, and transmission of disease and parasites to indirectly affect marine ESA-listed cetacean, pinniped, sea turtle and fish species through habitat and prey availability impacts. Therefore, the effects of secondary stressors from NWTT activities on ESA-listed species are considered discountable.

8.1.2 Leatherback Sea Turtles

We determined that several of the acoustic stressors, all of the energy stressors, entanglement stressors, ingestion stressors, and potential secondary stressors are not likely to adversely affect leatherback sea turtles. Our analysis of the effects of these stressors on leatherback sea turtles is summarized below. It should be noted that green sea turtles (East Pacific DPS), loggerhead sea turtles (North Pacific Ocean DPS), and olive ridley sea turtles (both the Mexico Pacific Coast breeding population and all other areas) were previously determined to be not likely adversely affected by any stressors associated with the proposed action (see Section 6.1).

8.1.2.1 Acoustic Stressors – Leatherback Sea Turtles

The discussion below focuses on a subset of the acoustic stressors associated with the proposed action. In the following section we discuss the acoustic stressors we determined are not likely to adversely affect leatherback sea turtles. The effects of acoustic stressors which we determined are likely to adversely affect leatherbacks (i.e., explosives) are discussed in Section 8.2.2.1.

8.1.2.1.1 Effects of Vessel Noise on Leatherback Sea Turtles

Additional information on vessel noise as a potential stressor associated with the proposed action can be found in Section 5.1.3.

Leatherback sea turtles could be exposed to a range of vessel noises within their hearing abilities. The Navy vessels will produce low-frequency, broadband underwater sound below one kilohertz

for larger vessels, and higher-frequency sound between one kilohertz to 50 kilohertz for smaller vessels, although the exact level of sound produced varies by vessel type. Depending on the context of exposure, responses of leatherback sea turtles in the action area to vessel noise disturbance would include startle responses, avoidance, or other behavioral reactions, physiological stress responses, or no measurable response at all.

Limited information is available on how or if sea turtles may respond to noise from Navy vessels during NWTT activities. As discussed previously, Hazel et al. (2007) suggested that sea turtles may rely more on visual than auditory cues when reacting to approaching vessels. Additionally, there is evidence that reptiles may rely primarily on senses other than hearing for interacting with their environment, such as vision (Narazaki et al. 2013) and magnetic orientation (Avens and Lohmann 2003; Putman et al. 2015). This suggests that if leatherback sea turtles were to respond to a Navy vessel, the animal might not respond to the vessel based on noise alone. Popper et al. (2014a) noted that available information on the effects of vessel noise or other continuous sounds on sea turtles is lacking. The only potential effect Popper et al. (2014a) suggested could occur from vessel noise was a behavioral response or masking, with a higher likelihood of a behavioral response occurring the closer the sea turtle is to the vessel.

Compared to marine mammals that are highly adapted to use sound in the marine environment, sea turtles are less dependent on sound and their hearing is more limited in range to very low frequencies. Any masking of biologically important sounds for sea turtles would be temporary, occurring only when a vessel and sea turtle are in close proximity to one another. The short, temporary exposure, would not have any measurable effects on an animal's fitness.

If a sea turtle responded behaviorally to noise from a Navy vessel, most responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives. Changes in behavior would likely consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (foraging, active swimming or traveling) and then returning to the resting or milling behavior shortly thereafter. Any behavioral responses to vessel noise are expected to be temporary (e.g., a startle response, brief avoidance behavior) and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individual leatherbacks that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. We do not expect these short term behavioral reactions to increase the likelihood of injury or result in fitness consequences to exposed individuals.

For these reasons, vessel noise is expected to cause minimal disturbance to leatherback sea turtles. If a sea turtle detects a vessel and avoids it, or has a temporary stress response from the noise disturbance, these responses are expected to be temporary and only endured while the vessel transits through the area where the sea turtle encountered it. Sea turtle responses to vessel noise disturbance are extremely minor, and a sea turtle would be expected to return to normal

behaviors and baseline stress levels shortly after the vessel passes. As a result, we find that the likely effects from exposure to vessel noise resulting from the proposed action on leatherback sea turtles are insignificant and thus not likely to cause adverse effects.

8.1.2.1.2 Effects of Aircraft Noise on Leatherback Sea Turtles

Additional information on aircraft as a potential stressor associated with the proposed action can be found in Section 5.1.4.

Based on sea turtle sensory biology (Ketten and Bartol 2006; Lenhardt et al. 1994; Ridgway et al. 1969), sound from low flying aircraft could be heard by a sea turtle at or near the surface. Turtles might also detect low flying aircraft via visual cues such as the aircraft's shadow. This suggests that sea turtles might not respond to aircraft overflights based on noise alone.

Exposure to fixed-wing aircraft noise would be brief (seconds) as an aircraft quickly passes overhead at relatively high speeds. Exposure to helicopter overflights may last longer and would have a higher likelihood of causing a behavioral response from a sea turtle due to the lower flight altitudes and longer duration the helicopter could be in proximity to an animal. The Navy proposes to conduct exercises involving helicopters both during the day and night. These exercises may occur for extended periods of time, up to a couple of hours in some areas. During these activities, helicopters would typically transit throughout an area and may hover over the water. Longer duration activities (such as a couple of hours) and periods of time where helicopters hover may increase the chance that a sea turtle may startle, change swimming patterns, or have a physiological stress response. Exposures to both sorts of aircraft would be infrequent based on the transitory and dispersed nature of the overflights and repeated exposure to individual animals over a short period of time (hours or days) is extremely unlikely. Furthermore, the SEL would be relatively low to sea turtles that spend the majority of their time underwater and may not even detect the aircraft depending on where they are at in the water column at the time of the overflight.

As with vessel disturbance above, little information is available on how sea turtles respond to aircraft. The working group that developed the 2014 ANSI Guidelines for fish and sea turtles Popper et al. (2014a) did not consider this specific acoustic stressor for sea turtles, in part because it is not considered to pose a great risk. For the purposes of this consultation, we assume leatherback sea turtles in the action area may exhibit similar short-term behavioral responses (e.g., diving, changes in swimming direction, etc.) consistent with those behaviors observed during aerial research surveys of sea turtles. We are unaware of any data on the physiological responses sea turtles exhibit to aircraft, but we conservatively assume a low-level, short-term stress response is possible. There could also be temporary masking of biologically relevant cues from exercises that generate longer duration of sound exposure with a hovering helicopter. However, in general aircraft overflight is brief, and does not persist in the action area for significant periods of time (not longer than a few hours), nor is the sound expected to be transmitted well into the water column. Thus, the risk of masking any biologically relevant sound

to sea turtles is considered very low. A sea turtle could leave the area where noise disturbance persists for a few hours, and thereby avoid continued disturbance. Any startle reactions that occur are expected to be brief, with sea turtles resuming normal behaviors once the aircraft is no longer detectable or leaves the area. Due to the short-term nature of any exposures to aircraft and the brief responses expected to the noise or visual disturbance produced, the effects of aircraft overflight noise on leatherback sea turtles is considered temporary and minor. As a result, we find that the likely effects from exposure to aircraft overflight noise resulting from the proposed action on leatherback sea turtles are insignificant and thus not likely to cause adverse effects.

8.1.2.1.3 Effects of Weapons Firing, Launch, and Impact Noise on Leatherback Sea Turtles

Additional information on weapons noise as a potential stressor associated with the proposed action can be found in Section 5.1.5.

Leatherback sea turtles may be exposed to sounds caused by weapons firing (guns, missile, torpedoes), objects dropping in the water, and inert impact of non-explosive munitions on the water's surface. In general, these are impulsive sounds generated in close proximity to or at the water surface (with the exception of items that are launched underwater). Most in-air weapons noise is expected to be reflected at the air-water interface, and as such is not expected to transmit deep into the water column, nor to propagate across a large expanse of surface waters. The resulting noise would be limited and strongest underwater just below the surface and directly under the firing point of the weapon. Sound produced from missile and target launches is typically the highest near the initiation of the booster rocket and rapidly fades as the missile or target travels downrange from the firing point (Navy 2018b).

The highest level of sound expected to transmit to the water would be from large-caliber guns fired at the lowest elevation angle with peak levels of sound directly below the blast. These peak levels are approximately 200 dB (re 1 µPa). These levels are lower than the impulsive sound pressure thresholds that are thought capable of causing hearing impairment or injury to sea turtles, but higher than the rms value (175 dB) that could elicit a behavioral response. Therefore, the potential effects that are more likely to result from weapons noise exposure for leatherback sea turtles are temporary behavioral responses, masking, and concurrent stress responses.

Noise produced from firing weapons is expected to last only a few seconds. Most incidents of impulsive sounds produced by weapons firing, launch, or inert object impacts would be single events, with the exception of gunfire activities (Navy 2018b). Gunfire activities could produce multiple shots fired in a brief period of time. Given that these sounds are below injury criteria for sea turtles, and are expected to be very brief and intermittent over the duration of activities in the action area, only brief startle reactions, diving responses or other avoidance behaviors are likely to occur for sea turtles. For the same reasons, masking of biologically relevant sounds is also not expected to occur for sea turtles because weapons noise would not persist for a long enough

duration, and sea turtles are more likely to rely on other senses to detect environmental cues such as visually or through orientation to the earth's magnetic field.

For the reasons above, any physiological stress and behavioral reactions from weapons firing noise would likely be brief and are expected to return to normal shortly after the weapons noise ceases. Therefore, the effects on leatherback sea turtles from weapons noise exposure are anticipated to be minor, temporary and will not lead to a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Likely responses to weapons noise would be short-term with sea turtles returning to normal behaviors and baseline stress levels shortly after the weapon is fired. In summary, we find that the likely effects from exposure to weapons noise resulting from the proposed action on leatherback sea turtles are insignificant and thus not likely to cause adverse effects.

8.1.2.1.4 Effects of Explosions in the Air on Leatherback Sea Turtles

Additional information on in-air explosions as a potential stressor associated with the proposed action can be found in Section 5.1.2 above.

Explosions that occur during air warfare would typically be at sufficient altitude that a large portion of the sound refracts upward due to cooling temperatures with increased altitude and would not reach the water's surface where leatherback sea turtles could occur. Sea turtles within the audible range of sound from explosions in air may exhibit a behavioral startle response but are expected to quickly return to normal behavior. Due to the short duration and sporadic nature of explosions in the air and the extremely low likelihood of a leatherback sea turtle being within close enough proximity to detect sounds from such explosions, we do not expect this stressor would result in a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, the effects of sound from explosions in air on ESA-listed sea turtles would be insignificant and thus not likely to cause adverse effects.

8.1.2.1.5 Effects of Sonars and Other Transducers on Leatherback Sea Turtles

The potential effects of leatherback sea turtle sonar exposure include hearing impairment, an observable behavioral response, a stress response that may not be detectable, or masking. These potential effects are discussed below, with reference to Section 2.2.3 as appropriate, which describes the criteria and thresholds for estimating potential effects from sonar. Additional information on sonars and other transducers as a stressor associated with the proposed action can be found in Section 5.1.1.

Hearing Impairment

Hearing loss is a noise-induced decrease in hearing sensitivity, which can either be temporary or permanent. To date, no studies have been conducted specifically related to sea turtle hearing loss. The Navy evaluated sea turtle susceptibility to hearing loss (from sonar exposure) based upon what is known about sea turtle hearing abilities in combination with non-impulsive auditory effect data from other species such as marine mammals and fish. The exposure estimates used for

our sea turtle acoustic effects analysis were produced by the Navy based on NAEMO output and post-processing techniques, and are based on the maximum number of activities in a given year under the proposed action.

Physiological stress

Stress caused by acoustic exposure has not been studied for sea turtles. As described for cetaceans, a stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor. If the magnitude and duration of the stress response is too great or too long it can have negative consequences to the animal such as decreased reproductive rates, decreased immune function, diminished foraging capacity, etc. Physiological stress is typically analyzed by measuring stress hormones (such as cortisol), other biochemical markers, and vital signs. To our knowledge, there is no direct evidence indicating that sea turtles will experience a stress response if exposed to acoustics stressors. However, physiological stress has been measured for sea turtles during nesting, capture, and handling (Flower et al. 2015; Gregory and Schmid 2001; Jessop et al. 2003; Lance et al. 2004), and when caught in entanglement nets and trawls (Hoopes et al. 2000; Snoddy et al. 2009). Therefore, based on their response to these other anthropogenic stressors, and including what is known about cetacean stress responses, we assume that some sea turtles will experience a stress response if exposed to a detectable sound stressor. Compared to other marine animals, such as marine mammals which are highly adapted to use sound in the marine environment, sea turtle hearing is limited to lower frequencies and is less sensitive. As such, the range of sounds that may produce a stress response in sea turtles is expected to be more limited compared with other taxa that are more sensitive to acoustic stressors.

Animals often respond to anthropogenic stressors in a manner that resembles a predator response (Beale and Monaghan 2004; Frid 2003; Frid and Dill 2002; Gill et al. 2001; Harrington and Veitch 1992; Lima 1998; Romero 2004). As predators generally induce a stress response in their prey (Dwyer 2004; Lopez and Martin 2001; Mateo 2007), we assume that sea turtles may experience a stress response if exposed to acoustic stressors, especially loud sounds. We expect breeding adult females may experience a lower stress response, as studies on loggerhead, hawksbill, and green turtles have demonstrated that females appear to have a physiological mechanism to reduce or eliminate hormonal response to stress (predator attack, high temperature, and capture) in order to maintain reproductive capacity at least during their breeding season; a mechanism apparently not shared with males (Jessop 2001; Jessop et al. 2000; Jessop et al. 2004). However, anthropogenic sound producing activities may have the potential to provide additional stressors beyond those that naturally occur.

Due to the limited information about acoustically induced stress responses in sea turtles, we assume physiological stress responses would occur concurrently with any other response such as hearing impairment or behavioral disruptions. However, we expect such responses to be brief, with animals returning to a baseline state within hours to days. As with cetaceans, such a short,

low level stress response may in fact be adaptive and beneficial as it may result in sea turtles exhibiting avoidance behavior, thereby minimizing their exposure duration and risk from more deleterious, high sound levels.

Masking

Although it has not been documented, masking could occur in sea turtles if the external sound source interferes with the animal's use of acoustic cues to navigate their environment. Compared to other marine animals, such as marine mammals which are highly adapted to use sound in the marine environment, sea turtle hearing is limited to lower frequencies and is less sensitive. Because sea turtles likely use their hearing to detect broadband low-frequency sounds in their environment, the potential for masking would be limited to certain sound exposures. Only continuous anthropogenic sounds that have a significant low-frequency component, are not of brief duration, and are of sufficient received level could create a meaningful masking situation (e.g., long-duration vessel noise affecting natural background and ambient sounds). Other intermittent, short-duration sound sources with low-frequency components (e.g., low-frequency sonar) would have more limited potential for masking, depending on how frequently the sound occurs.

As described previously, there is evidence that sea turtles may rely primarily on senses other than hearing for interacting with their environment, such as vision (Narazaki et al. 2013), magnetic orientation (Avens and Lohmann 2003; Putman et al. 2015), and scent (Shine et al. 2004). Thus, any effect of masking on sea turtles could be mediated by their normal reliance on other environmental cues.

Behavioral Responses

Behavioral responses fall into two major categories: alterations in natural behavior patterns and avoidance. The response of a sea turtle to an anthropogenic sound would likely depend on the frequency, duration, temporal pattern, and amplitude of the sound as well as the animal's prior experience with the sound and the context in which the sound is encountered. In the ANSI Guidelines (Popper et al. 2014a), qualitative risk factors were developed to assess the potential for sea turtles to respond to various underwater sound sources.

To date, very little research has been conducted on sea turtle behavioral responses relative to sonar exposure. Because of this, the working group that prepared the ANSI Guidelines provide parametric descriptors of sea turtle behavioral responses to sonar and other transducers. The working group estimate that the risk of a sea turtle responding to a low-frequency sonar (less than one kilohertz) is low regardless of proximity to the source, and that there is no risk of a sea turtle responding to a mid-frequency sonar (one to ten kilohertz).

For purposes of our effects analysis, we requested the Navy estimate the number of sea turtles that could be exposed to sonar within their hearing range at received levels of 175 dB rms re: 1 µPa SPL or greater. This level is based upon work by McCauley et al. (2000b), who experimentally examined behavioral responses of sea turtles in response to seismic air guns. The

authors found that loggerhead turtles exhibited avoidance behavior at estimated sound levels of 175 to 176 dB rms (re: 1 µPa), or slightly less, in a shallow canal. They reported a noticeable increase in swimming behavior for both green and loggerhead turtles at received levels of 166 dB rms (re: 1 µPa). At 175 dB rms (re: 1 µPa), both green and loggerhead turtles displayed increased swimming speed and increasingly erratic behavior (McCauley et al. 2000b). Based on these data, NMFS assumes that sea turtles would exhibit a significant behavioral response in a manner that constitutes harassment or other adverse behavioral effects, when exposed to received levels of 175 dB rms (re: 1 µPa). This is the level at which sea turtles are expected to begin to exhibit avoidance behavior based on experimental observations of sea turtles exposed to multiple firings of nearby or approaching air guns. Because data on sea turtle behavioral responses to non-impulsive sounds, such as sonars, is limited, the air gun dataset was used to inform our analysis. We considered that the relative risk of a sea turtle responding to air guns would be higher than the risk of responding to sonar, so it is likely that potential sea turtle behavioral responses to sonar exposures are a sub-set of sea turtles exposed to received levels of 175 dB rms (re: 1 µPa) or greater.

Exposure and Response Analysis

The Navy's quantitative analysis (discussed in Section 2.2.1) predicts that no leatherback sea turtles are likely to be exposed to the high received levels of sound from sonars or other transducers that could cause PTS, TTS, or injury during a maximum year of training and testing activities under the proposed action. Only a limited number of sonar and other transducers with frequencies within the range of sea turtles' hearing (less than two kilohertz) and high source levels have the potential to cause TTS and PTS. The quantitative analysis, also predicts no leatherback sea turtles are likely to be exposed to received levels from sonars in their hearing range at or exceeding 175 dB re 1 µPa SPL (rms), the received level associated with onset of avoidance behavior in air gun studies. Therefore, no sea turtles are expected to exhibit avoidance or any other higher severity behavioral response to sonars or other transducers during a maximum year of training and testing activities.

Although masking of biologically relevant sounds by the limited number of sonars and other transducers operated in sea turtle hearing range is possible, this would only occur in certain circumstances. Sea turtles most likely use hearing to detect nearby broadband, continuous environmental sounds, such as the sounds of waves crashing on the beach. The use characteristics of most sonars, including limited bandwidth, beam directionality, limited beam width, relatively low source levels, low duty cycle, and limited duration of use, would both greatly limit the potential for a sea turtle to detect these sources and limit the potential for masking of broadband, continuous environmental sounds. In addition, broadband sources within sea turtle hearing range, such as countermeasures used during anti-submarine warfare, would typically be used in offshore areas, not in nearshore areas where detection of beaches or concentrated vessel traffic is relevant. Implementation of mitigation may further reduce the already low risk of auditory impacts on sea turtles. Depending on the sonar source, mitigation includes powering down the sonar or ceasing active sonar transmission if a sea turtle is observed in the mitigation zone, as discussed in Section 3.6.1 (Procedural Mitigation).

Due to the short term and infrequent nature of any exposures to sonar and transducers and the brief responses that could follow such exposure, the effects of sonar and transducers on ESA-listed sea turtles is considered temporary and minor. In summary, we find that the likely effects from exposure to sonar and transducers resulting from the proposed action are insignificant for leatherback sea turtles, and thus not likely to cause adverse effects.

8.1.2.1.6 Effects of Other Acoustic Sources Not Quantitatively Analyzed

Several of the acoustic sources associated with NWTT activities were not quantitatively analyzed in terms of their effects on ESA-listed species. These include the following: broadband sound sources; Doppler sonar; fathometers; hand-held sonar; imaging sonar; high-frequency acoustic modems; tracking pingers; acoustic releases; and side-scan sonars (see Table 19 above for details). When used during routine training and testing activities, and in a typical environment, these sources fall into one or more of the following categories:

- Transmit primarily above 200 kilohertz: Sources above 200 kilohertz are above the hearing range of the ESA-listed species in the action area.
- Source levels of 160 dB re 1 µPa or less: Low-powered sources with source levels less than 160 dB re 1 µPa are typically hand-held sonars, range pingers, transponders, and acoustic communication devices. Assuming spherical spreading for a 160 dB re 1 µPa source, the sound will attenuate to less than 140 dB re 1 µPa within ten meters and less than 120 dB re 1 µPa within 100 meters of the source. Ranges would be even shorter for a source less than 160 dB re 1 µPa source level. As discussed above (Section 2.2.3) we assume that sea turtles would not exhibit a behavioral response when exposed to such low source levels.
- Acoustic source classes listed in Table 19: Sources with operational characteristics, such as short pulse length, narrow beam width, downward-directed beam, and low energy release, or manner of system operation, which exclude the possibility of any significant impact on an ESA-listed species (actual source parameters are classified). Even if there is a possibility that some species may be exposed to and detect some of these sources, any response is expected to be short term and insignificant.

Therefore, these acoustic sources associated with NWTT activities (as described in Table 19) would either have no effect on leatherback sea turtles, or the effects would be insignificant, depending on the particular source considered.

8.1.2.2 Energy Stressors – Leatherback Sea Turtles

This section summarizes our analysis of the effects of energy stressors used during proposed NWTT activities on leatherback sea turtles within the action area. In Section 5.2 we described two energy stressors that could affect ESA-listed species within the action area: 1) in-water electromagnetic devices and 2) lasers. NWTT activities involving in-water electromagnetic devices would occur only inside Puget Sound or the Strait of Juan de Fuca, in Washington State where leatherback sea turtles are not expected to occur. Therefore, this stressor would have no

effect on leatherback sea turtles. Below we discuss the potential effects of lasers on leatherback sea turtles.

8.1.2.2.1 Effects of Lasers on Leatherback Sea Turtles

The maximum potential for laser exposure is at the ocean's surface, where laser intensity is greatest (U.S. Department of the Navy 2010). As the laser penetrates the water, 96 percent of a laser beam is absorbed, scattered, or otherwise lost (Ulrich 2004). An assessment on the use of low-energy lasers by the Navy determined that low-energy lasers, including those involved in the proposed NWTT activities, have an extremely low potential to impact any marine species (U.S. Department of the Navy 2010).

The primary concern with lasers used during Navy training and testing is the potential for a sea turtle to be struck by a high-energy laser beam. As discussed above, high-energy laser weapons testing involves the use of up to 30 kilowatts of directed energy as a weapon against small surface vessels and airborne targets. These weapons systems are deployed from surface ships and helicopters to create small but critical failures in potential targets and used at short ranges from the target (Navy 2018b). Traumatic burns from the high-energy beam could result in injury or death of a sea turtle. Sea turtles could only be exposed to the beam if the laser missed the target and inadvertently hit a sea turtle was located near the target. If this were to occur it would likely be for turtles located at or near the surface. The following safeguards are in place to reduce the probability of the a high-energy laser striking the water: 1) the high-energy laser platform has provisions that prevent misfiring (i.e., firing when not intended) that all but eliminate the possibility of that event, 2) the high-energy laser platforms have built-in constraints that only permit firing when it is locked onto a target, and 3) the operators are trained to stop firing when the laser aim point moves off of the selected target. Laser platforms are typically on helicopters and ships, which may cause sea turtles to move away from the area for reasons such as ship or aircraft noise, making a strike from the laser beam less likely.

The potential for sea turtles to be directly hit by a high-energy laser beam was evaluated by the Navy using statistical probability modeling to estimate the potential direct strike exposures to a sea turtle for a worst-case scenario. We adopt the Navy's quantitative analysis as the best available information for analyzing the effects of high-energy laser weapons on sea turtles. Model input values include high-energy laser use data (e.g., number of high-energy laser exercises and laser beam footprint), size of the training or testing area, sea turtle density data, and animal footprint. To estimate the probability of hitting a sea turtle in a worst-case scenario (based on assumptions listed below), the impact area for all laser training and testing events was summed over one year. Conservative assumptions incorporated into the Navy's model are as follows:

- The model is two-dimensional and assumes that all animals would be at or near the surface 100 percent of the time.

- The model assumes the animal is stationary and does not account for any movement of the sea turtles or any potential avoidance of the training or testing activity.

The Navy's model estimates no annual exposures of any sea turtles species (or DPS). Based on the Navy's quantitative analysis, the characteristics of activities that would use high-energy laser weapons (e.g., short range distance from source to target, high-precision targeting, short duration of the energized beam), and likely avoidance behavior of stressors, NMFS considers it extremely unlikely that leatherback sea turtles would be exposed to high energy lasers. Therefore, potential effects from lasers on leatherback sea turtles are discountable.

8.1.2.3 Physical Disturbance and Strike Stressors – Leatherback Sea Turtles

Additional discussion on physical disturbance and strike stressors is included in Section 5.3. This section summarizes our analysis of the potential impacts of the various types of physical disturbance, including the potential for strike, during training and testing activities within the action area from vessels, in-water devices, military expended materials (including non-explosive practice munitions and fragments from high-explosive munitions), and seafloor devices.

8.1.2.3.1 Effects of Vessel Strike on Leatherback Sea Turtles

Additional information on vessel strike as a potential stressor associated with the proposed action can be found in Section 5.3.1.

Leatherback sea turtles are not expected to be found in the inland portions of the action area where Navy vessel traffic would be concentrated. Although possible, leatherback vessel strike is unlikely to occur offshore given the anticipated low offshore density (estimated at 0.000114 leatherbacks per km^2) and sporadic, widely dispersed Navy vessel traffic throughout the vast offshore area. In addition, there has never been a documented case of a leatherback vessel strike by a Navy vessel in the NWTT action area. In summary, we find that the likelihood of a leatherback sea turtle vessel strike as a result of the proposed action is sufficiently low as to be extremely unlikely, and thus considered discountable.

8.1.2.3.2 Effects of Military Expended Materials on Leatherback Sea Turtles

Additional information on military expended materials as a potential stressor associated with the proposed action can be found in Section 5.3.2.

While no strike of sea turtles from Navy military expended materials has ever been reported or recorded, the possibility of a strike still exists. We considered the potential for leatherback sea turtle strike resulting from NWTT activities involving the following types of military expended materials: 1) all sizes of non-explosive practice munitions, 2) fragments from high-explosive munitions, 3) expendable targets and target fragments; and 4) expended materials other than munitions, such as sonobuoys, expended bathythermographs, and torpedo accessories.

Leatherback sea turtles are expected to be widely distributed in the offshore portion of the action area. While disturbance or strike from any expended material as it falls through the water column is possible, it is extremely unlikely because the objects will slow in velocity as they sink toward

the bottom (e.g., guidance wires sink at an estimated rate of 0.2 meters per second; heavier items such as non-explosive munitions would likely sink faster, but would still be slowed as they sink to the bottom), and can be avoided by highly mobile species such as sea turtles. The anticipated low offshore density (estimated at 0.000114 per km^2) of leatherbacks in the action area further decreases the likelihood of a strike from military expended materials. In addition, the Navy has proposed procedural mitigation for vessel movement and towed-in water devices to limit the potential for strikes of sea turtles where military expended materials are used in offshore environments (see Sections 0 and 3.5.1.12 for details).

In summary, NMFS considers it extremely unlikely for a leatherback sea turtle to be struck by military expended materials. Any individuals encountering military expended materials as they fall through the water column are likely to move to avoid them. Given the effort expended by individuals to avoid them will be minimal (i.e., a few meters distance) and temporary, behavioral avoidance of military expended materials sinking through the water column is likely inconsequential to an individual sea turtle. For these reasons, we find the potential effects from physical disturbance and strike with military expended materials for leatherback sea turtles are discountable (in the case of strikes) or insignificant (in the case of behavioral response) and thus not likely to cause adverse effects.

8.1.2.3.3 Effects of In-water Devices on Leatherback Sea Turtles

Additional information on in-water devices as a potential stressor associated with the proposed action can be found in Section 5.3.3.

Despite thousands of Navy exercises in which in-water devices have been used, there have been no recorded instances of marine species strikes from these devices. Most devices do not have a realistic potential to strike living marine species because they either move slowly through the water column (e.g., most unmanned underwater vehicles) or are closely monitored by observers manning the towing platform who ensure the towed in-water device does not run into objects in the water. The Navy will implement mitigation to avoid potential impacts from in-water device strikes on sea turtles throughout the action area. Mitigation includes training Lookouts and watch personnel to identify sea turtles and requiring underway vessels and in-water devices that are towed from manned surface platforms to maintain a specified distance from sea turtles. For these reasons, NMFS considers it extremely unlikely for any ESA-listed sea turtle to be struck by an in-water device. It is possible that sea turtles that occur in areas that overlap with in-water device use may experience some level of physical disturbance, but it is not expected to result in more than a momentary behavioral response. Any avoidance behavior would be of short duration and intensity such that it would be minor and temporary for the animal. Therefore, the potential effects on leatherback sea turtles from an in-water device strike are extremely unlikely to occur and therefore considered discountable. The potential effects on leatherback sea turtles from physical disturbance caused by in-water devices are insignificant, and thus not likely to cause adverse effects.

8.1.2.3.4 Effects of Seafloor Devices on Leatherback Sea Turtles

Additional information on seafloor devices as a potential stressor associated with the proposed action can be found in Section 5.3.4.

The types of activities that use seafloor devices include items placed on the seafloor, dropped on the seafloor, or that move along the seafloor such as mine shapes, anchor blocks, anchors, bottom-placed instruments, and bottom-crawling unmanned underwater vehicles. There have been no known instances of physical disturbance or strike to any sea turtles as a result of training and testing activities involving the use of seafloor devices in the NWTT action area. The likelihood of a leatherback sea turtle encountering seafloor devices is considered low because these items are either stationary or move very slowly along the ocean bottom. Sea turtles would be expected to ignore or avoid any slowly moving or stationary device. In addition, leatherback would generally not be anywhere near the seafloor in the areas where Navy seafloor devices would typically be deployed. Considering the extremely low probability of occurrence, we considers it extremely unlikely for a leatherback sea turtles to be exposed to seafloor devices as part of the proposed action. Therefore, potential effects on leatherback sea turtles from seafloor devices are considered discountable.

8.1.2.4 Entanglement Stressors – Leatherback Sea Turtles

Leatherback sea turtles could encounter expended materials used during NWTT activities that may result in entanglement. Many factors influence the degree of entanglement risk for sea turtles such as and life stage and size, sensory capabilities, and foraging methods (i.e. along the seafloor or in the water column). Similar to other marine animals, most entanglements associated with sea turtles are from materials that float or are suspended at the ocean's surface for long periods of time, such as fishing gear. This is particularly true for leatherback turtles which feed almost exclusively on soft-bodied invertebrates (i.e., jellyfish and tunicates) floating in the water column.

Entanglement stressors associated with the proposed action (i.e., wires, cables, decelerators, and parachutes) are discussed further in Section 5.4. Turtles could encounter these items at the water's surface, in the water column, or along the seafloor. Biodegradable polymers, which are also discussed as an entanglement stressor (Section 5.4.3), would only be used in the Inland Waters portion of the action area. Because the occurrence of leatherback sea turtles in inland waters is so rare, the likelihood of this species becoming entangled in biodegradable polymers is extremely unlikely and thus discountable. Therefore, biodegradable polymers are not discussed further as a potential stressor for sea turtles.

8.1.2.4.1 Effects of Entanglement in Cables and Wires on Leatherback Sea Turtles

Expended fiber optic cables, which range in size up to 3,000 meters in length, can pose a potential entanglement risk for sea turtles. However, because expended fiber optic cables sink rapidly and are not expected to remain suspended in the water column for long periods, the

likelihood of a leatherback sea turtle at the surface or in the water column encountering them is low. In addition, the material from these cables is very brittle and breaks easily if bent or twisted, which also decreases the likelihood that a turtle would become ensnared.

Similar to fiber optic cables, guidance wires in the water column may pose an entanglement threat to leatherback sea turtles. However, the likelihood of a leatherback sea turtle encountering and becoming entangled in a guidance wire is low. The sink rate to the seafloor (at an estimated rate of 0.2 meters per second) is fast, and the likelihood of a sea turtle encountering a guidance wire as it descends is very low. Also, similar to fiber optic cables, guide wires have a relatively low tensile breaking strength (between 10 and 42 pounds) which further reduces the entanglement risk for sea turtles.

Sonobuoy wires, consist of a thin-gauge, hard draw copper strand wire, wrapped by a hollow rubber tubing or bungee. The tensile breaking strength of the sonobuoy wire and rubber tubing is no more than 40 pounds Operationally, sonobuoys can remain suspended in the water column for up to 30 hours, after which they sink to the seafloor, which would increase the likelihood that a leatherback sea turtle could encounter a sonobuoy wire either while it is suspended or as it sinks (Navy 2017). However, as with fiber optic wires, sonobuoys are weak and likely to break if wrapped around a sea turtle. Bathythermograph wires are similar to sonobuoys, and expected to have the same fate, as such are expected to pose little risk for sea turtles.

Given the low concentration of expended wires and cables, the rapid sink rates, the relatively low tensile breaking strength of these materials, and the low estimated density of leatherback sea turtles in the action area, the likelihood of a leatherback sea turtle encountering a wire or cable and becoming entangled is extremely low. Based on the extremely low probability of occurrence, we considers it extremely unlikely for any sea turtles to be exposed to entanglement in cables and wires as part of the proposed action. Therefore, potential effects to leatherback sea turtles from entanglement in cables and wires expended during NWTT activities are considered discountable.

8.1.2.4.2 Effects of Entanglement in Decelerators and Parachutes on Leatherback Sea Turtles

The large majority of the decelerators and parachutes used for MITT activities are in the small size category and are associated with sonobuoys (i.e., 11,337 out of 11,439 used annually). Both small- and medium-sized decelerators and parachutes are made of cloth and nylon and have weights attached to their short attachment lines (i.e., from 0.3 to 5.8 meters) to speed their sinking to the seafloor. Thus, the large majority of parachutes and decelerators would not remain suspended in the water column for more than a few minutes. Small and medium decelerators and parachutes with weights are expected to remain at the surface for 5 to 15 seconds before the housing sinks to the seafloor where it becomes flattened (Navy 2019c).

A smaller number (i.e., 98 per year) of large decelerators/parachutes are also proposed for use in the action area. In contrast to small and medium parachutes, large parachutes do not have weights attached and may remain at the surface or suspended in the water column for some time prior to eventual settlement on the seafloor. The relatively small number of large parachutes proposed for use annually, and generally low leatherback sea turtle density, reduces the potential for turtles to encounter and become entangled in these items. Since leatherback are known to forage on jellyfish at or near the surface, exposure would involve either the decelerator or parachute landing directly on the turtle or the turtle swimming into it before it sinks. The likelihood of this occurring is very low. During activities that involve recoverable targets (e.g., aerial drones), the Navy recovers the target and any associated decelerators/parachutes to the maximum extent practicable consistent with personnel and equipment safety. This standard operating procedure could further reduce the potential entanglement of leatherbacks in large decelerators/parachutes.

Overall, given the low probability of a leatherback sea turtle being in the vicinity of a deployed large decelerator or parachute, and the short amount of time that small/medium decelerators/parachutes would be in the water column where leatherbacks could be exposed, we find the likelihood of entanglement to be extremely low. Therefore, the potential effects to leatherback sea turtles from entanglement in decelerators and parachutes are considered extremely unlikely and thus discountable.

8.1.2.5 Ingestion Stressors – Leatherback Sea Turtles

Additional information on ingestions stressors associated with the proposed action can be found in Section 5.5.

The munitions and other expended materials that we consider small enough to be ingested by leatherback sea turtles are small and medium caliber projectiles (up to 2.25 inches), broken pieces of firing targets, chaff, flare casings (caps and pistons), decelerators and parachutes (cloth, nylon and metal weights) and shrapnel fragments from high-explosives ordnance. Types of munitions that can result in fragments small enough to be ingested include demolition charges, projectiles, missiles, and bombs. The size of these fragments would vary depending on the NEW and munitions type. Other munitions and munitions fragments such as large-caliber projectiles or intact training and testing bombs are too large for sea turtles to consume.

Most expendable materials would be used over deep water and are expected to sink quickly and settle on the seafloor, with the exception of chaff and some firing target materials (Navy 2018b). Because they typically forage in the water column either at or near the surface, it is unlikely that leatherback sea turtles would be susceptible to ingesting expended materials that sink quickly to the bottom.

Chaff is composed of fine fibers of silicon dioxide coated with aluminum alloy. Due to their light weight and small size, chaff float and can be carried great distances in both air and water currents

(Navy 2018b). Their dispersal in wind and water results in chaff fibers likely occurring in low densities on the ocean surface. Chaff recovery studies conducted within chaff release areas within several U.S. military operating areas have shown that the concentrations of chaff dipoles (fibers) in the environment are very small (Farrell and Siciliano 2004). The amount of chaff needed to raise environmental concentrations of these metals above background levels far exceeds the number than can be realistically deposited in the open ocean (Farrell and Siciliano 2004). As such, chaff releases are not expected to have any significant effect on marine ecosystem functioning (Farrell and Siciliano 2004).

Chaff fibers are too small for sea turtles to confuse with prey and forage, but there is the possibility that sea turtles could come in contact or accidentally ingest some of the chaff material. Given the low concentration that would be ingested, the small size of the fibers, and the anticipated low toxicity (Arfsten et al. 2002), any accidental ingestion of chaff by leatherback sea turtles feeding at the ocean surface is not expected to result in an injury or an increased likelihood of injury from a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Firing target materials, which may also float on the surface, are normally retrieved before sinking so it is unlikely that leatherback sea turtles would ingest such materials (Navy 2018b).

Chaff cartridge plastic end caps and pistons and flare pads and pistons would also be released into the marine environment during NWTT activities. These materials may persist in the environment for long periods, and therefore could be ingested by sea turtles while initially floating on the surface or sinking through the water column (Navy 2018b). These materials would eventually sink to the seafloor where they would be less likely to be ingested by leatherback sea turtles that forage at or near the surface.

The chances of a sea turtle ingesting expended materials in the water column increase if it is within close proximity to falling munitions, mistakes a sinking munition for prey, and reacts quickly enough to ingest the sinking material. This is an unlikely scenario given their feeding habits and low density of leatherbacks (i.e., estimated at 0.000114 per km^2) in the action area. The likelihood of this occurring would be further reduced by the Navy's mitigation measures, such as avoiding mats of floating vegetation and having Lookouts posted to detect sea turtle presence in the area prior to discharging weapons (Navy 2018b).

We have no information indicating that military expended materials have been found in sea turtles that have been necropsied, unlike plastics that appear similar to jellyfish or other turtle prey and are found in a large proportion of sea turtles worldwide (Schuyler et al. 2016). Sea turtles may attempt to ingest a projectile fragment and then reject it, after realizing it is not a food item. If material is ingested, most ingestible-sized items would likely be spit out or passed through the digestive tract without significantly impacting the individual. Therefore, negative impacts of fragment ingestion may be limited to the unlikely event of an item that becomes embedded in tissue or is too large to be passed through the digestive system. If the material or

fragment is particularly large in proportion to the turtle ingesting it, the item could become permanently encapsulated in the stomach lining and, although rare, could impede the turtle's ability to feed or take in nutrients. However, the likelihood of this occurring would be low. In addition, given the anticipated wide dispersal of expended materials (other than munitions) throughout the action area, and the short duration of time these military expended materials would remain in the water column, the probability of a leatherback sea turtle encountering these materials is low. Therefore, adverse effects resulting from the ingestion of expended materials is considered extremely unlikely and thus the effects on leatherback sea turtles from ingestion are discountable.

8.1.2.6 Stressors Resulting in Effects to Leatherback Sea Turtle Habitat or Prey

Stressors from training and testing activities that could result in secondary effects on leatherback sea turtles via impacts to habitat, prey, and water quality include explosives, metals, chemicals, and other expended materials.

Underwater explosions could impact other species in the food web, including prey species that ESA-listed species feed upon. The abundance of prey near the detonation point could be diminished for a short period of time, before being repopulated by animals from adjacent waters. Any of these scenarios would likely be short-term and temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web expected. In addition, the impacts of explosions would differ depending on the type of prey species in the area of the blast. For the reason state above, the indirect effects of explosives on leatherback sea turtles via impacts to their prey are insignificant.

Leatherback sea turtles could be exposed to metals introduced into the water column as a result of NWTT activities involving ship hulks, targets, munitions, and other military expended materials. Some metals bioaccumulate and physiological impacts begin to occur only after several trophic transfers concentrate the toxic metals. A variety of heavy metals have been found in sea turtles tissues in levels that increase with turtle size. These include arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, and zinc, (Barbieri 2009; Fujihara et al. 2003; García-Fernández et al. 2009; Godley et al. 1999; Storelli et al. 2008). Cadmium has been found in leatherbacks at the highest concentration compared to any other marine vertebrate (Gordon et al. 1998). However, biomagnification of trace elements via trophic transfer might be limited in leatherbacks due to their lower trophic level diet of cnidarian zooplankton (Harris et al. 2011).

Evidence from a number of studies indicate metal contamination is highly localized and that bioaccumulation resulting from munitions cannot be demonstrated (Briggs et al. 2016; Edwards and coauthors. 2016; Kelley et al. 2016; Koide et al. 2016; Navy 2013c). Due to the extremely low concentrations of metals resulting from the proposed action in the open ocean, it is extremely unlikely that leatherback sea turtles or their prey would be indirectly impacted by toxic metals via the water. Since leatherbacks do not forage at or near the seafloor,

contamination from metals accumulated in sediments is also extremely unlikely to occur. Therefore, the effects of metals on leatherback sea turtles introduced into seawater and sediments as a result of the proposed action are discountable.

Navy training and testing activities also introduce chemicals into the marine environment that are potentially harmful in higher concentrations. However, rapid dilution would be expected and toxic concentrations are unlikely to be encountered by leatherback sea turtles or their prey. Chemicals introduced are principally from flares and propellants for missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment. Flares and missiles that operationally fail may release perchlorate, which is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals if in sufficient concentration. However, such concentrations would be localized and are not likely to persist in the ocean. Research has demonstrated that perchlorate does not bioconcentrate or bioaccumulate, which is consistent with the expectations for a water-soluble compound (Furin et al. 2013). Given the dynamic nature of the environment (currents, tides, etc.), long-term impacts from perchlorate in the environment near the expended item are not expected. It is extremely unlikely that perchlorate from failed expendable items would compromise water quality to the point that it would result in adverse effects on ESA-listed sea turtle prey or habitat.

Chemical and biological simulants used during simulant testing could affect water quality, leatherback prey and lower trophic levels within the food web of leatherback sea turtles. All chemical simulants proposed for use by the Navy would have low toxicity to humans and the environment and all biological simulants are considered to be Biosafety Level 1 organisms (Navy 2020e). While some minor effects of chemical and biological simulants on water quality and prey are possible, such effects, if they occur at all, would be extremely short-term and localized. Simulants released in the offshore portion of the action area would be expected to disperse rapidly into the ocean environment. Therefore, even if simulant testing resulted in adverse effects to prey and water quality, it is extremely unlikely that such secondary effects would result in adverse effects on leatherbacks.

In summary, the effects of chemicals and simulants (chemical and biological) used during Navy training and testing on leatherback sea turtles via water quality and prey are considered discountable.

8.1.3 Fishes

We determined that several of the acoustic stressors and all of the energy stressors, entanglement stressors, ingestion stressors, and potential secondary stressors are not likely to adversely affect the following fish species: ESA-listed chinook salmon (all ESUs listed in Table 64), steelhead (all DPSs listed in Table 64), coho salmon (all ESUs listed in Table 64), Columbia River ESU and Hood Canal Summer-Run ESU of chum salmon, Snake River ESU of sockeye salmon, Puget Sound/Georgia Basin DPS of bocaccio and yelloweye rockfish, Southern DPS eulachon, and

Southern DPS green sturgeon. As noted above, our analysis for these stressors is organized on the taxa level (i.e., fishes) because the pathways for effects for these stressors is generally similar for all fishes, and we would not expect different effects at the species level. While there is variation among species within each taxa, the fish species considered in this consultation share many similar life history patterns and other factors (e.g., morphology) which make them similarly vulnerable (or not) to the stressors associated with the proposed action. Where species-specific information is relevant, this information is provided in this section. Our analysis for these stressors and fishes is summarized below.

8.1.3.1 Acoustic Stressors – Fishes

The discussion below focuses on a subset of the acoustic stressors associated with the proposed action. We determined that these acoustic stressors are not likely to adversely affect ESA-listed fishes. The effects of explosives, which we determined was likely to adversely affect ESA-listed fishes, is discussed in Section 8.2.3 below.

8.1.3.1.1 Effects of Vessel Noise on ESA-listed Fishes

Navy vessel movements involve transits to and from ports to various locations within the action area, and many proposed activities within the action area involve maneuvers by various types of surface ships, boats, and submarines (collectively referred to as vessels), as well as unmanned vehicles. Training and testing events involving vessel movements occur intermittently and range in duration from a few hours up to a few weeks. Navy vessel traffic could occur anywhere within the action area, but is concentrated in proximity to the naval ports at Everett and Bremerton. See Section 5.1.3 for a general discussion of vessel noise as a potential stressor associated with the proposed action.

Individuals from all ESA-listed fishes may be exposed to sound from vessel movement during Navy training and testing activities. In general, information regarding the effects of vessel noise on fish hearing and behaviors is limited. Although some TTS has been observed in fishes exposed to elevated background noise and other white noise, a continuous sound source similar to noise produced from vessels. For example, caged studies on sound pressure sensitive fishes show some TTS after several days or weeks of exposure to increased background sounds, although the hearing loss appeared to recover (e.g., Scholik and Yan 2002; Smith et al. 2006; Smith et al. 2004). Smith et al. (2004) and Smith et al. (2006) exposed goldfish (a fish with hearing specializations, unlike any of the ESA-listed species considered in this opinion) to noise with an SPL of 170 dB re 1 μPa and found a clear relationship between the amount of TTS and duration of exposure, until maximum hearing loss occurred at about 24 hours of exposure. A short duration (e.g., 10-minute) exposure resulted in 5 dB of TTS, whereas a three-week exposure resulted in a 28 dB TTS that took over two weeks to return to pre-exposure baseline levels (Smith et al. 2004). Recovery times were not measured by researchers for shorter exposure durations, so recovery time for lower levels of TTS was not documented.

Vessel noise may also affect fish behavior by causing them to startle, swim away from an occupied area, change swimming direction and speed, or alter schooling behavior (Engas et al. 1998; Engas et al. 1995; Mitson and Knudsen 2003). Physiological responses have also been documented for fish exposed to increased boat noise. Nichols et al. (2015) demonstrated physiological effects of increased noise (playback of boat noise) on coastal giant kelpfish. The fish exhibited stress responses when exposed to intermittent noise, but not to continuous noise. These results indicate variability in the acoustic environment may be more important than the period of noise exposure for inducing stress in fishes. However, other studies have also shown exposure to continuous or chronic vessel noise may elicit stress responses indicated by increased cortisol levels (Scholik and Yan 2001; Wysocki et al. 2006). These experiments demonstrate physiological and behavioral responses to various boat noises that could affect species' fitness and survival but may also be influenced by the context and duration of exposure. It is important to note that most of these exposures were continuous, not intermittent, and the fish were unable to avoid the sound source for the duration of the experiment because this was a controlled study. In contrast, wild fish are not hindered from movement away from an irritating sound source, if detected, they are less likely to be subjected to accumulation periods that lead to the onset of hearing damage as indicated in these studies. In other cases, fish may eventually become habituated to the changes in their soundscape and adjust to the ambient and background noises.

All fish species can detect vessel noise due to its low-frequency content and their hearing capabilities. Navy vessels produce moderate to low-level passive sound sources (larger Navy ships would produce low-frequency, broadband underwater sound below 1 kilohertz; and smaller vessels emit higher-frequency sound between 1 kilohertz to 50 kilohertz). As a result, ESA-listed fishes could be exposed to a range of vessel noises, depending on the source and context of the exposure. Because of the characteristics of vessel noise, sound produced from Navy vessels is unlikely to result in direct injury, hearing impairment, or other trauma to fishes. Plus, in the near field, fish are able to detect water motion as well as visually locate an oncoming vessel. In these cases, most fishes located in close proximity that detect the vessel either visually, via sound and motion in the water would be capable of avoiding the vessel or move away from the area affected by vessel sound. Thus, fish are more likely to react to vessel noise at close range than to vessel noise emanating from a greater distance away. These reactions may include physiological stress responses, or avoidance behaviors. Auditory masking due to vessel noise can potentially mask vocalizations and other biologically important sounds that fish may rely on. Navy vessel noise would be intermittent, temporary and localized, and such responses would not be expected to compromise the general health or condition of individual fish from continuous exposures. The only impacts expected from exposure to Navy vessel noise for fishes may include temporary auditory masking, physiological stress, or minor changes in behavior.

Therefore, exposure to vessel noise for fishes could result in short-term behavioral or physiological responses (e.g., avoidance and stress). Vessel noise would only result in brief periods of exposure for fishes and would not be expected to accumulate to the levels that would

lead to any injury, hearing impairment or long-term masking of biologically relevant cues. For these reasons, exposure to vessel noise is not expected to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, the likely effects of vessel noise on ESA-listed salmonids, rockfish, eulachon, and sturgeon in the action area are considered insignificant and thus not likely to cause adverse effects.

8.1.3.1.2 Effects of Aircraft Noise on ESA-listed Fishes

See Section 5.1.4 for a general discussion of aircraft noise as a potential stressor associated with the proposed action.

Individuals from all ESA-listed fishes may be exposed to aircraft-generated overflight noise throughout the action area. Should sound transmit from aircraft travel into the water column, it would likely only penetrate shallow depths and would be below the range of any injury criteria for fishes. Furthermore, aircraft quickly pass overhead, with helicopters potentially hovering for a few minutes or up to a few hours over the water's surface. Sound transmission into deep depths of the water column is not likely, and sound that is transferred into the water from air is only within a narrow cone under the aircraft. Therefore, only fishes located at or near the surface of the water and within the limited area where transmission of aircraft noise is expected to occur have the potential to detect any noise produced from low-flying aircraft.

Direct injury and hearing impairment in fishes is unlikely to occur from aircraft overflight noise, because sounds from aircraft noise, including occasional sonic booms, lack the amplitude or duration to cause any physical damage to fishes underwater. Furthermore, due to the brief and dispersed nature of aircraft overflights, masking of biologically relevant sounds for fishes is also extremely unlikely. In the rare circumstance a fish detects sound produced from an aircraft overhead, only very brief startle or avoidance responses would be expected. Additionally, due to the short-term, transient nature of aircraft noise, ESA-listed fishes are unlikely to be exposed multiple times within a short period of time that could lead to ongoing behavioral disruptions or stress. Any physiological stress and behavioral reactions would be short-term (seconds or minutes) and are expected to return to normal shortly after the aircraft disturbance ceases. Therefore, the effects on fishes from aircraft overflight noise are anticipated to be minor, temporary and will not lead to a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. As such the effects from aircraft overflight noise on ESA-listed salmonids, rockfish, eulachon, and sturgeon in the action area are considered insignificant and thus not likely to cause adverse effects.

8.1.3.1.3 Effects of Weapons Noise on ESA-listed Fishes

See Section 5.1.5 for a general discussion of weapons noise as a potential stressor associated with the proposed action.

ESA-listed fishes at the surface of the water could be exposed to weapons noise, in a narrow footprint under a weapons trajectory. In general, these are impulsive sounds generated in close

vicinity to or at the water surface, with the exception of items that are launched underwater. In addition, any objects that are dropped and impact the water with great force could produce a loud broadband sound at the water's surface from large-caliber non-explosive projectiles, non-explosive bombs, and intact missiles and targets (Mclennan 1997).

Naval gunfire could also elicit a brief behavioral reaction such as a startle response or avoidance and could expose fishes to multiple shots within a few seconds. The firing of a weapon may have several components of associated noise including sound generated in air by firing a gun (muzzle blast) and a crack sound due to a low amplitude shock wave generated by a supersonic projectile flying through the air. Most in-air sounds would be reflected at the air-water interface. Underwater sounds would be strongest just below the surface and directly under the firing point. The sound produced from missile and target launches is typically at a maximum during initiation of the booster rocket, but rapidly fades as the missile or target travels downrange; therefore this noise is unlikely to affect fishes underwater. These are launched from aircraft which would produce minimal sound in the water due to the altitude of the aircraft when these are fired.

For exposed fishes, most of the weapons noise produced from these activities lack sound characteristics such as duration and high intensity that would accumulate or cause mortality, injury, or hearing impairment. Additionally, because these activities are brief in duration and widely dispersed throughout the action area, accumulation of levels high enough to cause TTS or masking of biologically relevant sound for fishes is also extremely unlikely. As with the other stressors for fishes discussed in this section, exposure to the sound produced from weapons would only be expected to cause brief behavioral or stress responses should they detect the noise. Fish may react by exhibiting startle responses, rapid bursts in movement, changes in swimming direction or orientation, or leaving the immediate area of the sound. Concurrent with these behavioral responses, fishes could also experience temporary increases in heart rate or stress hormones. However, any behavioral reactions and physiological stress would likely be brief, and are expected to return to normal shortly after the weapons noise ceases. Therefore, the effects on ESA-listed fishes in the action area from weapons noise are anticipated to be minor, temporary, and are not expected to lead to a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. As such, the effects from weapons noise on ESA-listed salmonids, rockfish, eulachon, and sturgeon are considered insignificant and thus not likely to cause adverse effects.

8.1.3.1.4 Effects of Sonar and Transducers on ESA-Listed Fishes

General categories and characteristics of Navy sonar systems proposed for use during activities considered in this biological opinion are described in Sections 3.2 and 5.1.1.

All ESA-listed fishes considered in this opinion have the potential to be exposed to sonar and other transducers during Navy activities in the action area. However, direct injury from sonar and other transducers is considered unlikely. These types of sound sources are considered to pose less risk to fish species because the sound produced from sonar characteristically has lower peak

pressures and slower rise times than other acoustic stressors that are known to injure fish such as impulsive sounds from pile driving, or the strong shock waves produced from detonation of explosives. Direct injury from sound levels produced from the type of sonar the Navy uses has not been documented in fishes (Halvorsen et al. 2012; Kane et al. 2010; Popper et al. 2014a; Popper et al. 2007b; Popper et al. 2013). However, some short term hearing impairment could occur, as well as brief behavioral and stress responses which are discussed below.

As described previously, fishes are not equally sensitive to noise at all frequencies. Some species of fishes have specialized adaptations which increases their ability to detect sounds at higher frequencies. However, none of the ESA-listed fishes that may be affected by Navy activities possess any hearing specializations and all hear primarily below 2 kilohertz. For these reasons, grouping fish according to the presence of a swim bladder and whether or not that swim bladder is involved in hearing and their known hearing frequency ranges (audiograms) was determined to be the best approach for our analyses. Of the ESA-listed fish species considered in this opinion that possess a swim bladder, none have a swim bladder that is associated with hearing. Thus, the sound criteria used for our analysis of the acoustic effects on fishes are based upon 1) fishes with swim bladders not involved in hearing (i.e., salmonids, rockfish, and sturgeon), and 2) fishes that do not possess a swim bladder (Pacific eulachon).

Exposure to SURTASS low-frequency active sonar has been tested at maximum received levels of 193 dB re 1 µPa (218 dB SEL_{cum}) and has not been shown to cause mortality or any injury in fish with swim bladders (Kane et al. 2010; Popper et al. 2007b). The researchers exposed three freshwater species of fish, the rainbow trout (*Oncorhynchus mykiss*), channel catfish (*Ictalurus punctatus*), and the hybrid sunfish (*Lepomis sp*.), to both low- and mid-frequency sonar. Low-frequency active sonar exposures with received SPLs of 193 dB re 1 µPa occurred for either 324 or 648 seconds. Although this laboratory study exposed the fish to low-frequency active sonar pulses for time intervals that would be substantially longer than what would occur in the wild (e.g., unconfined fishes), the exposed fish did not experience mortality or damage to body tissues at the gross or histological level. Hearing was measured both immediately post-exposure and for several days thereafter. Catfish and some specimens of rainbow trout showed a temporary hearing loss of 10 to 20 dB immediately after exposure to high intensity low-frequency active sonar when compared to baseline and control fish; however, another group of rainbow trout showed no hearing loss. Recovery in trout took at least 48 hours, but studies on recovery were not completed. The reason for the different results between rainbow trout groups is not known, although the researchers speculated it may be due to developmental or genetic differences in the various groups of fish. Catfish hearing returned to, or close to, normal within about 24 hours after exposure to low-frequency active sonar. Furthermore, examination of the inner ears of the fish during necropsy revealed no differences from the control groups in ciliary bundles or other inner ear features indicative of hearing loss (Kane et al. 2010). Lesser potential for injurious effects would be expected for fish without swim bladders, because the presence of a swim

bladder increases risk of injury as the sound wave passes through a fish's body and causes the swim bladder to resonate with the sound frequency.

No studies have indicated any physiological damage to adult fish from mid-frequency sonar. However, one study on juvenile herring survival found that intense sonar exposures affected less than 0.3 percent of the total juvenile stock (Kvadsheim and Sevaldsen 2005). Similarly, Jorgensen et al. (2005) exposed larvae and juvenile Atlantic herring (*Clupea harengus*) Atlantic cod (*Gadus morhua*), saithe (*Pollachius virens*), and spotted wolffish (*Anarhichas minor*) to sounds that were designed to simulate mid-frequency active sonar transmissions (1 to 6.5 kilohertz) to study the effects of the exposure on the survival, development, and behavior. The fish were placed in plastic bags three meters from the sound source and exposed to between four and 100 pulses of one-second duration of pure tones at 1.5, 4, and 6.5 kilohertz. The fish in only two groups out of the 42 tested exhibited adverse effects beyond a behavioral response. These two groups were both composed of herring (a fish with hearing specializations), and were tested with SPLs of 189 dB re 1 µPa, which resulted in a post-exposure mortality of 20 to 30 percent. In the remaining 40 tests, there were no observed effects on behavior, growth (length and weight), or the survival of fish that were kept as long as 34 days post exposure. While statistically significant losses were documented in the two groups impacted, the researchers only tested that particular sound level once, so it is not known if this increased mortality was due to the level of the test signal or to other unknown factors. It is also important to note that none of the ESA-listed fish species considered in this biological opinion have the hearing specializations similar to herring. As such, the ESA-listed fishes evaluated in this opinion are not considered as sensitive to sound exposures and associated hearing damage as herring.

In another mid-frequency active sonar experiment, Halvorsen et al. (2012) exposed rainbow trout to simulated mid-frequency active (2.8 to 3.8 kilohertz) sonar at received SPLs of 210 dB re 1 uPa, resulting in cumulative SELs of 220 dB re 1 uPa. The researches did not observe any mortality or hearing sensitivity changes in rainbow trout and suggested that the frequency range of mid-frequency active sonar may be above the most sensitive hearing range of the species.

Some studies have suggested that there may be some loss of sensory hair cells due to high intensity sources; none of these studies concurrently investigated effects on hearing. Enger (1981) found loss of ciliary bundles of the sensory cells in the inner ears of Atlantic cod following one to five hours of exposure to pure tone sounds between 50 and 400 Hz with an SPL of 180 dB re 1 µPa. Similarly, Hastings (1995) found auditory hair-cell damage in a species with notable anatomical hearing specializations, the goldfish (*Carassius auratus*) exposed to 250 Hz and 500 Hz continuous tones with maximum peak levels of 204 dB re 1 µPa and 197 dB re 1 µPa, respectively. Compared to Navy sonar exposures anticipated, these were long duration exposures of about two hours in laboratory settings, much longer than any exposure a fish would encounter in the wild during the Navy's proposed activities (i.e., due to the transient nature of Navy sonar use and that fishes are not confined in the wild as they are in a laboratory setting).

The fish exposed in the lab were held in a cage for the duration of the exposure, unable to avoid the source.

Hastings et al. (1996) also demonstrated damage to some sensory hair cells in oscars (*Astronotus ocellatus*) following a 1-hour exposure to a pure tone at 300 Hz with a peak pressure level of 180 dB re 1 μPa. Although in none of the studies was the hair cell loss more than a relatively small percent (less than a maximum of 15 percent) of the total sensory hair cells in the hearing organs.

Hastings (1990) and Hastings (1995) also demonstrated 'acoustic stunning' (loss of consciousness) in blue gouramis (*Trichogaster trichopterus*) following an 8-minute exposure to a 150 Hz pure tone with a peak SPL of 198 dB re 1 μPa. This species of fish has an air bubble in the mouth cavity directly adjacent to the animal's braincase that may have caused this injury. The researchers also found that goldfish exposed to two hours of continuous wave sound at 250 Hz with peak pressures of 204 dB re 1 μPa, and fathead minnows exposed to 0.5 hours of 150 Hz continuous wave sound at a peak level of 198 dB re 1 μPa did not survive. The only study on the effect of exposure of the lateral line system to continuous sound was conducted on a freshwater species, and suggests no effect on these sensory cells by intense pure tone signals (Hastings et al. 1996).

The research described above, and the most recent literature review and summary completed by Popper et al. (2014a) regarding fish response to low-frequency active and mid-frequency active sonar indicate that those species tested to date can be used as viable surrogates for estimating injury in other species exposed to similar sources. The research conducted to date has not provided evidence that injury or mortality could occur from the sonar used by the Navy. Although fishes have been injured and killed due to intense, long duration, non-impulsive sound exposures, fish exposed under more realistic conditions have shown no signs of injury. Exposures would need to be of a much longer duration than those that would realistically occur with the Navy's proposed activities. In addition, the relative risk of injury or mortality to fish with no swim bladders exposed to low and mid-frequency sonar is lower than fish with swim bladders, no matter the distance from the source. The recommended criteria and thresholds in the 2014 ANSI Guidelines are used to predict potential impact to fishes from sonar and transducers. These criteria are shown in Table 8 above (see Section 2.3 *Criteria and Thresholds to Predict Impacts to Fishes*).

Because of the sheer number and diversity of fishes, only a limited number of species have had hearing capabilities tested. Figure 88 below, provides a summary of hearing threshold data from available literature to demonstrate the potential overall range of frequency detection for each hearing group. These estimated hearing ranges may be overly conservative in that they may extend beyond actual species hearing capabilities for a particular group. The upper bounds of each fish hearing group frequency range are outside of the range of best sensitivity for all fishes within that group. As a result, fishes within each group would only be able to detect those upper frequencies from sources with relatively high source levels. Figure 88 is not intended as a

composite audiogram, but rather displays the basic overlap in potential detectable frequencies for each fish hearing group associated with the Navy's defined sonar classes (i.e., low-, mid-, high- and very high-frequency) as discussed in Section 3.2 and above.

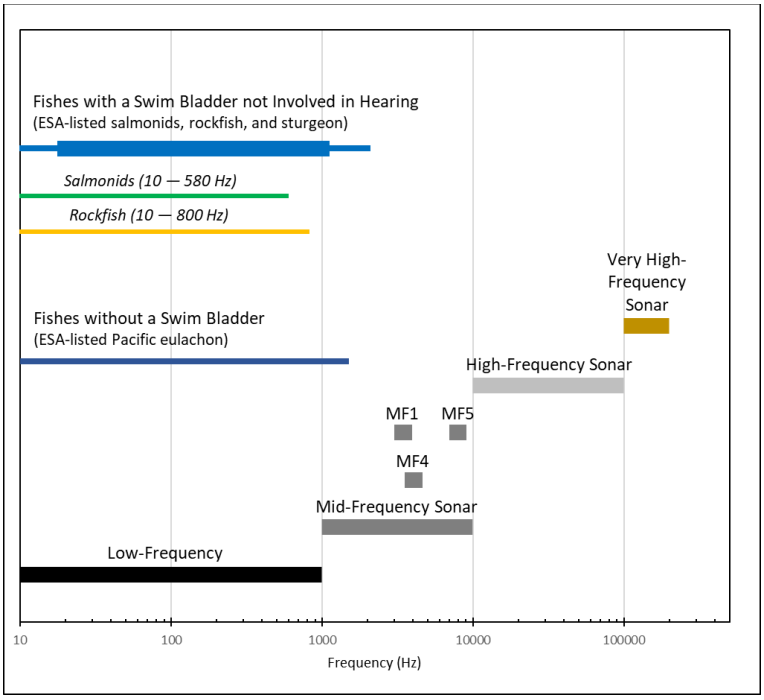


Figure 88. Frequency ranges of ESA-listed fishes affected by NWTT activities. Sources: (Hawkins and Johnstone 1978; Popper 2008; Popper et al. 2007a; Popper et al. 2014b; Tavolga and Wodinsky 1963).

Notes: Thin blue lines represent the estimated minimum and maximum range of frequency detection for each group. All hearing groups are assumed to hear down to 0.01 kilohertz regardless of available data. Thicker portions of each blue line represent the estimated minimum and maximum range of best sensitivity for that group. Currently, no data are available to estimate the range of best sensitivity for fishes without a swim bladder. When available, hearing data for specific ESA-listed species, or surrogate species, are provided (see green and yellow lines). Although the horizontal black, grey and brown bars represent each sonar class graphically, not all sources within each class would operate at all the displayed frequencies. Example mid-frequency classes are provided to further demonstrate this. kilohertz = kilohertz, MF1 = 3.5 kilohertz, MF4 = 4 kilohertz, MF5 = 8 kilohertz.

Based upon the fish hearing and frequency overlap, the ESA-listed fishes considered in this biological opinion would be able to detect most of the Navy sonars within the low-frequency active sonar ranges, and would have limited ability to detect mid-frequency active sonar frequencies. For example, both fish groups (with and without swim bladders) would not be able to detect mid-frequency active sonar sources within bins MF1, MF4, or MF5. Also, it is anticipated that most ESA-listed fishes would not be able to hear Navy sonars or other transducers with operating frequencies greater than about 1 to 2 kilohertz. Further, none of the fish species considered in this opinion can detect high- and very high-frequency sonars and other transducers. Therefore, these species would not be affected by high-frequency Navy sonar sources. As described above, mortality or injury from exposure to sonar is highly unlikely for the fish species potentially present in areas where the Navy will use sonar or other transducers. Thus, the most probable effects would be masking, physiological stress and behavioral responses. As

stated above, if TTS occurred it would likely only occur for fishes with swim bladders (salmonids, rockfish, and sturgeon). Fishes without swim bladders (Pacific eulachon) are not expected to sustain TTS from sonar exposure. However, as shown in Table 87, the majority of naval sonar activities will be within the mid-frequency range which fishes with swim bladders consulted on in this opinion are unlikely to hear.

To estimate the range to effects for fish exposed to sonar, the Navy calculated the range to effects based upon their NAEMO model and the respective hearing criteria. We use the Navy's range to effect calculations as the best available information for estimating the effects to fish from sonar. Sonar durations of 1, 30, 60 and 120 seconds were used in the range to effects calculations. Due to the relatively low source levels from this sonar source level and duration of sonar exposures, a range of zero meters was predicted for TTS. Therefore, it is unlikely that any fishes with a swim bladder not involved in hearing would experience TTS or any injury from exposure to Navy activities using sonar and other transducers. Although ranges to effect are predicted, the Navy did not estimates the total number of fishes that could be affected by sonar and other transducers using the NAEMO model, as was done for sea turtles and marine mammals.

Fishes that are able to detect low-frequency active sonar, and perhaps some mid-frequency active sonar, could experience brief periods of masking, or exhibit brief behavioral reactions and stress responses. Fish located closer to the sonar sound source would likely experience more substantive responses, whereas fish located further away from the source are less likely to react to the sound levels. However, because the Navy's sonar is moving, and fish are also capable of moving away from the disturbance, the overall exposure duration is expected to be brief. If masking did occur, it would not occur for a significant amount of time, nor would it likely prevent fish from detecting biologically relevant cues at meaningful levels. Additionally, any physiological stress responses or behavioral reactions would also be expected to be temporary, lasting only a few seconds or minutes during sonar pings. For these reasons, no long-term consequences for any exposed ESA-listed fish are expected, and the effects described above are not anticipated to lead to a significant disruption of normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. As such, the effects from sonar and other transducers on ESA-listed salmonids, rockfish, eulachon, and sturgeon are considered insignificant and thus not likely to cause adverse effects.

8.1.3.2 Energy Stressors – Fishes

This section analyzes the effects of energy stressors used during training and testing activities on fishes within the action area. For a general discussion of energy stressors associated with the proposed action see Section 5.2. This section describes the analysis of the potential impacts of in-water electromagnetic devices and high-energy laser weapons.

8.1.3.2.1 Effects of In-water Electromagnetic Devices on Fishes

A synthesis of information provided by Normandeau et al. (2011) provides a comprehensive review of information regarding the sensitivity of marine organisms to electric and magnetic

impulses. Available data suggests that while many fish species are sensitive to electromagnetic fields (Hore 2012), more research is necessary to understand the physiological response and magnitude of the potential impacts from these sources on fish.

Many fish groups have been demonstrated to have an acute sensitivity to electrical fields, known as electroreception (Bullock et al. 1983; Helfman et al. 2009). Fish are thought to use the same sensory organs used for near field water motion and sound pressure (e.g., lateral line system) for electroreception. In general, fish possess two types of electroreceptor organs: (1) ampullary receptors within the skin, which are connected to the surface by a canal filled with a conductive gel that is sensitive to electric fields of low-frequency (less than 0.1 to 25 Hz) and (2) tuberous receptors, embedded in the epidermis, and are covered with loosely packed epithelial cells; these receptors detect higher frequency electric fields (50 Hz to greater than two kilohertz) (Helfman et al. 2009). The distribution of electroreceptors on the head, and especially around the mouth, suggests that these sensory organs may be used in foraging and perhaps social communication (Collin and Whitehead 2004). Electroreceptors are also thought to aid in navigation, orientation, and migration of sharks and rays (Kalmijn 2000).

Although some individual fish species may exhibit a response to electromagnetic exposure, the fields generated are typically well below physiological and behavioral responses of magnetoreceptive fish (Navy 2019d). The strength of the electromagnetic devices used by the Navy is relatively minute and quickly dissipates at short distances away from the source. The devices work by emitting an electromagnetic field and mechanically generated underwater sound to simulate the presence of a ship. The magnetic field away from the device is comparable to the Earth's magnetic field (see sea turtle section above). Based on the small area around each electromagnetic device that will have an altered magnetic field, we assume that any potential disruption in an individual fish's orientation ability in the action area would only occur very close to the source. Additionally, this disruption would be temporary and last only as long as the fish remains within the area where the magnetic field is altered, which is likely to be very brief. Furthermore, most fish would be expected to avoid the device prior to entering the area where the magnetic field would be altered. We considers it extremely unlikely that ESA-listed fish would be exposed to electromagnetic energy at sufficient intensities to create an adverse effect through behavioral disruption or otherwise. Therefore, the effects of electromagnetic devices on all ESA-listed fish species considered in this opinion are insignificant and thus not likely to cause adverse effects.

8.1.3.2.2 Effects of Lasers on Fish

High-energy laser weapons would be used for testing activities in the action area. Fish could be exposed to a laser only if the beam missed the target. The following safeguards are in place to reduce the probability of the a high-energy laser striking the water: 1) the high-energy laser platform has provisions that prevent misfiring (i.e., firing when not intended) that all but

eliminate the possibility of that event, 2) the high-energy laser platforms have built-in constraints that only permit firing when it is locked onto a target, and 3) the operators are trained to stop firing when the laser aim point moves off of the selected target. Should the laser strike the sea surface, individual fish at or near the surface could be exposed. The potential for exposure to a high-energy laser beam decreases as the water depth increases. Most fish are unlikely to be exposed to laser activities because these species primarily occur more than a few meters below the sea surface. It is extremely unlikely that an ESA-listed fish would surface at the exact moment in the exact place that the laser misses its target and hits the surface. In summary, ESA-listed fish are extremely unlikely to be exposed to high-energy laser weapons based on 1) the relatively low number of events per year, 2) the very localized potential impact area of the laser beam, 3) the temporary duration of potential impact (seconds), 4) the low probability of fish at or near the surface at the exact time and place a laser misses its target, 5) the low probability of a laser missing its target; and 6) the relatively low density of ESA-listed fish species in the marine areas where activities using lasers are conducted. Therefore, the effects from high-energy laser weapons on the ESA-listed fish species considered in this opinion are discountable.

8.1.3.3 Physical Disturbance and Strike Stressors – Fish

Below we summarize the likely impacts of the various types of physical disturbance, including the potential for strike, during training and testing activities within the action area from 1) vessels and in-water devices, 2) military expended materials, including non-explosive practice munitions and fragments from high-explosive munitions, and 3) seafloor devices. For a general discussion of physical disturbance and strike stressors associated with the proposed action see Section 5.3.

8.1.3.3.1 Effects of Vessels and In-water Devices on Fish

Vessel traffic and in-water device use during Navy training and testing activities would be concentrated in certain portions of the action area such as areas near ports (e.g., Everett and Bremerton) or naval installations and ranges, but could occur throughout the action area. Sturgeon, in general, are more susceptible to vessel strike than other species due to their large body size. Although we are not aware of any reported incidences of green sturgeon vessel strike in the action area (by Navy vessels or other vessels), vessel strike has been documented and identified as a risk factor for other ESA-listed sturgeon species (ASSRT 2007; SSSRT 2010). However, green sturgeon densities in areas where Navy ship traffic is concentrated is anticipated to be low. In addition, Navy vessel traffic makes up approximately one-tenth of one percent of overall vessel traffic in the inland portion of the action area.

Other ESA-listed fish species (including salmonids, eulachon, and juvenile rockfish) spend at least some time in the upper portions of the water column and are, therefore, may also be susceptible to vessel strike, although reported ship strikes for these species are extremely rare. Despite these species' utilization of the upper portion of the water column for at least some of their life history, in most cases, we would anticipate the ESA-listed fish considered in this opinion would be able to detect vessels or other in-water devices and avoid them. Fish are able to

use a combination of sensory cues to detect approaching vessels, such as sight, hearing, and their lateral line (for nearby changes in water motion). A study on fish behavioral responses to vessels showed that most adults exhibit avoidance responses to engine noise, sonar, depth finders, and fish finders (Jørgensen et al. 2004), reducing the potential for vessel strikes. Misund (1997) found that fish ahead of a ship showed avoidance reactions at ranges of 50–350 meters. When the vessel passed over them, some fish responded with sudden escape responses that involved movement away from the vessel laterally or through downward compression of the school.

Regardless of the response, there is the potential for some type of stress or energetic cost as an individual fish must stop its current activity and divert its physiological and cognitive attention to responding to the vessel (Helfman et al. 2009). It is possible that fish may experience some level of physical disturbance, but it is not expected to result in more than a momentary behavioral response. Any avoidance behavior would be of short duration and intensity such that it would be insignificant to the animal.

Given the anticipated low density of the ESA-listed fish species in the action area, particularly around Navy ports or Naval installations, the ability of these species to maneuver to avoid any oncoming vessels, the low number of vessels associated with NWTT activities relative to non-military traffic in the area, and the lack of documented cases of Navy vessels or in-water devices striking these species (or any other fish species) in the action area, it is extremely unlikely that a Navy vessel associated with NWTT activities will strike an ESA-listed fish species. Any behavioral or stress response from fish avoiding an oncoming vessel or in-water device would be short-term, temporary and have no lasting impact of individual fitness. Therefore, potential effects on the ESA-listed fish species considered in this opinions from vessels and in-water devices are discountable (in the case of strikes) or insignificant (in the case of behavioral or stress response) and thus not likely to cause adverse effects.

8.1.3.3.2 Effects of Military Expended Materials on Fish

This section summarizes the strike potential to ESA-listed fish species from military expended materials including the following: 1) all sizes of non-explosive practice munitions, 2) fragments from high-explosive munitions, 3) expendable targets and target fragments; and 4) expended materials other than munitions, such as sonobuoys, expended bathythermographs, and torpedo accessories. While no strike of ESA-listed fish species from military expended materials has ever been reported or recorded, the possibility of a strike still exists. Given the large geographic area involved and the relatively low densities of ESA-listed fish species in the action area, we do not believe such interactions are likely. Additionally, while disturbance or strike from any expended material as it falls through the water column is possible, it is not likely because the objects will slow in velocity as they sink toward the bottom (e.g., guidance wires sink at an estimated rate of 0.2 meters per second; heavier items such as non-explosive munitions would likely sink faster, but would still be slowed as they sink to the bottom), and can be avoided by highly mobile organisms such the fish considered in this opinion.

In summary, it is extremely unlikely that an ESA-listed fish will be struck by military expended materials and the effects are therefore discountable. Any individuals encountering military expended materials as they fall through the water column are likely to move to avoid them. Given the effort expended by individuals to avoid them will be minimal (i.e., a few meters distance) and temporary, behavioral avoidance of military expended materials sinking through the water column is insignificant and thus not likely to cause adverse effects.

8.1.3.3.3 Effects of Seafloor Devices on Fish

The types of activities that use seafloor devices include items placed on the seafloor, dropped on the seafloor, or that move along the seafloor such as mine shapes, anchor blocks, anchors, bottom-placed instruments, and bottom-crawling unmanned underwater vehicles (Navy 2018b). The likelihood of any ESA-listed species encountering seafloor devices is very low given the densities of these species near the seafloor and in areas where such devices would be found. If encountered, ESA-listed fish would be expected to ignore or avoid any slowly moving or stationary device on the seafloor. In summary, we find that the probability of exposure to adverse effects from seafloor devices is extremely unlikely and considered discountable for the ESA-listed fish species considered in this opinion.

8.1.3.4 Entanglement Stressors – Fish

Some fish species are more susceptible to entanglement in derelict fishing gear and other marine debris, compared to other fish groups. For example, the shape and large body size of sturgeon may increase their risk of entanglement compared to smaller, more streamlined fish. For many pelagic species, including salmonids, eulachon, and rockfish the risk of entanglement is unlikely given their body shape and ability to avoid materials that could entangle them in the water column.

Although it is possible that some species of fish could become entangled in the guidance wires and fiber optic cables, the risk for most fish species is considered low. A portion of the fiber optic cable may be recovered, but some used for remotely operated mine neutralization activities would not. The length of this expended tactical fiber would vary (see Section 5.4.1) depending on the activity. Tactical fiber has an 8 µm (0.008 mm) silica core and acylate coating and looks and feels like thin monofilament fishing line; tactical fiber is relatively brittle and breaks if knotted, kinked, or abraded against a sharp object (Navy 2018b). Therefore, if this becomes looped around an underwater object or animal, it is unlikely to tighten. Although this material will not be recovered, it is expected to only remain in the water column for a short duration, and ultimately sink. Similarly, once a guidance wire is released it is expected to rapidly sink, settle and remain on the seafloor. If a wire were to snag or be partially resuspended, in theory a fish could swim through loops in the wire that may entangle the fish. Due to their rigidity and size, loops are less likely to form in a guidance wire or sonobuoy wire (Environmental Sciences Group 2005). Torpedo guidance wire is resistant to looping and coiling, suggesting it has a low entanglement potential compared to other entanglement hazards (Swope and McDonald 2013).

Similarly, fiber optic wire material is more resistant to forming loops and would easily break when tightly kinked or bent at a sharp angle. This is in contrast to fishing gear materials which are more common entanglement threats for fish and have breaking strengths much greater than that of guidance wire and fiber optic cables used during Navy activities.

Similarly, sonobuoy surface antennae, float units, and subsurface hydrophones are attached through a thin gauge, dual-conductor, and hard-draw copper strand wire, which is wrapped by a hollow rubber tubing or bungee. The tensile breaking strength of the wire and rubber tubing is no more than 40 pounds. The length of the cable is housed in a plastic canister dispenser, which remains attached upon deployment. The length of wire that extends out is no more than 1,500 feet and is dependent on the water depth and type of sonobuoy. Attached to the wire is a kite-drogue and damper disk stabilizing system made of non-woven nylon fabric. This nylon fabric is very thin and can be broken by hand; therefore, it does not pose a risk of entanglement for fish. Sonobuoys may remain suspended in the water column for no more than 30 hours, after which they sink to the seafloor. Sonobuoy wires may be expended within any of the range complexes throughout the action area. The wire that runs through the stabilizing system and leads to the hydrophone components of the sonobuoy hangs vertically in the water column, reducing the risk of ESA-listed fish becoming entangled.

Parachutes and decelerators could be encountered by ESA-listed fish at the sea surface, in the water column, or on the seafloor. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with these materials have the potential to result in mortality, adverse sub-lethal effects, and behavioral responses if a fish encounters them. Throughout the action area, the vast majority of expended decelerator and parachutes are small (18 inches) cruciform shaped decelerators used with sonobuoys. They have short attachment lines and, upon water impact, may remain at the surface for 5 to 15 seconds before the decelerator/parachute and its housing sink to the seafloor. Entanglement of a fish in a parachute assembly at the surface or within the water column would be unlikely, since the parachute would either have to land directly on the fish or the fish would have to swim into it before it sinks.

Large decelerators and parachutes may pose a higher degree of risk for ESA-listed fish because these parachutes have long lines (large chutes have 28 cords, approximately 40 to 70 feet long; extra-large chutes have 64 cords, up to 82 feet long), associated with them. Additionally, large parachutes are not weighted with anything to help them sink rapidly, and could remain suspended in the water column for an extended period of time. However, the chance of an encounter is remote given the relatively small number (i.e., 98 annually) of the large parachutes proposed to be deployed and the anticipated low abundance of this species in the offshore portion of the action area where large parachutes would be used. Given the vast area over which any one of these large parachutes would be deployed and the limited number of them deployed annually, the chances of an ESA-listed fish encountering them and becoming entangled is extremely low. During activities that involve recoverable targets (e.g., aerial drones), the Navy

recovers the target and any associated decelerators/parachutes to the maximum extent practicable consistent with personnel and equipment safety. This standard operating procedure could further reduce the potential entanglement of ESA-listed fish in decelerators/parachutes.

The Navy has proposed to test the use of biodegradable polymers during four total events per year at the Dabob and Keyport range sites. The following ESA-listed fish species that occur in the inland portion of the action area could be exposed to the effects of biodegradable polymers: Puget Sound/Georgia Basin DPS bocaccio; Puget Sound/Georgia Basin DPS yelloweye rockfish; Southern DPS eulachon; Southern DPS green sturgeon; Puget Sound ESU Chinook salmon; Hood Canal Summer-Run ESU chum salmon; and Puget Sound DPS steelhead. Unlike other entanglement stressors, biodegradable polymers only retain their strength for a relatively short period of time; the longer the biodegradable polymer remains in the water, the weaker it becomes, making it more brittle and likely to break (Navy 2020e). Therefore, the potential for entanglement by a fish would be limited. A fish would have to encounter the biodegradable polymer immediately after it was expended for it to be a potential entanglement risk (Navy 2020e). Given the anticipated low density of ESA-listed fish within the Dabob and Keyport range sites, particularly during active anti-submarine warfare countermeasure testing, and the relatively small number of events proposed, it is extremely unlikely that an ESA-listed fish would encounter the biodegradable polymer immediately after it was expended and become entangled. A more plausible scenario is that an ESA-listed fish would encounter the biodegradable polymer some time after it was expended when it has weakened and become more brittle. If a fish were to encounter the polymer a few hours after it was expended, it is expected to break easily and would not be an entanglement stressor (Navy 2020e).

In summary, for the reasons discussed above, the likelihood of ESA-listed fish species becoming entangled with expended materials such as parachutes, decelerators, cables, or wires is extremely unlikely. Therefore, we consider the effects from entanglement stressors on the ESA-listed fish species considered in this opinion to be discountable.

8.1.3.5 Ingestion Stressors – Fish

ESA-listed fish occurring in the action area have the potential to ingest military expended materials resulting from NWTT activities. The Navy expends the following types of materials during training and testing in the action area that could become ingestion stressors: non-explosive practice munitions (small- and medium-caliber), fragments from high-explosives, and fragments from targets, chaff, flare casings (including plastic end caps and pistons).

Most of the items that could be ingested by ESA-listed fish would be expended in the offshore portion of the action area. Pelagic species (i.e., salmonids, eulachon, and rockfish) are more likely to ingest expended materials floating in the water column. Military expended materials that could impact pelagic species that feed at or just below the surface or in the water column include those items that float or are suspended in the water column for some period of time (e.g., end caps and pistons from chaff cartridges or flares). If an ESA-listed fish accidentally ingested

such an item at or near the surface it would likely expel it after determining it was not a prey item. Expended materials made of metal would sink quickly through the water column before settling on the seafloor. Once the item sinks to the seafloor, it would be unavailable for ingestion by pelagic species. Benthic associated species such as sturgeon could feed on military expended items that have settled on the seafloor. However, this is unlikely to occur considering the depths at which most expended items would be found and the relatively low density of ESA-listed green sturgeon in areas where ingestible items would be expended. Shiny fragments of sinking munitions in the water column could attract and be ingested by fast, mobile predators that chase moving prey. However, this is an unlikely scenario considering: 1) the small amount of time such objects would be in the water column and, 2) that highly mobile predators would be expected to evacuate an area where an explosion has just occurred. In addition, ESA-listed species are relatively rare and dispersed throughout the offshore portion of the action area, which further decreases the likelihood that one would encounter sinking expended materials in the water column.

For the reasons provided above, we consider it extremely unlikely that ESA-listed fish species would ingest materials resulting in adverse effects to the fish's normal behavior, growth, survival, or reproductive success. Therefore, we consider the effects from ingestion stressors on the ESA-listed fish species considered in this opinion to be discountable.

8.1.3.6 Stressors Resulting in Effects to Fish Habitat or Prey

Stressors from training and testing activities that could result in secondary or indirect effects on ESA-listed fish via impacts to habitat, prey, sediment and water quality include explosives and byproducts, metals, and chemicals.

Underwater explosions could impact other species in the food web, including prey species that the ESA-listed fish species considered in the opinion feed upon. The impacts of explosions would differ depending on the type of prey species in the area of the blast. Underwater explosions may reduce available prey items for ESA-listed fish species by either directly killing prey or by scaring them from the area. In addition to physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that could include swimming to the surface or scattering away from the source. The abundance of fish prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Any of these scenarios would likely be short-term and temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web expected. Due to the infrequent use of underwater explosives and the limited locations at which underwater explosives are used, it is not expected their use will have a persistent effect on prey availability or the health of the aquatic food web. As highly mobile, open water predators, salmonids would not likely be adversely affected by such short-term, localized impacts to their

prey base in the open ocean. Thus, the effects of explosives on ESA-listed salmonids via impacts on their prey are considered insignificant.

Due to the limited number and spatial distribution of explosives proposed for use in inland waters, it is unlikely that yelloweye rockfish or bocaccio prey would be impacted. Similarly, green sturgeon occur primarily in nearshore and inland portions of the action area where explosive use would be very limited. In addition, green sturgeon feed primarily on benthic invertebrates (Dumbauld et al. 2008; Houston 1988; Moyle et al. 1992b; Wilson and McKinley 2004), which are less likely to be affected by Navy explosives as compared to pelagic fish prey. Eulachon feed on plankton (e.g., crustaceans such as copepods and euphausiids) which are also less likely to be affected by Navy explosives as compared to pelagic fish prey. Thus, the effects of explosives on ESA-listed rockfish, green sturgeon, and eulachon via impacts on their prey are considered discountable.

In terms of explosive byproducts, high-order explosions consume most of the explosive material, creating typical combustion byproducts. Explosion by-products associated with high order detonations present no indirect stressors to marine ESA-listed species because most byproducts are common in seawater and the rest are quickly diluted below appreciable levels. Explosive byproducts are not expected to result in detectable changes in sediment or water quality. Low-order explosives leave more explosive material in the water but this material is not water soluble, degrades quickly, and is quickly dispersed. The levels of explosive materials and byproducts are not detectable above background levels one to two meters from a degrading source. As such, the effects of explosive byproducts on ESA-listed fish species considered in this opinion via impacts on water quality are extremely unlikely and considered discountable.

Metals can be introduced into seawater and sediments as a result of Navy training and testing activities involving ship hulks, targets, munitions, and other military expended materials (Environmental Sciences Group, 2005). Fish could be exposed to released metals through contact with contaminants in the sediment or water, and ingestion of contaminated sediments. Certain metals are harmful to fish at concentrations above background levels (e.g., cadmium, chromium, lead, mercury, zinc, copper, manganese, and many others) (Wang and Rainbow 2008). Most metals used in Navy expendables is benign and all corroding metals would either be diluted into the ocean currents or be sequestered in the sediments immediately surrounding the source (Navy 2013b). Concentrations of metals in seawater are considerably lower than concentrations in sediments. As such, it is extremely unlikely that fish would be indirectly impacted by toxic metals via the water given the vast open ocean area over which metals would be released. Metals deposited on the sea floor will be buried in sediment and slowly degrade over time. ESA-listed fish species that feed primarily in the water column (i.e., salmonids, rockfish, and eulachon) would not likely come into contact with metals in marine sediments.

Benthic associated fish such as green sturgeon may be more susceptible to contact with metal contaminants in the sediment and through ingestion of contaminated sediments or prey. Research

has demonstrated that some smaller marine organisms are attracted to metal munitions as a hard substrate for colonization or as shelter (Kelley et al. 2016; Smith and Marx Jr. 2016). However, as mentioned above, green sturgeon occur primarily in nearshore and inland portions of the action area where very few Navy activities that result in the introduction of metals would occur.

Some metals bioaccumulate and physiological impacts begin to occur only after several trophic transfers concentrate the toxic metals. Evidence from a number of studies (Briggs et al. 2016; Edwards and coauthors. 2016; Kelley et al. 2016; Koide et al. 2016; Navy 2013c) indicate metal contamination is highly localized and that bioaccumulation resulting from munitions cannot be demonstrated. Specifically, in sampled marine life living on or around munitions on the seafloor, metal concentrations could not be definitively linked to the munitions since comparison of metals in sediment next to munitions show relatively little difference in comparison to other "clean" marine sediments used as a control/reference (Koide et al. 2016). The research cited above indicates that metals introduced into the action area are unlikely to have adverse effects on ESA-listed fish prey or habitat. Thus, the effects of metals introduced into seawater and sediments as a result of NWTT activities on ESA-listed fish species, either directly or through impacts on their prey or habitat, are extremely unlikely and considered discountable.

Several NWTT activities introduce chemicals into the marine environment that are potentially harmful to fish in higher concentrations. However, rapid dilution would be expected and toxic concentrations are unlikely to be encountered by ESA-listed fish or their prey. Chemicals introduced are principally from flares and propellants for missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment. Flares and missiles that operationally fail may release perchlorate, which is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals if in sufficient concentration. However, such concentrations would be localized and are not likely to persist in the ocean environment. Research has demonstrated that perchlorate does not bioconcentrate or bioaccumulate, which is consistent with the expectations for a water-soluble compound (Furin et al. 2013). Given the dynamic nature of the environment (currents, tides, etc.), long-term impacts from perchlorate in the environment near the expended item are not expected. It is extremely unlikely that perchlorate from failed expendable items would compromise water quality to the point that it would result in adverse effects on ESA-listed fish prey or habitat.

Chemical and biological simulants used during simulant testing could affect water quality and lower trophic levels within the food web of ESA-listed fishes. All chemical simulants proposed for use by the Navy would have low toxicity to humans and the environment and all biological simulants are considered to be Biosafety Level 1 organisms (Navy 2020e). While some minor effects of chemical and biological simulants on water quality and prey are possible, such effects, if they occur at all, would be extremely short-term and localized. Simulants released in the offshore portion of the action area would be expected to disperse rapidly into the ocean

environment. Therefore, even if simulant testing resulted in adverse effects to prey and water quality, it is extremely unlikely that such secondary effects would result in adverse effects on ESA-listed fishes. In summary, the effects of chemicals and simulants (chemical and biological) used during Navy training and testing on ESA-listed fishes via water quality and prey are considered discountable.

8.2 Stressors Likely to Adversely Affect ESA-listed Species

We determined that the following stressors from the proposed action are likely to adversely affect ESA-listed species:

1) Acoustic stressors from sonar and other transducers – cetaceans and pinnipeds;
2) Stressors from in-water explosives – cetaceans, sea turtles, and fish;
3) Physical disturbance and strike stressors from vessels – cetaceans.

The following sections describe the effects of these stressors on ESA-listed species. For each type of stressor, we 1) describe the potential adverse effects of the stressor, 2) summarize the exposure analysis which (where possible) estimates the number and life stages of individuals of each ESA-listed species that may be exposed to the stressor; and 3) provide our assessment of the likely responses these species would exhibit to this exposure.

While NMFS recognizes that Navy training and testing requirements change over time in response to global or geopolitical events and other factors, the general types of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assume that the training and testing activities proposed by the Navy during the seven-year period of NMFS' proposed incidental take authorization pursuant to the MMPA would continue into the reasonably foreseeable future at levels similar to those assessed during this consultation.

8.2.1 Marine Mammals

In the sections below we analyze the effects of stressors resulting from the proposed action that are likely to adversely affect ESA-listed marine mammals, including sonar and other transducers, explosives, and vessels.

8.2.1.1 Sonar and Other Transducers

As described in Section 5.1.1, sonar and other transducers includes a variety of acoustic devices used to obtain and transmit information about the undersea environment. Some examples are: mid-frequency hull-mounted sonars used to find and track submarines; high-frequency small object detection sonars used to detect mines; high-frequency underwater modems used to transfer data over short ranges; and extremely high-frequency (greater than 200 kilohertz) Doppler sonars used for navigation, like those used on commercial and private vessels.

Assessing whether a sound may disturb or injure a marine mammal involves understanding the characteristics of the acoustic sources, the marine mammals that may be present in the vicinity of the sources, and the effects that sound may have on the physiology and behavior of those animals. Although it is known that sound is important for marine mammal communication, navigation, and foraging, there are many unknowns in assessing impacts such as the potential interaction of different effects and the significance of responses by marine mammals to sound exposures (Nowacek et al. 2007; Southall et al. 2007a). Furthermore, many other factors besides the received level of sound may affect an animal's reaction such as the duration of the sound-producing activity, the animal's physical condition, prior experience with the sound, activity at the time of exposure (e.g., feeding, traveling, resting), the context of the exposure (e.g., in a semi-enclosed bay vs open ocean), and proximity to the sound source.

The potential effects of acoustic exposure range from physical injury or trauma, to an observable behavioral response, to a stress response that may not be detectable. Injury can occur to organs or tissues of an animal due to exposure to pressure waves. Non-auditory injury (i.e., other than PTS) and mortality of ESA-listed marine mammals from sonar and other transducers is considered so unlikely as to be discountable under normal conditions, and is, therefore, not considered further in this opinion for marine mammals[19].

Noise-induced hearing loss is a decrease in hearing sensitivity, which can either be temporary or permanent. Stress can help an animal cope with changing conditions, but too much stress can result in negative physiological effects. Masking can occur when the perception or communication of a biologically-important sound is interfered with by a second sound (e.g., noise from Navy training and testing). Behavioral responses range from brief distractions to avoidance of a sound source to prolonged flight. The sections below provide additional information on the potential effects of sonar and other transducers on marine mammals. We use this information to discuss the likely effects of Navy sonar use on ESA-listed cetaceans in our exposure, response, and risk analyses that follow.

8.2.1.1.1 Hearing Loss and Auditory Injury

Exposure to intense sound may result in noise-induced hearing loss that persists after cessation of the noise exposure. Hearing loss may be temporary or permanent, depending on factors such as the exposure frequency, received SPL, temporal pattern, and duration. The frequencies affected by hearing loss will vary depending on the frequency of the noise, with frequencies at and above the noise frequency most strongly affected (i.e., higher amount of threshold shift). The amount of hearing loss may range from slight to profound. Hearing loss has only been studied in a few species of marine mammals, although hearing studies with terrestrial mammals are also informative. Hearing threshold shifts in mid-frequency cetaceans exposed to non-impulsive

[19] Non-auditory injury from sonar is not anticipated due to the lack of fast rise times, lack of high peak pressures, and the lack of high acoustic impulse of sonar. Note that non-auditory injury is possible from impulsive sources such as explosions.

sound (e.g., active sonar tones) has been investigated in multiple studies (e.g., Finneran et al. 2010; Finneran et al. 2005; Finneran and Schlundt 2013; Mooney et al. 2009a; Mooney et al. 2009b).

Hearing loss is typically quantified in terms of threshold shift — the amount (in dB) that hearing thresholds at one or more specified frequencies are elevated, compared to their pre-exposure values, at some specific time after the noise exposure. The amount of threshold shift measured usually decreases with increasing recovery time — the amount of time that has elapsed since a noise exposure. If the threshold shift eventually returns to zero (i.e., the hearing threshold returns to the pre-exposure value), the threshold shift is considered temporary or TTS. If the threshold shift does not completely recover (the threshold remains elevated compared to the pre-exposure value), the remaining threshold shift is determined to be permanent or PTS. Figure 89 shows two hypothetical threshold shifts: one that completely recovers, a TTS, and one that does not completely recover, leaving some PTS.

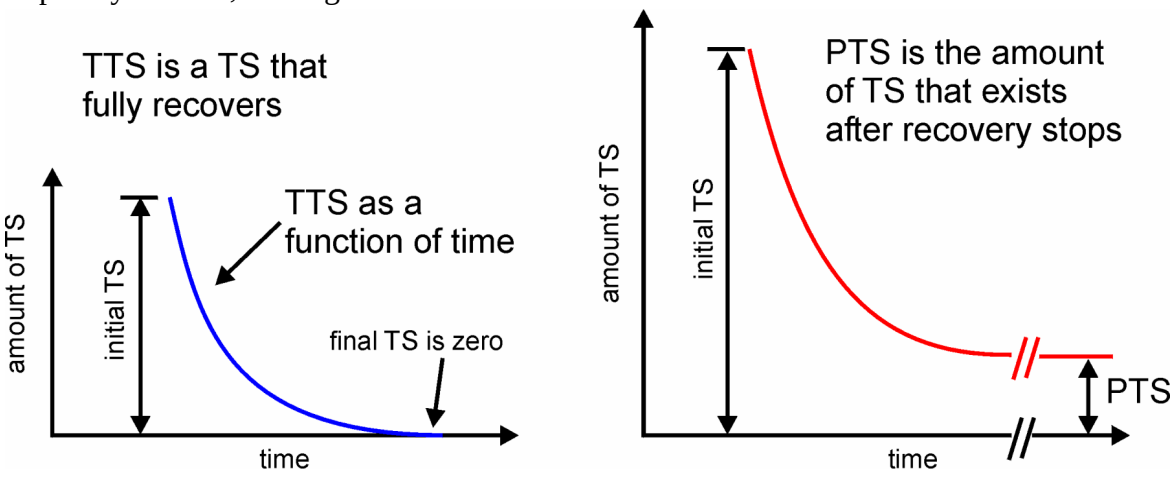


Figure 89. Two hypothetical temporary threshold shifts.

Studies have revealed that intense noise exposures may also cause auditory system injury that does not result in PTS (i.e., hearing thresholds return to normal after the exposure, but there is injury nonetheless). Kujawa and Liberman (2009) found that noise exposures sufficient to produce a TTS of 40 dB, measured 24 hours post-exposure, resulted in acute loss of nerve terminals and delayed degeneration of the cochlear nerve in mice. Lin et al. (2011) found a similar result in guinea pigs with a TTS in auditory-evoked potential up to approximately 50 dB, measured 24 hours post-exposure resulting in neural degeneration. These studies demonstrate that PTS should not be used as the sole indicator of auditory injury because exposures producing high levels of TTS (40 to 50 dB measured 24 hours after exposure) — but no PTS — may result in auditory injury or impairment.

There are no simple functional relationships between TTS and the occurrence of PTS or other auditory injury (e.g., neural degeneration). Further, TTS and PTS are mutually exclusive because

an exposure that produces TTS cannot also produce PTS within the same frequency band in the same individual (Reichmuth et al. 2019). If an initial threshold shift results in only partial recovery, resulting in some amount of PTS, the difference between the initial threshold shift and the PTS is not called TTS. As TTS increases, the likelihood that additional exposure or duration to sound will result in PTS and/or other injury also increases. An exception to this is that researchers might not be able to observe gradual growth of TTS with increased levels of sound exposure before onset of PTS (Reichmuth et al. 2019). Similarly, PTS can occur without measurable behavioral modifications (Reichmuth et al. 2019). Exposure thresholds for the occurrence of PTS or other auditory injury can therefore be defined based on a specific amount of TTS. The specific upper limit of TTS is based on experimental data showing amounts of TTS that have not resulted in PTS or injury. In other words, we do not need to know the exact functional relationship between TTS and PTS or other injury. We only need to know the upper limit for TTS before some PTS or injury is possible.

A variety of human and terrestrial mammal data indicate that threshold shifts up to 40 dB may be induced without PTS, and that 40 dB is a reasonable upper limit for allowable threshold shift to prevent PTS (e.g., Kryter et al. 1965; Ward 1960). It is reasonable to assume the same relationship would hold for marine mammals because there are many similarities between the inner ears of marine and terrestrial mammals. Experiments with marine mammals have revealed similarities to terrestrial mammals for features such as TTS, age-related hearing loss, drug-induced hearing loss, masking, and frequency selectivity (Finneran et al. 2005; Finneran et al. 2015; Ketten 2000). Therefore, we assume that sound exposures sufficient to produce 40 dB of TTS measured approximately four minutes after exposure represent the limit of a non-injurious exposure; i.e., higher level exposures have the potential to cause auditory injury. Exposures sufficient to produce a TTS of 40 dB, measured approximately four minutes after exposure therefore represent the threshold for auditory injury. The predicted injury could consist of either hair cell damage/loss resulting in PTS, or other auditory injury such as the delayed neural degeneration identified by Kujawa and Liberman (2009) and Lin et al. (2011) that may not result in PTS.

Numerous studies have directly examined noise-induced hearing loss in marine mammals (See Finneran et al. 2015). In these studies, hearing thresholds were measured in marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds was then used to determine the amount of TTS at various post-exposure times. The major findings from these studies include the following:

- The method used to test hearing may affect the resulting amount of measured TTS, with neurophysiological measures producing larger amounts of TTS compared to psychophysical measures (Finneran et al. 2007; Finneran et al. 2015).
- The amount of TTS varies with the hearing test frequency. The higher the SPL, the higher the TTS induced at frequencies higher than the exposure frequency; below 148 dB re 1 μPa, the maximum TTS was at 6.5 kilohertz, whereas above 148 dB re 1 μPa, the

maximum TTS was at 9.2 kilohertz. (Kastelein et al. 2014b). For high level exposures to tonal or octave band sounds, the maximum TTS typically occurs one-half to one octave above the exposure frequency (Finneran et al. 2007; Mooney et al. 2009a; Nachtigall et al. 2004; Popov et al. 2013; Popov et al. 2011; Reichmuth et al. 2019; Schlundt et al. 2000). The overall spread of TTS from tonal exposures can therefore extend over a large frequency range; i.e., narrowband exposures can produce broadband (greater than one octave) TTS.

- The amount of TTS usually increases with exposure SPL and duration, and is correlated with SEL, especially if the range of exposure durations is relatively small (Kastak et al. 2007; Kastelein et al. 2014b; Popov et al. 2014). As the exposure duration increases, the relationship between TTS and SEL begins to break down. Specifically, duration has a more significant effect on TTS than would be predicted on the basis of SEL alone (Finneran and Schlundt 2010; Kastak et al. 2005; Mooney et al. 2009a). This means if two exposures have the same SEL but different durations, the exposure with the longer duration (thus lower SPL) will tend to produce more TTS than the exposure with the higher SPL and shorter duration. In most acoustic impact assessments, the scenarios of interest involve shorter duration exposures than the cetacean experimental data from which impact thresholds are derived; therefore, use of SEL tends to over-estimate the amount of TTS. Despite this, SEL continues to be used in many situations because it is relatively simple, more accurate than SPL alone, and lends itself easily to scenarios involving multiple exposures with different SPL.
- The amount of TTS depends on the exposure frequency. Sounds that are well below the frequency level of best sensitivity are less hazardous than those at or near the level of best sensitivity (Finneran and Schlundt 2013). The onset of TTS — defined as a threshold shift of six dB measured approximately four minutes after exposure (i.e., clearly above the typical variation in threshold measurements) — also varies with exposure frequency. At low frequencies TTS onset exposure levels are higher compared to those in the region of best sensitivity. However, gradual increases of TTS may not be directly observable with increasing exposure levels before the onset of PTS, which can occur without measurable behavioral modifications (Reichmuth et al. 2019).
- TTS can accumulate across multiple exposures, but the resulting TTS will be less than the TTS from a single, continuous exposure with the same SEL (Finneran et al. 2010; Kastelein et al. 2015c; Kastelein et al. 2014b; Mooney et al. 2009b). This means that TTS predictions based on the total, cumulative SEL will likely overestimate the amount of TTS from intermittent exposures such as sonars and impulsive sources.
- The amount of observed TTS tends to decrease with increasing time following the exposure; however, the relationship is not monotonic (i.e., increasing exposure does not always increase TTS). The time required for complete recovery of hearing depends on the magnitude of the initial shift; for relatively small shifts recovery may be complete in a few minutes, while large shifts (e.g., approximately 40 dB) may require several days or

longer for recovery. Recovery times are consistent for similar-magnitude shifts, regardless of the type of fatiguing sound exposure (impulsive, continuous noise band, or sinusoidal) (Kastelein et al. 2019b). Under many circumstances TTS recovers linearly with the logarithm of time (Dear et al. 2010; Finneran et al. 2010; Finneran and Schlundt 2013; Kastelein et al. 2013a; Kastelein et al. 2012a; Kastelein et al. 2012b; Kastelein et al. 2014b; Kastelein et al. 2014c; Popov et al. 2014; Popov et al. 2013; Popov et al. 2011). This means that for each doubling of recovery time, the amount of TTS will decrease by the same amount (e.g., six dB recovery per doubling of time), although this may not hold for all sound sources and species.

Due to the higher exposure levels or longer exposure durations required to induce hearing loss, only a few types of man-made sound sources have the potential to cause a threshold shift to a marine mammal in the wild. These include some sonars and other transducers that would be used by the Navy as part of NWTT, and impulsive sound sources such as air guns and impact pile driving that would not be used by the Navy as part of NWTT. Recent studies have begun to show that some cetaceans may learn to reduce their hearing sensitivity (presumably to protect their hearing) when warned of an impending intense sound exposure (Finneran 2018; Nachtigall and Supin 2013; Nachtigall et al. 2016). The marine mammal criteria and thresholds for hearing impairment and non-auditory injury from sonars and other transducers used for the Navy's quantitative model were described in Section 2.2.2.1.

Southall et al. (2019c) evaluated Southall et al. (2007a) and used updated scientific information to propose revised noise exposure criteria to predict onset of auditory effects in marine mammals (i.e., PTS and TTS onset). Southall et al. (2019c) note that the quantitative processes described and the resulting exposure criteria (i.e., thresholds and auditory weighting functions) are largely identical to those in Finneran (2015b) and NMFS (2016l); NMFS (2018b). However, they differ in that the Southall et al. (2019c) exposure criteria are more broadly applicable as they include all marine mammal species (rather than those only under NMFS jurisdiction) for all noise exposures (both in air and underwater for amphibious species), and that while the hearing group compositions are identical they renamed the hearing groups. The thresholds discussed in the paper (TTS/PTS only) are the same as Navy's criteria and NMFS criteria.

8.2.1.1.2 Physiological Stress

The growing field of conservation physiology relies in part on the ability to monitor stress hormones in populations of animals, particularly those that are threatened or endangered. The ability to make predictions from stress hormones about impacts to individuals and populations exposed to various forms of stressors, natural and human-caused, relies on understanding the linkages between changes in stress hormones and resulting physiological impacts. At this time, the sound characteristics that correlate with specific stress responses in cetaceans are poorly understood, as are the ultimate consequences due to these changes. Efforts are underway to try to improve understanding of, and the ability to predict, how stressors ultimately affect marine

mammal populations (e.g., New et al. 2013a; New et al. 2013b; Pirotta et al. 2015). With respect to acoustically-induced stress, this includes not only determining how and to what degree various types of anthropogenic sounds cause stress in cetaceans, but what factors can mitigate those responses. Factors potentially affecting an animal's response to a stressor include the animal's life history stage, sex, age, reproductive status, overall physiological and behavioral plasticity, and whether they are naïve or experienced with the sound (e.g., prior experience with a stressor may result in a reduced response due to habituation; Finneran and Branstetter 2013; St Aubin and Dierauf 2001). Because there are many unknowns regarding the occurrence of acoustically-induced stress responses in marine mammals, it is a reasonable assumption that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

Marine mammals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to disease and naturally occurring toxins, lack of prey availability, and interactions with predators all contribute to the stress a marine mammal experiences (Atkinson et al. 2015). Breeding cycles, periods of fasting, social interactions with members of the same species, and molting (for pinnipeds) are also stressors, although they are natural components of an animal's life history. Anthropogenic activities have the potential to provide additional stressors beyond those that occur naturally (Fair et al. 2014; Meissner et al. 2015; Rolland et al. 2012). Anthropogenic stressors potentially include fishery interactions, pollution, and ocean noise.

The stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor (Moberg 2000). The generalized stress response is classically characterized by the release of cortisol, a hormone that has many functions including elevation of blood sugar, suppression of the immune system, and alteration of the biochemical pathways that affect fat, protein, and carbohydrate metabolism. It is now known that the endocrine response (glandular secretions of hormones into the blood) to a stressor can extend to other hormones. For instance, thyroid hormones can also vary under the influence of certain stressors, particularly food deprivation. These types of responses typically occur on the order of minutes to days. The "fight or flight" response, an acute stress response, is characterized by the very rapid release of hormones that stimulate glucose release, increase heart rate, and increase oxygen consumption.

Rolland et al. (2017) studied glucocorticoid hormones in North Atlantic right whales, evaluating and comparing healthy whales with whales that were chronically entangled in fishing gear. The authors found that stress hormones in the entangled whales were elevated compared to those of healthy whales. The authors also cited several studies to conclude that stress responses over a short period of time (i.e., hours/days) can be beneficial and life-saving. However, chronic elevations of glucocorticoids (i.e., weeks/months) may result in decreased growth, depressed immune system function, and suppression of reproduction (e.g., Romero and Wikelski 2001; Sapolsky et al. 2000). If the magnitude and duration of the stress response is too great, too long,

or occurs at a time when the animal is in a vulnerable state, it can have negative consequences to the organism (e.g., decreased immune function, decreased reproduction).

What is known about the function of the various stress hormones is based largely upon observations of the stress response in terrestrial mammals. The endocrine response of marine mammals to stress may not be the same as that of terrestrial mammals because of the selective pressures marine mammals faced during their evolution in an ocean environment (Atkinson et al. 2015). For example, due to the necessity of breath-holding while diving and foraging at depth, the physiological role of the epinephrine and norepinephrine (catecholamines) may be different in marine versus terrestrial mammals. Catecholamines increase during breath-hold diving in seals, co-occurring with a reduction in heart rate, peripheral vasoconstriction (constriction of blood vessels), and an increased reliance on anaerobic metabolism during extended dives (Hance et al. 1982; Hochachka et al. 1995; Hurford et al. 1996). The catecholamine increase is not associated with an increased heart rate, glycemic release, and increased oxygen consumption typical of terrestrial mammals. Other hormone functions may also be different, such as aldosterone, which has been speculated to not only contribute to electrolyte balance, but possibly also the maintenance of blood pressure during periods of vasoconstriction (Houser et al. 2011). In marine mammals, aldosterone is thought to play a particular role in stress mediation because of its noted role in mitigating stress response (St Aubin and Dierauf 2001; St. Aubin and Geraci 1989).

Relatively little information exists on the linkage between anthropogenic sound exposure and stress in marine mammals, and even less information exists on the ultimate consequences of sound-induced stress responses (either acute or chronic). Most studies to date have focused on acute responses to sound either by measuring catecholamines or by measuring heart rate as an assumed proxy for an acute stress response. Belugas demonstrated no catecholamine response to the playback of oil drilling sounds (Thomas et al. 1990) but showed a small but statistically significant increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al. 2004). A bottlenose dolphin exposed to the same seismic water gun signals did not demonstrate a catecholamine response, but did demonstrate a statistically significant elevation in aldosterone (Romano et al. 2004), albeit the increase was within the normal daily variation observed in this species (St. Aubin et al. 1996). Increases in heart rate were observed in bottlenose dolphins to which known calls of other dolphins were played, although no increase in heart rate was observed when background tank noise was played back (Miksis et al. 2001). Unfortunately, it cannot be determined from this study whether the increase in heart rate was due to stress or an anticipation of being reunited with the dolphin to which the vocalization belonged. Similarly, a young beluga's heart rate was observed to increase during exposure to noise, with increases dependent upon the frequency band of noise and duration of exposure, and with a sharp decrease to normal or below normal levels upon cessation of the exposure (Lyamin et al. 2011). However, this response may have been in part due to the conditions during testing. Kvadsheim et al. (2010a) measured the heart rate of captive hooded

seals during exposure to sonar signals, and found an increase in the heart rate of the seals during exposure periods versus control periods when the animals were at the surface. When the animals dove, the normal dive-related bradycardia (decrease in heart rate) was not impacted by the sonar exposure. Similarly, researchers observed a rapid but short-lived decrease in heart rates in harbor and gray seals exposed to seismic air guns (Gordon et al. 2003). Williams et al. (2017) found a non-linear increase in oxygen consumption with both stroke rate and heart rate in swimming and diving bottlenose dolphins, and found that the average energy expended per stroke increased from 2.81 Joules/kilogram/stroke during preferred swim speeds to a maximum expenditure of 6.41 Joules/kilogram/stroke when freely following a boat. Houser et al. (2020) measured cortisol and epinephrine levels in bottlenose dolphins and found no correlation between these stress hormone levels and received sound pressure levels from mid-frequency sonar signals. Houser et al. (2020) and Houser et al. (2013c) observed that the severity of bottlenose dolphin behavioral responses scaled with sound pressure level. Therefore, behavioral reactions to sonar signals may not be indicative of a hormonal stress response.

Similarly, a limited amount of work has addressed how chronic exposure to acoustic stressors affect stress hormones in cetaceans, particularly as it relates to survival or reproduction. Rolland et al. (2012) compared the levels of cortisol metabolites in North Atlantic right whale feces collected before and after September 11, 2001. Following the events of September 11, shipping was significantly reduced in the region where fecal collections were made, and regional ocean background noise declined. Fecal cortisol metabolites significantly decreased during the period of reduced ship traffic and ocean noise (Rolland et al. 2012). Considerably more work has been conducted in an attempt to determine the potential effect of boating on smaller cetaceans, particularly killer whales (Bain 2002; Erbe 2002; Noren et al. 2009). Most of these efforts focused primarily on estimates of metabolic costs associated with altered behavior or inferred consequences of boat presence and noise, but did not directly measure stress hormones. Ayres et al. (2012) investigated Southern Resident killer whale fecal thyroid hormone and cortisol metabolites to assess two potential threats to the species recovery: lack of prey (salmon) and impacts from exposure to the physical presence of vessel traffic (but without measuring vessel traffic noise). Ayres et al. (2012) concluded from these stress hormone measurements that the lack of prey overshadowed any population-level physiological impacts on Southern Resident killer whales due to vessel traffic. Collectively, these studies indicate the difficulty in teasing out factors that are dominant in exerting influence on the secretion of stress hormones, including the separate and additive effects of vessel presence and vessel noise.

8.2.1.1.3 Masking

Clark et al. (2009) developed a method for estimating masking effects on communication signals for low-frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, their technique calculates that a right whale's optimal communication space (around 20 km) is decreased by 84 percent when two commercial ships pass through it. Similarly, Aguilar Soto et al. (2006) found that a 15 dB increase in background noise due to

vessels led to a communication range of only 18 percent of its normal value for foraging beaked whales. Their method relies on empirical data on source levels of calls (which is unknown for many species) and requires many assumptions such as pre-industrial ambient noise conditions and simplifications of animal hearing and behavior, but it is an important step in determining the impact of anthropogenic noise on animal communication. Erbe (2016) developed a model with a noise source-centered view of masking to examine how a call may be masked from a receiver by a noise as a function of caller, receiver, and noise-source location, distance relative to each other, and received level of the call.

Vocal changes in response to anthropogenic noise can occur across the repertoire of sound production modes used by marine mammals, such as whistling, echolocation click production, calling, and singing. Vocalization changes may result from a need to compete with an increase in background noise and include increasing the source level, modifying the frequency, increasing the call repetition rate of vocalizations, or ceasing to vocalize in the presence of increased noise (Hotchkin and Parks 2013). In cetaceans, vocalization changes were reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying (e.g., Holt 2008; Holt et al. 2011; Rolland et al. 2012) as well as changes in the natural acoustic environment (Dunlop et al. 2014). Vocal changes can be temporary, or can be permanent, as seen in the increase in starting frequency for the North Atlantic right whale upcall over the last 50 years (Tennessen and Parks 2016). This shift in frequency was modeled, and it was found that it led to increased detection ranges between right whales. The frequency shift, coupled with an increase in call intensity by 20 dB, led to a call detectability range of less than 3 km to over 9 km (Tennessen and Parks 2016). In some cases, these vocal changes may have fitness consequences, such as an increase in metabolic rates and oxygen consumption, as was found for bottlenose dolphins when increasing their call amplitude (Holt et al. 2015). A switch from vocal communication to physical, surface-generated sounds such as pectoral fin slapping or breaching was observed for humpback whales in the presence of increasing natural background noise levels, indicating that adaptations to masking may also move beyond vocal modifications (Dunlop et al. 2010). These changes all represent possible tactics by the sound-producing animal to reduce the impact of masking. The receiving animal can also reduce masking by using active listening strategies such as orienting to the sound source, moving to a different location to improve binaural cues (time or intensity differences between the ears due to a sound source's location relative to the animal's head), or going still to reduce noise associated with hydrodynamic flow. The structure of some noises (e.g., amplitude modulation) may also provide some release from masking through comodulation masking release (the difference in masking when a noise is broadband versus having the same bandwidth as the signal; Branstetter and Finneran 2008). Signal characteristics (e.g., whether the signal has harmonics, or is frequency modulated) may further enhance the detectability of a signal in noise (Cunningham et al. 2014).

Evidence suggests that at least some marine mammals have the ability to acoustically identify potential predators (Allen et al. 2014; Cummings and Thompson 1971a; Cure et al. 2015), which

may be reduced in the presence of a masking noise, particularly if it occurs in the same frequency band. Therefore, the occurrence of masking may prevent marine mammals from responding to the acoustic cues produced by their predators. Whether this is a possibility depends on the duration of the masking and the likelihood of encountering a predator during the time that detection and identification of predator cues are impeded. For example, harbor seals that reside in the coastal waters off British Columbia are frequently targeted by certain groups of killer whales. The seals discriminate between the calls of threatening and non-threatening killer whales (Deecke et al. 2002), a capability that should increase survivorship while reducing the energy required to attend to all killer whale calls. Similarly, sperm whales (Isojunno et al. 2016), long-finned pilot whales (Visser et al. 2016), and humpback whales (Cure et al. 2015) changed their behavior in response to killer whale vocalization playbacks. These findings indicate that some recognition of predator cues could be missed if the killer whale vocalizations were masked.

Masking could occur as a result of sonar and other transducers. As stated previously, masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Because traditional military sonars typically have low duty cycles, the effects of such masking would likely be limited when compared with continuous sources (e.g., vessel noise). Low-frequency active sonar could overlap with mysticete vocalizations (e.g., minke and humpback whales). For example, in the presence of low-frequency active sonar, humpback whales were observed to increase the length of their songs (Fristrup et al. 2003; Miller et al. 2000), possibly due to the overlap in frequencies between the whale song and the low-frequency active sonar.

Newer high duty cycle or continuous active sonars have more potential to mask vocalizations, particularly for mid-frequency cetaceans. These sonars transmit more frequently (greater than 80 percent duty cycle) than traditional sonars, but at a substantially lower source level. Similarly, high frequency acoustic sources such as pingers that operate at higher repetition rates (e.g., two to ten kilohertz with harmonics up to 19 kilohertz, 76 to 77 pings per minute (Culik et al. 2001) also operate at lower source levels. While the lower source levels of these systems limits the range of impact compared to more traditional systems, animals close to the sonar source are likely to experience masking on a much longer time scale than those exposed to traditional sonars. The frequency range at which high duty cycle systems operate overlaps the vocalization frequency of a number of mid-frequency cetaceans (e.g., ESA-listed sperm whales). Continuous noise at the same frequency of communicative vocalizations may cause disruptions to communication, social interactions, and acoustically-mediated cooperative behaviors such as foraging or reproductive activities. With mid-frequency high duty cycle systems, there is the potential for the sonar signals to mask important environmental cues like predator vocalizations (e.g. killer whales), possibly affecting survivorship for targeted animals. While there are currently no available studies of the impacts of high duty cycle sonars on cetaceans, masking due to these systems is likely analogous to masking produced by other continuous sources (e.g. vessel noise and low-frequency cetaceans), and will likely have similar short-term consequences,

though longer in duration due to the duration of the masking noise. These may include changes to vocalization amplitude and frequency (Brumm and Slabbekoorn 2005; Hotchkin and Parks 2013) and behavioral impacts such as avoidance of the area and interruptions to foraging or other essential behaviors (Gordon et al. 2003; Sivle et al. 2016). Long-term consequences could include changes to vocal behavior and vocalization structure (Foote et al. 2004; Parks et al. 2007), abandonment of habitat if masking occurs frequently enough to significantly impair communication (Brumm and Slabbekoorn 2005), a potential decrease in survivorship if predator vocalizations are masked (Brumm and Slabbekoorn 2005), and a potential decrease in recruitment if masking interferes with reproductive activities or mother-calf communication (Gordon et al. 2003).

8.2.1.1.4 Behavioral Reactions

Acoustic stimuli in the marine environment can cause a behavioral response in marine mammals and can also influence how or if a marine mammal responds to a sound such as the presence of predators, prey, or conspecifics. The response of a marine mammal to anthropogenic sound may depend on the frequency, duration, temporal pattern and amplitude of the sound, as well as the animal's prior experience with the sound and their behavioral state (i.e., what the animal is doing and their energetic needs at the time of the exposure) (Ellison et al. 2012). The distance from the sound source and whether it is approaching or moving away can also affect the way an animal responds to a sound (Wartzok et al. 2003).

A review of marine mammal responses to anthropogenic sound was first conducted by Richardson et al. (1995b). Other reviews (Gomez et al. 2016; Nowacek et al. 2007; Southall et al. 2007a) addressed studies conducted since 1995 and focused on observations where the received sound level of the exposed cetacean was known or could be estimated, and also examined the role of context. Southall et al. (2007a) synthesized data from many behavioral studies and observations to determine the likelihood of behavioral reactions at specific sound levels. Southall et al. (2016) reviewed the range of experimental field studies that have been conducted to measure behavioral responses of cetaceans to sonar. While, in general, the louder the sound source the more intense the behavioral response, it was clear that the proximity of a sound source and the animal's experience, motivation, and conditioning were also critical factors influencing the response (Southall et al. 2007a; Southall et al. 2016).

Ellison et al. (2012) outlined an approach to assessing the effects of sound on marine mammals that incorporates these contextual-based factors. They recommend considering not just the received level of sound, but also what activity the animal is engaged in, the nature and novelty of the sound (i.e., is this a new sound from the animal's perspective), and the distance between the sound source and the animal. They submit that this "exposure context" as described, greatly influences the type of behavioral response exhibited by the animal. Forney et al. (2017) also note that an apparent lack of response (e.g., no displacement or avoidance of a sound source) may not necessarily mean there is no cost to the individual or population, as some resources or habitats

may be of such high value that animals may choose to stay, even when experiencing stress or hearing loss. Forney et al. (2017) recommend considering both the costs of remaining in an area of noise exposure such as TTS, PTS or masking, which could lead to an increased risk of predation or other threats or a decreased capability to forage, and the costs of displacement, including potential increased risk of vessel strike or bycatch, increased risks of predation or competition for resources, or decreased habitat suitability for foraging, resting, or socializing.

Sonar and other transducers can range in frequency (from less than 1 kilohertz to over 200 kilohertz) and duty cycles (from one ping per minute to an almost continuous sound). These acoustic sources can also be stationary or operated from a moving platform, and there can one or multiple sources present at a time. This variability in parameters associated with sonar and other transducers makes the estimation of behavioral responses to these sources difficult, with observed responses ranging from no apparent change in behavior to more severe responses that could lead to some costs to the animal. Responses may also occur in the presence of different contextual factors regardless of received level, including the proximity and number of vessels, the behavioral state and prior experience of an individual, and even characteristics of the signal itself or the propagation of the signal through the environment. For some ESA-listed marine mammal species little or no data exist on behavioral responses to any sound source, and so these species have been grouped into taxonomic groups (e.g., mysticetes, odontocetes, and pinnipeds) from which general response information can be inferred.

Cetacean behavioral response studies have been conducted through the collaboration of various research and government organizations in Bahamian, U.S. (e.g., off Southern California, Hawaii, and the east coast), Mediterranean, Australian, and Norwegian waters. These studies have attempted to define and measure responses of cetaceans to controlled exposures of sonar and other sounds to understand their potential impacts. While controlling for as many variables as possible (e.g., the distance and movement of the source), these studies also introduce additional variables that do not normally occur in a real Navy training or testing activity, including the tagging of whales, following the tagged animals with multiple vessels, and continually approaching the animal to create a dose escalation. In addition, distances of the sound source from the whales during behavioral response studies were always within 1 to 8 km. Some of these studies have suggested that ramping-up a source from a lower source level would act as a protective measure to mitigate higher order (e.g., TTS or PTS) impacts of sonar. However, this practice may only be effective for more responsive animals, and for short durations (e.g., five minutes.) of ramp-up (von Benda-Beckmann et al. 2016; von Benda-Beckmann et al. 2014; Wensveen et al. 2017).

Passive acoustic monitoring and visual observational behavioral response studies have been conducted on Navy ranges, taking advantage of the existing seafloor hydrophones and real testing and training activity and associated sources to assess behavioral responses (Deakos and Richlen 2015; Henderson et al. 2016; Manzano-Roth et al. 2016; Martin et al. 2015; Mccarthy et al. 2011; Mobley and Deakos 2015; Moretti et al. 2014; Tyack et al. 2011b). In addition,

extensive aerial, visual, and acoustic monitoring is conducted before, during and after training events to watch for behavioral responses during training and look for injured or stranded animals after training (Campbell et al. 2010; Farak et al. 2011; HDR 2011; Navy 2011b; Navy 2013a; Navy 2014b; Navy 2015; Norris et al. 2012; Smultea and Mobley 2009; Smultea et al. 2009; Trickey et al. 2015a). When visual and passive acoustic data collected during a training event are combined with ship movements and sonar use they provide a unique and realistic scenario for analysis. During all of these monitoring efforts, only a few behavioral responses were observed, (discussed below in Mysticetes and Odontocetes – Behavioral Response) and no injured or dead animal was observed that was directly related to a training event (some dead animals were observed but typically before the event or appeared to have been deceased prior to the event (Smultea et al. 2011). It should be noted that passive acoustic studies are limited to observations of vocally-active cetaceans and visual studies are limited to what can be observed at the surface.

Harris and Thomas (2015) highlighted additional research approaches that may provide further information on behavioral responses to sonars and other transducers, beyond behavioral response type studies or passive acoustic monitoring, including conducting controlled exposures on captive animals with scaled sound sources (smaller sized and deployed at closer proximity) and on wild animals with both scaled and real but directed sound sources. Captive studies on odontocete species can provide insight into how these animals may respond in the wild (see Odontocetes – Behavioral Response below for details). The captive studies typically represent a more controlled approach, which allow researchers to better estimate the direct impact of the received level of sound leading to behavioral responses, and to potentially link behavioral to physiological responses. However, there are still contextual factors that must be acknowledged, including previous training to complete tasks and the presence of food rewards upon completion. There are no corresponding captive studies on mysticete whales, therefore some of the responses to higher level exposures must be extrapolated from odontocetes.

8.2.1.1.5 Mysticetes – Behavioral Response

The responses of mysticetes to sonar and other duty-cycled tonal sounds are highly dependent upon the characteristics of the signal, the behavioral state of the animal, the particular sensitivity and previous experience of an individual, and other contextual factors including distance of the source, movement of the source, and the physical presence of vessels in addition to the sonar (Goldbogen et al. 2013; Harris et al. 2015; Martin et al. 2015; Sivle et al. 2015). Behavioral response studies have been conducted over a variety of contextual and behavioral states, helping to identify which contextual factors may lead to a response beyond just the received level of the sound. Observed reactions during behavioral response studies have not been consistent across individuals based on received sound levels alone, and likely were the result of complex interactions between these contextual factors.

Surface feeding blue whales did not show a change in behavior in response to mid-frequency simulated and real sonar sources with received levels between 90 and 179 dB re 1 µPa, but deep

feeding and non-feeding whales showed temporary reactions including cessation of feeding, reduced initiation of deep foraging dives, generalized avoidance responses, and changes to dive behavior (DeRuiter et al. 2017; Goldbogen et al. 2013). These findings indicate that the behavioral state of the animal plays a role in the type and severity of a behavioral response. In fact, when the prey field was mapped and used as a covariate in similar models looking for a response in blue whales, the response in deep-feeding behavior by blue whales was even more apparent, reinforcing the need for contextual variables to be included when assessing behavioral responses (Friedlaender et al. 2016b; Southall et al. 2019a). However, even when responses did occur, the animals quickly returned to their previous behavior after the sound exposure ended (Goldbogen et al. 2013).

A behavioral response study by Harris et al. (2019) looked at the exposure of lunge feeding rates blue, fin, and humpback whales to simulated naval sonar. Results of their study showed that regardless of exposure levels, blue and fin whale lunge rates remained similar to baseline. However, their study did demonstrate that humpback whales – which were exposed to the highest sound levels of controlled exposures of simulated sonar – did show a greater degree of feeding disruption than either of the other two species, both during and up to 15 minutes after sonar exposure. In another study, humpback whales exposed to a three kilohertz pinger meant to act as a net alarm to prevent entanglement did not respond or change course, even when within 500 meters (Harcourt et al. 2014). Five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives. In this case, the alarm was comprised of a mixture of signals with frequencies from 500 to 4500 Hz, was long in duration (lasting several minutes), and was purposely designed to elicit a reaction from the animals as a prospective means of protecting them from ship strikes (Nowacek et al. 2004). Although the animals' received SPL was similar in the latter two studies (133–150 dB re 1 µPa), the frequency, duration, and temporal pattern of signal presentation were different.

Humpback whales in another behavioral response experiment in Australia also responded to a two kilohertz tone stimulus by changing their course during migration to move more offshore and surfacing more frequently (Dunlop et al. 2013). Humpback whales in a Norwegian behavioral response study may have habituated slightly between the first and second sonar exposure (Sivle et al. 2015), and actually responded more severely to killer whale vocalization playbacks than they did to the sonar playbacks. Several humpback whales have been observed during aerial or visual surveys during Navy training events involving sonar (Harris et al. 2019; Henderson et al. 2019; Sivle et al. 2016; Wensveen et al. 2017). No avoidance or other behavioral responses were ever noted, even when the whales were observed within 5 km of a vessel with active (or possibly active) sonar and maximum received levels were estimated to be between 135 and 161 dB re 1 µPa (e.g., Mobley 2011; Mobley and Pacini 2012; Smultea et al. 2009). One group of humpback whales approached a vessel with active sonar so closely that the sonar was shut-down and the vessel slowed. The animals continued approaching and swam under the bow of the vessel (Navy 2011a). Another group of humpback whales continued heading

towards a vessel with active sonar as the vessel was moving away for almost 30 minutes, with an estimated median received level of 143 dB re 1 µPa. This group was observed producing surface active behaviors such as pectoral fin slaps, tail slaps and breaches; however, these are very common behaviors in competitive pods during the breeding season and were not considered to have occurred in response to the sonar (Mobley et al. 2012).

The strongest baleen whale response in any behavioral response study was observed in a minke whale in the aforementioned Norwegian behavioral response study, where the whale responded at 146 dB re 1 µPa by strongly avoiding the sound source (Harris et al. 2019; Kvadsheim et al. 2017; Sivle et al. 2015). Although the minke whale increased its swim speed, directional movement and respiration rate, none of these were greater than rates observed in baseline behavior, and its dive behavior remained similar to baseline dives. A minke whale tagged in a Southern California behavioral response study also responded by increasing its directional movement, but maintained its speed and dive patterns, so did not demonstrate as strong of a response (Kvadsheim et al. 2017). In addition, the minke whale demonstrated some of the same avoidance behavior during the controlled ship approach with no sonar, indicating at least some of the response was to the vessel (Kvadsheim et al. 2017). Martin et al. (2015) found that the density of calling minke whales was reduced during periods of Navy training involving sonar relative to the periods before training, and increased again in the days after training was completed. The responses of individual whales could not be assessed, so in this case it is unknown whether the decrease in calling animals indicated that the animals left the range, or simply ceased calling. Similarly, minke whale detections made using marine acoustic recording instruments off Jacksonville, Florida were reduced or ceased altogether during periods of sonar use (Navy 2013c; Norris et al. 2012) especially with an increased ping rate (Charif et al. 2015). Two minke whales also stranded in shallow water after the U.S. Navy training event in the Bahamas in 2000. These animals were successfully returned to deep water with no physical examinations. Because there were no physical examinations of these animals, no final conclusions were drawn on whether the sonar led to their stranding (Commerce 2001; Filadelfo et al. 2009a; Filadelfo et al. 2009b).

Baleen whales have also been exposed to lower frequency sonars, with the hypothesis that these whales may react more strongly to lower frequency sounds that overlap with their vocalization range. One series of studies was undertaken in 1997 to 1998 pursuant to the Navy's Low-Frequency Sound Scientific Research Program. The frequency bands of the low-frequency sonars used were between 100 and 500 Hz, with received levels between 115 and 150 dB re 1 µPa, and the sound source was always stationary. Fin and blue whales were targeted on foraging grounds, singing humpback whales were exposed on breeding grounds, and gray whales were exposed during migratory behavior. These studies found only short-term responses to low-frequency sound by some fin and humpback whales, including changes in vocal activity and avoidance of the source vessel, while other fin, humpback, and blue whales did not respond at all. When the source was in the path of migrating gray whales, they changed course up to 2 km

to avoid the sound, but when the source was outside their path, little response was observed (Clark and Fristrup 2001; Croll et al. 2001a; Fristrup et al. 2003; Miller et al. 2000; Nowacek et al. 2007). Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were also not found to affect dive times of humpback whales in Hawaiian waters (Frankel and Clark 2000).

Opportunistic passive acoustic based studies have also detected behavioral responses to sonar. Blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low-frequency calls usually associated with feeding behavior, beginning at received levels of 110 to 120 dB re 1 µPa (Melcon et al. 2012). In another example, Risch et al. (2012); (2014) concluded that reductions in humpback whale songs in the Stellwagen Bank National Marine Sanctuary were a result of an Ocean Acoustic Waveguide Remote Sensing experiment occurring about 200 km away from the whales location. However, Gong et al. (2014) analyzed the same data set while also looking at the presence of herring in the region, and found that the singing humpbacks were actually located on nearby Georges Bank and not on Stellwagen, and that the song rate in their data did not change in response to the Ocean Acoustic Waveguide Remote Sensing experiment, but could be explained by natural causes.

Although some strong responses have been observed in mysticetes to sonar and other active acoustic sources (e.g., the single minke whale), for the most part mysticete responses appear to be fairly moderate across all received levels. While some responses such as cessation of foraging or changes in dive behavior could carry short-term impacts, in all cases behavior returned to normal after the signal stopped. Mysticete responses also seem to be highly mediated by behavioral state, with no responses occurring in some behavioral states, and contextual factors and signal characteristics having more impact than received level alone. Many of the contextual factors resulting from the behavioral response studies (e.g., close approaches by multiple vessels or tagging) would not likely occur during real Navy testing and training scenarios. While there is a lack of data on behavioral responses of mysticetes to continuously active sonars, these species are known to be able to habituate to novel and continuous sounds (Nowacek et al. 2004), suggesting that they could have similar responses to high duty cycle sonars. No significant behavioral responses such as panic or stranding have been observed during monitoring of actual training exercises (Navy 2011b; Navy 2014a; Smultea et al. 2009; Watwood et al. 2012).

8.2.1.1.6 Odontocetes – Behavioral Response

Behavioral response studies have been conducted on odontocete species since 2007, with a focus on beaked whale (not ESA-listed) responses to active sonar transmissions or controlled exposure playback of simulated sonar on various military ranges (Claridge and Durban 2009; Claridge et al. 2009; Henderson et al. 2015; Isojunno et al. 2020; Mccarthy et al. 2011; Moretti et al. 2009; Southall et al. 2013; Southall et al. 2012; Southall et al. 2014; Wensveen et al. 2019). Though below we will discuss results of behavioral response studies on many odontocete species (e.g.,

beaked whales), sperm and killer whales are the only odontocetes in the action area listed under the ESA.

Results to date suggest that sperm whales are not as sensitive to anthropogenic sound sources as some other odontocetes, such as beaked whales (Southall et al. 2009). However, in response to seismic surveys and naval sonar, sperm whales have demonstrated avoidance, changes in locomotion/orientation, changes in dive profiles, cessation of foraging, cessation of resting, and changes in vocal behavior (Isojunno et al. 2016; Miller et al. 2009; Miller et al. 2012; Sivle et al. 2012). Sperm whales may be more sensitive to larger amounts of cumulative sound energy they receive, rather than higher sound amplitudes during sonar exercises. Sperm whales in and around the Norwegian Sea were found to halt foraging behavior for longer periods of time following exposure to continuous sonar than they were following exposure to pulsed sonar of similar amplitudes, although these behavioral changes became more pronounced with increasing pulsed sonar amplitudes (Isojunno et al. 2020).

Observed reactions by Blainville's, Cuvier's, and Baird's beaked whales to mid-frequency sonar sounds have included cessation of clicking, termination of foraging dives, changes in direction to avoid the sound source, slower ascent rates to the surface, and other unusual dive behavior (Boyd et al. 2008; Cholewiak et al. 2017; Deruiter et al. 2013a; Joyce et al. 2019; Miller et al. 2015; Southall et al. 2019a; Stimpert et al. 2014; Tyack et al. 2011a). Falcone et al. (2017) modeled deep and shallow dive durations, surface interval durations, and inter-deep dive intervals of Cuvier's beaked whales against predictor values that included helicopter-dipping; mid-power mid-frequency active sonar; and hull-mounted, high-power mid-frequency active sonar along with other, non-mid-frequency active sonar predictors. They found both shallow and deep dive durations to increase as the proximity to both mid- and high-powered sources decreased, and found surface intervals and inter-deep dive intervals to also increase in the presence of both types of sonars, although surface intervals shortened during periods of no mid-frequency active sonar. The responses to the mid-power mid-frequency active sonar at closer ranges were comparable to the responses to the higher source level ship sonar, again highlighting the importance of proximity. This study also supports context as a response factor as helicopter-dipping sonars, which are shorter duration and randomly located, are more difficult for beaked whales to predict or track and therefore potentially more likely to cause a response, especially when they occur at closer distances (6 to 25 km in this study).

A response was observed in a northern bottlenose whale, which conducted the longest and deepest dive on record for that species after the sonar exposure and continued swimming away from the source for over seven hours (Miller et al. 2015). Responses occurred at received levels between 95 and 150 dB re 1 µPa. All of these exposures occurred within 1 to 8 km of the focal animal, within a few hours of tagging the animal, and with one or more boats within a few kilometers to observe responses and record acoustic data. One Cuvier's beaked whale was also incidentally exposed to real Navy sonar located over 100 km away, and the authors did not detect similar responses at comparable received levels. Received levels from the mid-frequency active

sonar signals from the controlled and incidental exposures were calculated as 84 to 144 and 78 to 106 dB re 1 µPa, respectively, indicating that context of the exposures (e.g., source proximity, controlled source ramp-up) may have been a significant factor in the responses to the simulated sonars (Deruiter et al. 2013a). Furthermore, recent long-term tagging work has demonstrated that the longer duration dives, considered a behavioral response by Deruiter et al. (2013a), fell within the normal range of dive durations found for eight tagged Cuvier's beaked whales on the Southern California Offshore Range (Schorr et al. 2014). However, the longer inter-deep dive intervals found by Deruiter et al. (2013a) were among the longest found by Schorr et al. (2014) and could indicate a response to sonar. In addition, Williams et al. (2017) note that in normal deep dives or during fast swim speeds, beaked whales and other cetaceans use strategies to reduce their stroke rates, including leaping or wave surfing when swimming, and interspersing glides between bouts of stroking when diving. They determined that in the post-exposure dives by the tagged Cuvier's beaked whales described in Deruiter et al. (2013a), the whales ceased gliding and swam with almost continuous strokes. This change in swim behavior was calculated to increase metabolic costs about 30.5 percent and increase the amount of energy expended on fast swim speeds from 27 to 59 percent of their overall energy budget. This repartitioning of energy was detected in the model up to 1.7 hours after the single sonar exposure. Therefore, while the overall post-exposure dive durations were similar, the metabolic energy calculated by Williams et al. (2017) was higher.

On Navy ranges, Blainville's beaked whales located on the range appeared to move off-range during sonar use and returned only after the sonar transmissions have stopped, sometimes taking several days to do so (Claridge and Durban 2009; Claridge et al. 2009; Henderson et al. 2015; Mccarthy et al. 2011; Moretti et al. 2009; Tyack et al. 2011a). Blainville's beaked whales remained on the Navy range to forage throughout the rest of the year (Henderson et al. 2016), and photo identification studies in the Southern California Range Complex have identified approximately 100 individual Cuvier's beaked whales, with 40 percent having been seen in one or more prior years, with re-sightings up to seven years apart, indicating a possibly resident population on the range (Falcone and Schorr 2014; Falcone et al. 2009). These results suggest that the range areas studied represent preferred foraging habitat regardless of the effects of the noise, and that there may be no long term consequences of the sonar activity on beaked whales in these areas.

Tyack et al. (2011a) hypothesized that beaked whale responses to sonar may represent an anti-predator response. Such anti-predator responses in Blainville's beaked whales and Cuvier's beaked whales may include overlapping foraging times among group members and coordinated silent ascents in unpredictable directions (de Soto et al. 2020). To test the sonar/anti-predator hypothesis, vocalizations of a potential predator—a killer whale—were also played back to a Blainville's beaked whale. This exposure resulted in a similar but more pronounced reaction than that elicited by sonar playback, which included longer inter-dive intervals and a sustained straight-line departure of more than 20 km from the area (Allen et al. 2014; Tyack et al. 2011a).

This anti-predator hypothesis was also tested by playing back killer whale vocalizations to pilot whales, sperm whales, and even other killer whales, to determine responses by both potential prey and conspecifics (Miller et al. 2011; Miller et al. 2012). Results varied, from no response by killer whales to an increase in group size and attraction to the source in pilot whales (Cure et al. 2012).

While there has been a focus on beaked whale responses to sonar, other species have been studied during behavioral response studies as well, including pilot whales, killer whales, and sperm whales. Responses by these species have also included horizontal avoidance, changes in behavioral state, changes in dive behavior, and reduced breathing rate (Antunes et al. 2014; Miller et al. 2011; Miller et al. 2014; Miller et al. 2012).

One study reported the temporary separation of a killer whale calf from its group during exposure to mid-frequency sonar playback (Miller et al. 2011). The separation event occurred during the third (of three) sonar exposures of a group of seven killer whales in a fjord surrounding the Lofoten Islands, Norway. During the first exposures, some calling was observed before the initial ping, and calling increased and continued throughout both the mid-frequency active and low-frequency active sonar exposures (Miller et al. 2011). A second mid-frequency active sonar exposure was conducted to try to achieve a closer approach to the whales after they had moved into a deeper part of the fjord. The whales made a strong change of direction during a long dive in the ramp-up phase of the second mid-frequency active sonar exposure and increased speed immediately after the dive (Miller et al. 2011). At the end of the second mid-frequency active sonar exposure, the smallest calf in the group was seen traveling alone, more than 1,000 meters behind the location of the group. The calf rejoined the group after traveling alone for at least 86 minutes (Miller et al. 2011).

Received level thresholds at the onset of avoidance behavior were generally higher for pilot whales (mean 150 dB re 1 µPa) and sperm whales (mean 140 dB re 1 µPa) than killer whales (mean 129 dB re 1µPa) (Antunes et al. 2014; Miller et al. 2011; Miller et al. 2012). A close examination of the tag data from the Norwegian groups showed that responses seemed to be behaviorally or signal frequency mediated. For example, killer whales only changed their dive behavior when doing deep dives at the onset of one to two kilohertz sonar (sweeping across frequencies), but did not change their dive behavior if they were deep diving during six to seven kilohertz sonar (sweeping across frequencies). Nor did they change their dive behavior if they were conducting shallow dives at the onset of either type of sonar. Similarly, pilot whales and sperm whales performed normal deep dives during six to seven kilohertz sonar, while during one to two kilohertz sonar the pilot whales conducted fewer deep dives and the sperm whales performed shorter and shallower dives (Sivle et al. 2012). In addition, pilot whales were also more likely to respond to lower received levels when non-feeding than feeding during six to seven kilohertz sonar exposures, but were more likely to respond at higher received levels when non-feeding during one to two kilohertz sonar exposures. Furthermore, pilot whales exposed to a 38 kilohertz downward-facing echosounder did not change their dive and foraging behavior

during exposure periods, although the animals' heading variance increased and fewer deep dives were conducted (Quick et al. 2017). In contrast, killer whales were more likely to respond to either sonar type when non-feeding than when feeding (Harris et al. 2015). These results again demonstrate that the behavioral state of the animal mediates the likelihood of a behavioral response, as do the characteristics (e.g., frequency) of the sound source itself.

Other responses during behavioral response studies included the synchronization of pilot whale surfacings with sonar pulses during one exposure, possibly as a means of mitigating the sound (Wensveen et al. 2015), and mimicry of the sonar with whistles by pilot whales (Alves et al. 2014), false killer whales (Deruiter et al. 2013c), and Risso's dolphins (Smultea et al. 2012). In contrast, in another study melon-headed whales had "minor transient silencing" (a brief, non-lasting period of silence) after each six to seven kilohertz signal, and (in a different oceanographic region) pilot whales had no apparent response (Deruiter et al. 2013b). The probability of detecting delphinid vocalizations (whistles, clicks, and buzzes) increased during periods of sonar relative to the period prior to sonar in a passive acoustic study using Marine Autonomous Recording Units in the Jacksonville Range Complex, while there was no impact of sonar to the probability of detecting sperm whale clicks (Charif et al. 2015; Navy 2013a).

In addition, killer whale sighting data from the same region in Norway as the behavioral response study was used to compare the presence or absence of whales from other years against the period with sonar. The authors found a strong relationship between the presence of whales and the abundance of herring, and only a weak relationship between the whales and sonar activity (Kuningas et al. 2013). Baird et al. (2013) also tagged four shallow-diving odontocete species (rough toothed dolphins, pilot whales, bottlenose dolphins, and false killer whales) in Hawaii off the Pacific Missile Range Facility before Navy training exercises. None of the tagged animals demonstrated a large-scale avoidance response to the sonar as they moved on or near the range, in some cases even traveling towards areas of higher noise levels, while estimated received SPLs varied from 130 to 168 dB re 1 µPa and distances from sonar sources ranged between 3.2 and 94.4 km. However, one pilot whale did have reduced dive rates (from 2.6 dives per hour before to 1.6 dives per hour during) and deeper dives (from a mean of 124 meters to 268 meters) during a period of sonar exposure. Baird et al. (2016) also tagged four short-finned pilot whales from both the resident island-associated population and from the pelagic population. The core range for the pelagic population was over 20 times larger than for the island-associated population, leading the researchers to hypothesize that that likelihood of exposure to mid-frequency active sonar, and therefore the potential for response, would be very different between the two populations. These diverse examples demonstrate that responses can be varied, are often context- and behaviorally-driven, and can be species and even exposure specific.

Other opportunistic observations of behavioral responses to sonar have occurred as well, although in those cases it is difficult to attribute observed responses directly to the sonar exposure, or to know exactly what form the response took. For example, both sperm and pilot whales potentially ceased sound production during the Heard Island feasibility test, with

transmissions centered at 57 Hz and up to 220 dB re 1 µPa (Bowles et al. 1994), although it could not be determined whether the animals ceased sound production or left the area. In May 2003, killer whales in Haro Strait, Washington exhibited what were believed by some observers to be aberrant behaviors, during which time the USS Shoup was in the vicinity and engaged in mid-frequency active sonar operations. Sound fields modeled for the USS Shoup transmissions (Fromm 2009; Navy 2003; NMFS 2005a) estimated a mean received SPL of approximately 169 dB re 1 µPa at the location of the killer whales at the closest point of approach between the animals and the vessel (estimated SPLs ranged from 150 to 180 dB re 1 µPa). However, attributing the observed behaviors to any one cause is problematic given there were six nearby whale watch vessels surrounding the pod, and subsequent research has demonstrated that "Southern Resident killer whales modify their behavior by increasing surface activity (breaches, tail slaps, and pectoral fin slaps) and swimming in more erratic paths when vessels are close" (NOAA 2014b). Several odontocete species, including bottlenose dolphins, Risso's dolphins, Pacific white-sided dolphins, and common dolphins have been observed near the Southern California Offshore Range during periods of mid-frequency active sonar; responses included changes in or cessation of vocalizations, changes in behavior, and leaving the area (at the highest received levels animals were not present in the area at all) (Henderson et al. 2014). However, these observations were conducted from a vessel off-range, and so any observed responses could not be attributed to the sonar with any certainty. Research on sperm whales in the Caribbean in 1983 coincided with the U.S. intervention in Grenada, where animals were observed scattering and leaving the area in the presence of military sonar, presumably from nearby submarines (Watkins et al. 1985). The authors did not report received levels from these exposures and reported similar reactions from noise generated by banging on their boat hull; therefore, it was unclear if the sperm whales were reacting to the sonar signal itself or to a potentially new unknown sound in general.

During aerial and visual monitoring of Navy training events involving sonar, rough-toothed dolphins and unidentified dolphins were observed approaching the vessel with active sonar as if to bowride, while spotted dolphins were observed nearby but did not avoid or approach the vessel (HDR 2011; Navy 2011a; Watwood et al. 2012). During small boat surveys near the Navy's Southern California Offshore Range in southern California, more dolphins were encountered in June compared to a similar survey conducted the previous November after seven days of mid-frequency sonar activity. It was not investigated if this change was due to the sonar activity or was a seasonal difference that could be observed in other years (Campbell et al. 2010). There were also fewer passive acoustic dolphin detections during and after longer sonar activities in the Marianas Islands Range Complex, with the post-activity absence lasting longer than the mean dolphin absence of two days when sonar was not present (Munger et al. 2014; Munger et al. 2015).

Acoustic harassment devices and acoustic deterrent devices have been used to deter cetaceans from approaching fishing gear both to prevent entanglement and to reduce depredation (taking

fish) (85 FR 53763). These devices have been used successfully to deter harbor porpoises and beaked whales from getting entangled in fishing nets. For example, Kyhn et al. (2015) tested two types of pingers, one with a ten kilohertz tone and one with a broadband 30 to 160 kilohertz sweep. Porpoise detection rates were reduced by 65 percent for the sweep and 40 percent for the tone and, while there was some gradual habituation after the first two to four exposures, longer term exposures (over 28 days) showed no evidence of additional habituation. Kindt-Larsen et al. (2018) also report on the effectiveness of pingers to deter harbor porpoise from depredating fishing nets and also indicate no evidence of habituation. Omeyer et al. (2020) found that a Banana Pinger (50 to 120 kilohertz, sound pressure level 145 dB re 1 µPa) was an effective harbor porpoise deterrent and there was no evidence of habituation over an eight-month period. Additionally, sperm whales in the Caribbean stopped vocalizing when presented with sounds from nearby acoustic pingers (Watkins and Schevill 1975). Acoustic harassment devices used to deter cetaceans from depredating long lines or aquaculture enclosures have proven less successful. For example, Tixier et al. (2014) used a 6.5 kilohertz pinger with a source level of 195 dB re 1 µPa on a longline to prevent depredation by killer whales, and although two groups of killer whales fled over 700 meters away during the first exposure, they began depredating again after the third and seventh exposures, indicating rapid habituation. In a review of cetacean deterrents, Schakner and Blumstein (2013) point out that both the characteristics of deterrents and the motivation of the animal play a role in the effectiveness of acoustic harassment devices. Deterrents that are strongly aversive either simulate a predator or are otherwise predictive of a threat are those more likely to be effective, unless the animal habituates to the signal or learns that there is no true threat associated with the signal. In some cases the net pingers may create a "dinner bell effect," where cetaceans have learned to associate the signal with the availability of prey (Jefferson and Curry 1996; Schakner and Blumstein 2013). This may be why net pingers have been more successful at reducing entanglements for harbor porpoise and beaked whales because these species are not depredating from the nets but are getting entangled when foraging in the area and are unable to detect the net (Carretta and Barlow 2008; Schakner and Blumstein 2013). Additional behavioral studies have been conducted with captive harbor porpoises using acoustic alarms, such as those used on fishing nets to help deter cetaceans from becoming caught or entangled (Kastelein et al. 2001; Kastelein et al. 2006). These studies have found that high-frequency sources with varied duration, interval, and sweep characteristics can prove to be effective deterrents for harbor porpoises (Kastelein et al. 2017; van Beest et al. 2017).

Controlled experiments have also been conducted on captive animals to estimate received levels at which behavioral responses occur. In one study, bottlenose dolphin behavioral responses were recorded when exposed to three kilohertz sonar-like tones between 115 and 185 dB re 1 µPa (Houser et al. 2013a), and in other studies bottlenose dolphins and beluga whales were presented with one-second tones up to 203 dB re 1 µPa to measure TTS (Finneran et al. 2001; Finneran et al. 2005; Finneran and Schlundt 2004; Schlundt et al. 2000). During these studies, responses included changes in respiration rate, fluke slaps, and a refusal to participate or return to the location of the sound stimulus. This refusal included what appeared to be deliberate attempts to

avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al. 2002; Schlundt et al. 2000). In the behavioral response experiment, bottlenose dolphins demonstrated a 50 percent probability of response at 172 dB re 1 µPa over ten trials. In the TTS study bottlenose dolphins exposed to one-second intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 µPa, and beluga whales did so at received levels of 180 to 196 dB re 1 µPa and above. In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway et al. 1997; Schlundt et al. 2000). Götz et al. (2020) found that magnitudes of startle responses in bottlenose dolphins increased exponentially with increasing sound received levels but decreased with increased rise times in the sound signals. While animals were commonly reinforced with food during these studies, the controlled environment and ability to measure received levels provide insight on received levels at which animals will behaviorally respond to noise sources.

Behavioral responses to a variety of sound sources have been studied in harbor porpoises, including acoustic alarms (Kastelein et al. 2001; Kastelein et al. 2006), emissions for underwater data transmission (Kastelein et al. 2005), and tones, including one to two kilohertz and six to seven kilohertz sweeps with and without harmonics (Kastelein et al. 2014d), and 25 kilohertz with and without sidebands (Kastelein et al. 2015a; Kastelein et al. 2015b). Responses include increased respiration rates, more jumping, or swimming further from the source, but responses were different depending on the source. For example, harbor porpoises responded to the one to two kilohertz upsweep at 123 dB re 1 µPa, but not to the downsweep or the six to seven kilohertz tonal at the same level (Kastelein et al. 2014d). When measuring the same sweeps for a startle response, the 50 percent response threshold was 133 and 101 dB re 1 µPa for one to two kilohertz and six to seven kilohertz sweeps respectively when no harmonics were present, and decreased to 90 dB re 1 µPa for one to two kilohertz sweeps with harmonics present (Kastelein et al. 2014d). Harbor porpoises responded to broadband signals up to 44 kilohertz with a slight respiration response at 117 dB re 1 µPa and an avoidance response at 139 dB re 1 µPa, but another source with a fundamental (lowest and strongest) frequency of 18 kilohertz didn't have an avoidance response until 151 dB re 1 µPa (Kastelein et al. 2014a). Exposure of the same acoustic pinger to a striped dolphin under the same conditions did not elicit a response (Kastelein et al. 2006), again highlighting the importance of understanding species' differences in the tolerance to underwater noise, although sample sizes in these studies was small so these could reflect individual differences as well.

Behavioral responses by odontocetes to sonar and other transducers appear to run the full gamut from no response at all to responses that could lead to long-term consequences for individual animals (e.g., mother-calf separation). This is likely in part due to the fact that this taxonomic group is so broad and includes some of the most sensitive species (e.g., beaked whales and harbor porpoise) as well as some of the least sensitive species (e.g., bottlenose dolphins). This is also the only group for which both field behavioral response studies and captive controlled exposure experiments have been conducted, leading to the assessment of both contextually-

driven responses as well as dose-based responses. This wide range in both exposure situations and individual- and species-sensitivities makes reaching general conclusions difficult. However, it does appear as though exposures in close proximity, with multiple vessels that approach the animal, lead to higher-level responses in most odontocete species regardless of received level or behavioral state. In contrast, in more "real-world" exposure situations, with distant sources moving in variable directions, behavioral responses appear to be driven by behavioral state, individual experience, or species-level sensitivities. These responses may also occur more in-line with received level such that the likelihood of a response would increase with increased received levels. These "real-world" responses are more likely to be short-term, lasting the duration of the exposure.

8.2.1.1.7 Pinnipeds – Behavioral Response

No field studies of pinniped behavioral responses to sonar have been conducted; however, there are several captive studies on some pinniped and odontocete species that can provide insight into how these animals may respond in the wild. The captive studies typically represent a more controlled approach, which allow researchers to better estimate the direct impact of the received level of sound leading to behavioral responses, and to potentially link behavioral to physiological responses. However, there are still contextual factors that must be acknowledged, including previous training to complete tasks and the presence of food rewards upon completion.

Different responses displayed by captive and wild phocid seals to sound judged to be "unpleasant" or threatening have been reported, including habituation by captive seals (they did not avoid the sound), and avoidance behavior by wild seals (Götz and Janik 2010). Captive seals received food (reinforcement) during sound playback, while wild seals were exposed opportunistically. These results indicate that motivational state (e.g., reinforcement via food acquisition) can be a factor in whether or not an animal tolerates or habituates to novel or unpleasant sounds.

Another study found that captive hooded seals reacted to 1–7 kilohertz sonar signals, in part with displacement (i.e., avoidance) to the areas of least SPL, at levels between 160 and 170 dB re 1 µPa (Kvadsheim et al. 2010b); however, the animals adapted to the sound and did not show the same avoidance behavior upon subsequent exposures. Captive harbor seals responded differently to three signals at 25 kilohertz with different waveform characteristics and duty cycles. The seals responded to the frequency modulated signal at received levels over 137 dB re 1 µPa by hauling out more, swimming faster, and raising their heads or jumping out of the water, but did not respond to the continuous wave or combination signals at any received level (up to 156 dB re 1 µPa) (Kastelein et al. 2015e). Captive California sea lions were exposed to mid-frequency sonar at various received levels (125–185 dB re 1 µPa) during a repetitive task (Houser et al. 2013b). Behavioral responses included a refusal to participate, hauling out, an increase in respiration rate, and an increase in the time spent submerged. Young animals (less than two years old) were more likely to respond than older animals. Dose-response curves were developed both including and

excluding those young animals. The majority of responses below 155 dB re 1 µPa were changes in respiration, whereas over 170 dB re 1 µPa more severe responses began to occur (such as hauling out or refusing to participate); many of the most severe responses came from the younger animals.

Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source centered at 75 Hz, with received levels between 118 and 137 dB re 1 µPa, were not found to overtly affect elephant seal dives (Costa et al. 2003). However, they did produce subtle effects that varied in direction and degree among the individual seals, again illustrating the equivocal nature of behavioral effects and consequent difficulty in defining and predicting them.

Harbor seals exposed to seal scarers (i.e., acoustic harassment devices) used to deter seals from fishing nets did not respond at levels of 109–134 dB re 1 µPa and demonstrated minor responses by occasionally hauling out at 128–138 dB re 1 µPa (Kastelein et al. 2015d). Pingers have also been used to deter marine mammals from fishing nets; in some cases, this has led to the "dinner bell effect," where the pinger becomes an attractant rather than a deterrent (Carretta and Barlow 2011). Steller sea lions were exposed to a variety of tonal, sweep, impulse, and broadband sounds. The broadband sounds did not cause a response, nor did the tones at levels below 165 dB re 1 µPa at 1 m, but the 8 kilohertz tone and 1–4 kilohertz sweep at source levels of 165 dB re 1 µPa caused the sea lions to haul out (Akamatsu et al. 1996).

Similar to the other taxonomic groups assessed, pinniped behavioral responses to sonar and other transducers seem to be mediated by the contextual factors of the exposure, including the proximity of the source, the characteristics of the signal, and the behavioral state of the animal. However, all pinniped behavioral response studies have been conducted in captivity, so while these results may be broadly applied to real-world exposure situations, it must be done with caution. Based on exposures to other sound sources in the wild (e.g., impulsive sounds and vessels), pinnipeds are not likely to respond strongly to Navy sonar that is not in close proximity to the animal or approaching the animal.

8.2.1.1.8 Impact Range to Effects from Sonar and Other Transducers

Section 2.2.1 presented information on the criteria and thresholds used to estimate impacts to marine mammals from sonar and other transducers. Additional information on these criteria is described in the technical report, *Criteria and Thresholds for U.S Navy Acoustic and Explosive Impact to Marine Mammals and Sea Turtles* (Navy 2017a). This section presents information on the range to effects for different sonar and other transducers to specific criteria determined using the NAEMO. Marine mammals within these ranges would be predicted to receive the associated effect. All information on range to effects, as presented below, is taken from the Navy's NWTT Phase III BA (Navy 2020e). We use the Navy's calculated range to effects as the best available information for analyzing effects of sonar and on marine mammals and sea turtles.

The estimated ranges to the PTS threshold for an exposure of 30 seconds are shown in Table 102 and are relative to the marine mammal's functional hearing group. The ranges provided in the

table include the average range to PTS, as well as the range from the minimum to the maximum distance at which PTS is possible for each hearing group. The 30 second exposure period was chosen based on examining the maximum amount of time a marine mammal would realistically be exposed to levels that could cause the onset of PTS based on platform (e.g., ship) speed and a nominal animal swim speed of approximately 1.5 meters per second. Estimated range to effects used for the NAEMO analysis are based on a number of factors including: 1) the assumed duty cycle for that bin (e.g., MF1 hull mounted sonar pings once every 50 seconds) relative to the durations over which SEL is accumulated in the range tables, 2) the source level of the bin, and 3) the susceptibility of the functional hearing group based on the source frequency.

Table 102. Range to permanent threshold shift for five representative sonar systems.

Hearing Group	*Approximate PTS (30 seconds) Ranges (meters)[1]*				
	Sonar bin HF4	*Sonar bin LF4*	*Sonar bin MF1*	*Sonar bin MF4*	*Sonar bin MF5*
Low-frequency cetaceans	0 (0–0)	2 (1–3)	67 (60–110)	15 (15–17)	0 (0–0)
Mid-frequency cetaceans	1 (0–3)	0 (0–0)	16 (16–19)	3 (3–3)	0 (0–0)
Otariids	0 (0–0)	0 (0–0)	6 (6–6)	0 (0–0)	0 (0–0)

[1] PTS ranges extend from the sonar or other transducer sound source to the indicated distance. The average range to PTS is provided as well as the range from the estimated minimum to the maximum range to PTS in parenthesis.
Notes: HF = high-frequency, LF = low-frequency, MF = mid-frequency, PTS = permanent threshold shift

The tables below illustrate the range to TTS for 1, 30, 60, and 120 seconds from five representative sonar systems (Table 103 through Table 107). Due to the lower acoustic thresholds for TTS versus PTS, ranges to TTS are longer. Therefore, successive pings can be expected to add together, further increasing the range to onset-TTS.

Table 103. Ranges to temporary threshold shift for sonar bin HF4 over a representative range of environments within the action area.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin HF4			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*
Low-frequency cetaceans	2 (0–3)	3 (1–6)	5 (3–8)	8 (5–12)
Mid-frequency cetaceans	12 (7–20)	21 (12–40)	29 (17–60)	43 (24–90)
Otariids	0 (0–0)	0 (0–0)	0 (0–0)	1 (0–1)

[1] Ranges to TTS represent the model predictions in different areas and seasons within the action area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin HF4			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*

Notes: HF = high frequency, TTS = temporary threshold shift

Table 104. Ranges to temporary threshold shift for sonar bin LF4 over a representative range of environments within the action area.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin LF4			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*
Low-frequency cetaceans	22 (19–30)	32 (25–230)	41 (30–230)	61 (45–100)
Mid-frequency cetaceans	0 (0–0)	0 (0–0)	0 (0–0)	0 (0–0)
Otariids	0 (0–0)	0 (0–0)	0 (0–0)	0 (0–0)

[1] Ranges to TTS represent the model predictions in different areas and seasons within the action area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.

Table 105. Ranges to temporary threshold shift for sonar bin MF1 over a representative range of environments within the action area.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin MF1			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*
Low-frequency cetaceans	1,054 (80–2,775)	1,054 (80–2,775)	1,480 (80–4,525)	1,888 (80–5,275)
Mid-frequency cetaceans	225 (80–380)	225 (80–380)	331 (80–525)	411 (80–700)
Otariids	67 (60–110)	67 (60–110)	111 (80–170)	143 (80–250)

[1] Ranges to TTS represent the model predictions in different areas and seasons within the action area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis. Ranges for 1-second and 30-second periods are identical for Bin MF1 because this system nominally pings every 50 seconds; therefore, these periods encompass only a single ping.

Notes: MF = mid-frequency, TTS = temporary threshold shift

Table 106. Ranges to temporary threshold shift for sonar bin MF4 over a representative range of environments within the action area.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin MF4			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*
Low-frequency cetaceans	87 (85–110)	176 (130–320)	265 (190–575)	477 (290–975)
Mid-frequency cetaceans	22 (22–25)	35 (35–45)	50 (45–55)	71 (70–85)
Otariids	8 (8–8)	15 (15–17)	19 (19–23)	25 (25–30)

[1] Ranges to TTS represent the model predictions in different areas and seasons within the action area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.
Notes: MF = mid-frequency, TTS = temporary threshold shift

Table 107. Ranges to temporary threshold shift for sonar bin MF5 over a representative range of environments within the action area.

Hearing Group	*Approximate TTS Ranges (meters)[1]*			
	Sonar Bin MF5			
	1 second	*30 seconds*	*60 seconds*	*120 seconds*
Low-frequency cetaceans	11 (10–13)	11 (10–13)	17 (16–19)	24 (23–25)
Mid-frequency cetaceans	6 (0–9)	6 (0–9)	12 (11–14)	18 (17–22)
Otariids	0 (0–0)	0 (0–0)	0 (0–0)	0 (0–0)

[1] Ranges to TTS represent the model predictions in different areas and seasons within the action area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.
Notes: MF = mid-frequency, TTS = temporary threshold shift

The range to received sound levels in 6 dB steps from five representative sonar bins and the percentage of animals that may exhibit a significant behavioral response under each behavioral response function are shown in Table 108 through Table 112, respectively.

Table 108. Ranges to a potentially significant behavioral response for sonar bin HF4 over a representative range of environments within the action area.

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin HF4*		
		Odontocete	*Mysticete*	*Pinniped*
196	4 (0–7)	100%	100%	100%
190	10 (0–16)	100%	98%	99%
184	20 (0–40)	99%	88%	98%
178	42 (0–85)	97%	59%	92%
172	87 (0–270)	91%	30%	76%
166	177 (0–650)	78%	20%	48%
160	338 (25–825)	58%	18%	27%
154	577 (55–1,275)	40%	17%	18%
148	846 (60–1,775)	29%	16%	16%
142	1,177 (60–2,275)	25%	13%	15%
136	1,508 (60–3,025)	23%	9%	15%
130	1,860 (60–3,525)	20%	5%	15%
124	2,202 (60–4,275)	17%	2%	14%
118	2,536 (60–4,775)	12%	1%	13%
112	2,850 (60–5,275)	6%	0%	9%
106	3,166 (60–6,025)	3%	0%	5%
100	3,470 (60–6,775)	1%	0%	2%

Notes: dB re 1 µPa = decibels referenced to 1 micropascal, HF = high-frequency

Table 109. Ranges to a potentially significant behavioral response for sonar bin LF4 over a representative range of environments within the action area.

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin LF4*		
		Odontocete	*Mysticete*	*Pinniped*
196	1 (0–1)	100%	100%	100%
190	3 (0–3)	100%	98%	99%
184	6 (0–8)	99%	88%	98%
178	13 (0–30)	97%	59%	92%
172	29 (0–230)	91%	30%	76%
166	64 (0–100)	78%	20%	48%
160	148 (0–310)	58%	18%	27%
154	366 (230–850)	40%	17%	18%
148	854 (300–2,025)	29%	16%	16%
142	1,774 (300–5,025)	25%	13%	15%
136	3,168 (300–8,525)	23%	9%	15%

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin LF4*		
		Odontocete	*Mysticete*	*Pinniped*
130	5,167 (300–30,525)	20%	5%	15%
124	7,554 (300–93,775)	17%	2%	14%
118	10,033 (300–100,000*)	12%	1%	13%
112	12,700 (300–100,000*)	6%	0%	9%
106	15,697 (300–100,000*)	3%	0%	5%
100	17,846 (300–100,000*)	1%	0%	2%

* Indicates maximum range to which acoustic model was run, a distance of approximately 100 km from the sound source.
Notes: Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms. dB re 1 µPa = decibels referenced to 1 micropascal, LF = low-frequency

Table 110. Ranges to a potentially significant behavioral response for sonar bin MF1 over a representative range of environments within the action area.

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin MF1*		
		Odontocete	*Mysticete*	*Pinniped*
196	112 (80–170)	100%	100%	100%
190	262 (80–410)	100%	98%	99%
184	547 (80–1,025)	99%	88%	98%
178	1,210 (80–3,775)	97%	59%	92%
172	2,508 (80–7,525)	91%	30%	76%
166	4,164 (80–16,025)	78%	20%	48%
160	6,583 (80–28,775)	58%	18%	27%
154	10,410 (80–47,025)	40%	17%	18%
148	16,507 (80–63,525)	29%	16%	16%
142	21,111 (80–94,025)	25%	13%	15%
136	26,182 (80–100,000*)	23%	9%	15%
130	31,842 (80–100,000*)	20%	5%	15%
124	34,195 (80–100,000*)	17%	2%	14%
118	36,557 (80–100,000*)	12%	1%	13%
112	38,166 (80–100,000*)	6%	0%	9%
106	39,571 (80–100,000*)	3%	0%	5%
100	41,303 (80–100,000*)	1%	0%	2%

* Indicates maximum range to which acoustic model was run, a distance of approximately 100 km from the sound source.
Notes: Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin MF1*		
		Odontocete	*Mysticete*	*Pinniped*

estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms. dB re 1 µPa = decibels referenced to 1 micropascal, MF = mid-frequency

Table 111. Ranges to a potentially significant behavioral response for sonar bin MF4 over a representative range of environments within the action area.

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin MF4*		
		Odontocete	*Mysticete*	*Pinniped*
196	8 (0–8)	100%	100%	100%
190	16 (0–20)	100%	98%	99%
184	34 (0–40)	99%	88%	98%
178	68 (0–85)	97%	59%	92%
172	155 (120–300)	91%	30%	76%
166	501 (290–975)	78%	20%	48%
160	1,061 (480–2,275)	58%	18%	27%
154	1,882 (525–4,025)	40%	17%	18%
148	2,885 (525–7,525)	29%	16%	16%
142	4,425 (525–14,275)	25%	13%	15%
136	9,902 (525–48,275)	23%	9%	15%
130	20,234 (525–56,025)	20%	5%	15%
124	23,684 (525–91,775)	17%	2%	14%
118	28,727 (525–100,000*)	12%	1%	13%
112	37,817 (525–100,000*)	6%	0%	9%
106	42,513 (525–100,000*)	3%	0%	5%
100	43,367 (525–100,000*)	1%	0%	2%

* Indicates maximum range to which acoustic model was run, a distance of approximately 100 km from the sound source.

Notes: Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms. dB re 1 µPa = decibels referenced to 1 micropascal, MF = mid-frequency

Table 112. Ranges to a potentially significant behavioral response for sonar bin MF5 over a representative range of environments within the action area.

Received Level (dB re 1 µPa)	*Mean Range (meters) with Minimum and Maximum Values in Parentheses*	*Probability of Behavioral Response for Sonar Bin MF5*		
		Odontocete	*Mysticete*	*Pinniped*
196	0 (0–0)	100%	100%	100%
190	1 (0–3)	100%	98%	99%
184	5 (0–7)	99%	88%	98%
178	14 (0–18)	97%	59%	92%
172	29 (0–35)	91%	30%	76%
166	58 (0–70)	78%	20%	48%
160	127 (0–280)	58%	18%	27%
154	375 (0–1,000)	40%	17%	18%
148	799 (490–1,775)	29%	16%	16%
142	1,677 (600–3,525)	25%	13%	15%
136	2,877 (675–7,275)	23%	9%	15%
130	4,512 (700–12,775)	20%	5%	15%
124	6,133 (700–19,275)	17%	2%	14%
118	7,880 (700–26,275)	12%	1%	13%
112	9,673 (700–33,525)	6%	0%	9%
106	12,095 (700–45,275)	3%	0%	5%
100	18,664 (700–48,775)	1%	0%	2%

Notes: Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms. dB re 1 µPa = decibels referenced to 1 micropascal, MF = mid-frequency

8.2.1.1.9 Exposure Analysis – Marine Mammal Exposure to Sonar and Other Transducers

In this section we discuss the estimated number of exposures of ESA-listed marine mammals to sonar and other transducers that are expected to rise to the level of take under the ESA, and the expected magnitude of effect from those exposures. The exposure estimates used for our effects analysis were produced by the Navy based on NAEMO output and post-processing techniques, and are based on the typical number of activities every seven years of the proposed action, and the maximum number of activities in a given year under the proposed action. NAEMO modeled-estimated exposures resulting in injury and mortality are further analyzed to account for mitigation proposed by the Navy to avoid or reduce impacts to cetaceans and for consideration of avoidance of multiple exposures that would be expected from individual animals once they sense the presence of Navy acoustic stressors. For details, see the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018d).

Blue Whale

Blue whales are seasonally present in the offshore area of the action area, from summer through winter, where they may be exposed to sounds from sonar and other transducers associated with training and testing activities. Blue whales are not expected to occur in the Inland Waters or Western Behm Canal portions of the action area. The Navy's quantitative analysis estimates 25 exposures resulting in behavioral harassment and 21 exposures resulting in TTS every seven years of the proposed action (Table 113). The Navy's analysis estimates five exposures resulting in behavioral harassment and four exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 113). The large majority of predicted impacts on blue whales from sonar and other transducers are from testing activities (i.e., 60 percent of behavioral harassment and all instances of TTS). All predicted impacts from testing activities would occur in the offshore portion of the action area (Figure 90).

Table 113. Estimated impacts on blue whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	11	0	0	14	21	0
One-year Maximum	2	0	0	3	4	0



Figure 90. Blue whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Fin Whale

Fin whales could be present year-round in the offshore area and occur in small-numbers in the Western Behm Canal, but are not expected to occur in the Inland Waters portion of the action area. Fin whales may be exposed to sounds from sonar and other transducers associated with training and testing activities throughout the year. The Navy's quantitative analysis estimates 491 exposures resulting in behavioral harassment and 237 exposures resulting in TTS every seven years of the proposed action (Table 114). The Navy's analysis estimates 86 exposures resulting in behavioral harassment and 43 exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 114). All aforementioned exposure estimates

apply to fin whales from the California, Oregon, and Washington and Northeast Pacific stocks. The majority of predicted impacts on fin whales from sonar and other transducers are from testing activities (i.e., nearly 53 percent of behavioral harassment and nearly 70 percent of TTS). Nearly all predicted impacts would occur in the offshore portion of the action area (Figure 91).

Table 114. Estimated impacts on fin whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	285	92	0	206	145	0
One-year Maximum	41	13	0	45	30	0



Figure 91. Fin whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Humpback Whale – Central America DPS and Mexico DPS

The ESA-listed Mexico (California, Oregon, and Washington stock) DPS and Central America (California, Oregon, and Washington stock) DPS of humpback whales both occur regularly in the action area. Specifically, they are present year-round in the offshore area and Western Behm Canal and seasonally in the Inland Waters portion of the action area, where they may be exposed to sounds from sonar and other transducers associated with training and testing activities that take place throughout the year. For the Central America DPS, the Navy's quantitative analysis estimates 19 exposures to result in behavioral harassment and TTS every seven years of the proposed action. The Navy's analysis estimates four exposures to result in behavioral harassment

and four exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 115). For the Mexico DPS, 191 exposures are estimated to result in behavioral harassment and 245 exposures are estimated to result in TTS every seven years of the proposed action. In addition, 35 exposures are estimated to result in behavioral harassment and 48 exposures in TTS per year of maximum activity as a result of the proposed action (Table 115). All predicted impacts on the Central America DPS from sonar and other transducers are from testing activities. The majority of predicted impacts on the humpback whale Mexico DPS from sonar and other transducers are from testing activities (i.e., nearly 92 percent of behavioral harassment and nearly 98 percent of TTS). Nearly all predicted impacts to both DPSs would occur in the offshore portion of the action area (Figure 92).

Table 115. Estimated impacts on humpback whales by DPS over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect							
DPS	*Timeframe*	*Training*			*Testing*		
		Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Central America	**Seven-year Period**	0	0	0	19	19	0
	One-year Maximum	0	0	0	4	4	0
Mexico	**Seven-year Period**	21	7	0	170	238	0
	One-year Maximum	3	1	0	32	47	0

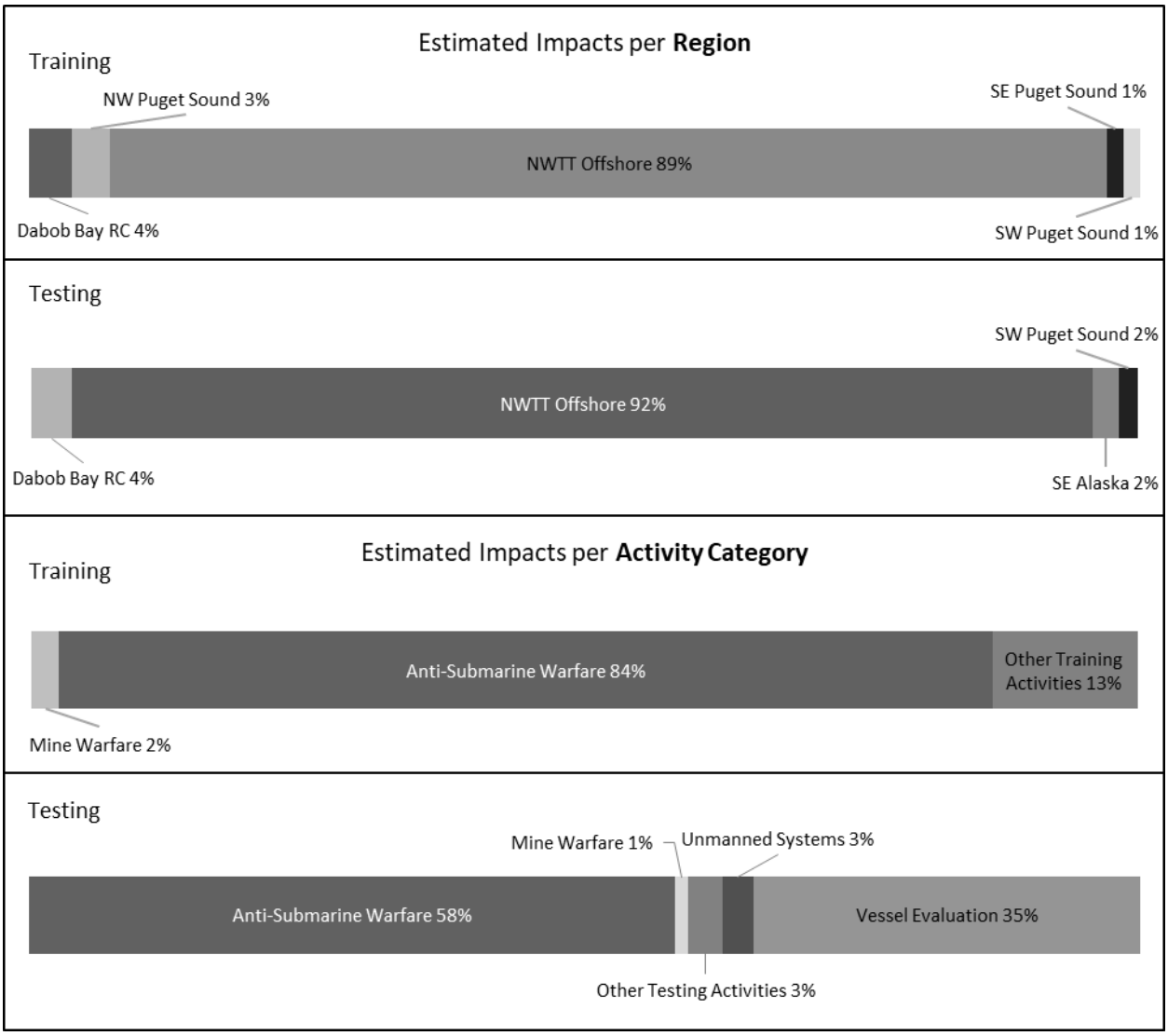


Figure 92. Humpback whale (Central America and Mexico DPSs) impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Southern Resident Killer Whale DPS

The Southern Resident killer whale DPS could be present year-round in the offshore area and inland portions of the action area, where they may be exposed to sounds from sonar and other transducers associated from NWTT activities throughout the year. The Navy's quantitative analysis accounts for group size and density (animals/area) when distributing animats (representative animal dosimeters) in the sound field for modeling acoustic impacts. In instances where the potential for a species to be present at any point in time is very low, as in the case of Southern Resident killer whales, the predicted impacts are weighted by the large majority of instances in which no impacts would occur during various activities. Additionally, because the behavioral response functions predict the likelihood of response to a given received level, the

predicted responses are lower at lower received levels. Both of these factors can result in predicted annual impacts that are less than the typical group size.

For this DPS, the Navy's quantitative analysis estimates 232 exposures to result in behavioral harassment and 11 exposures to result in TTS every seven years of the proposed action. The Navy's analysis estimates 49 exposures resulting in behavioral harassment and two exposures resulting in TTS per year of maximum sonar activity as a result of the proposed action (Table 116). Based on NAEMO results, no exposures resulting in PTS of Southern Resident killer whales were predicted (i.e., none of the predicted TTS exposures were reduced from PTS due to mitigation and avoidance factors). The Navy also provided the following unprocessed annual exposure estimates for Southern Resident killer whales: 2,221 > 103 dB; 1,291 > 121 dB; 118 > 163 dB; 16 > 181 dB; 1 > 205 dB (Navy 2020g). These represent the number of times per year individuals Southern Resident killer whales could be exposed to the acoustic environment that is a result of NWTT activities, regardless of whether they respond in a way that would significantly disrupt normal behavioral patterns as a result of that exposure (i.e., the total number of exposures prior to the application of a dose-response curve or criteria). We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

The large majority of predicted impacts on Southern Resident killer whales from sonar and other transducers are from testing activities (i.e., nearly 94 percent of behavioral harassment and all instances of TTS). Over 90 percent of predicted impacts would occur in the offshore portion of the action area (Figure 93). By activity, the largest number of estimated annual exposures resulting in behavioral harassment would result from Undersea Warfare Testing (19.6), Torpedo (non-explosive) Testing (9.08), Countermeasure Testing (5.36), and Anti-Submarine Warfare Testing (4.70) (Table 18). The largest number of estimated annual exposures resulting in TTS would result from Unmanned Underwater Vehicle Testing (0.57), Undersea Warfare Testing (0.41), Mine Detection and Classification Testing (0.40), and Countermeasure Testing (0.23).

Table 116. Estimated impacts on Southern Resident killer whales over a seven-year period and over a year of maximum activity as a result of the proposed NWTT activities using sonar and other transducers.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	15	0	0	217	11	0
One-year Maximum	3	0	0	46	2	0

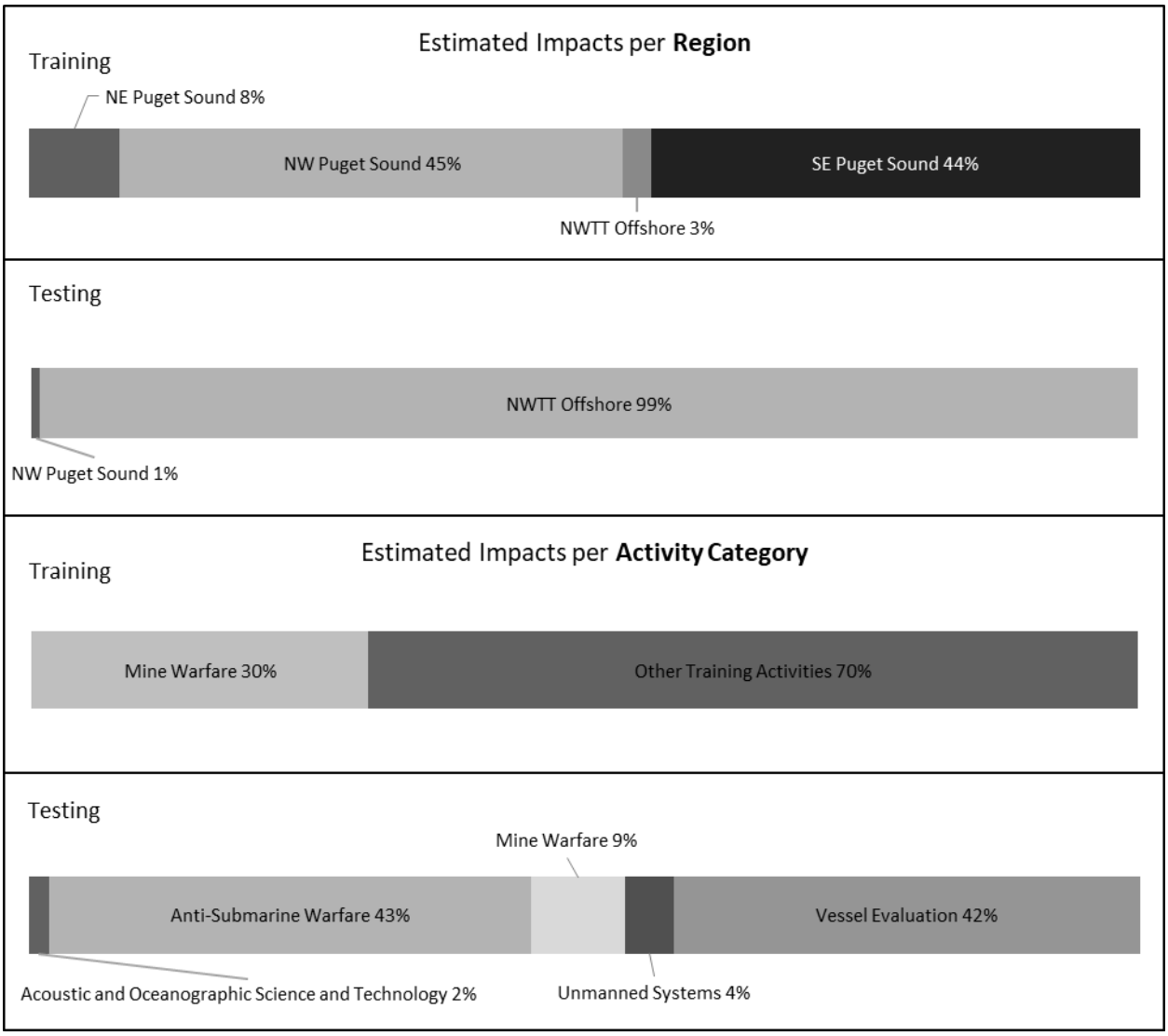


Figure 93. Southern Resident killer whale impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Table 117. Southern Resident killer whale DPS estimated maximum annual impacts by training activity (BEH = behavioral harassment).

Annual Take Estimate (Rounded to Nearest Integer)	TTS and BEH by Region	Activity Name, Location, Acoustic Sources (additional activity detail in NWTT EIS/OEIS Appendix A: Activity Descriptions)	TTS and BEH by Activity	Typical Duration	Events/Year
3 BEH	**Offshore** 0.08 BEH	Unmanned Underwater Vehicle Training. Quinault: LFS2; M3	0.08 BEH	Up to 24 hours Average 8 hours per event	2
	Inland Waters 2.45 BEH	Surface Ship Sonar Maintenance. Everett, Bremerton: MF1	0.01 BEH	Up to 4 hours	22 (3 hours of MF1 for Inland Waters)
		Civilian Port Defense—Homeland Security Anti-Terrorism/Force Protection Exercises. Indian Island, Everett, Kitsap (NBK) Bangor, Bremerton, Manchester, Port Angeles, Port of Seattle: HF4, SAS2	0.78 BEH	3-5 Days: 24 hours over several days; UUV use 24 hours over several days; 24 hours underway time of Surface combatant - MCM Avenger class, or LCS Independence Class; 24 hours of small boat activity; Small boat use 8 hour/day over several days in support of 4 hours of diving time per day	1 event every other year
		Unmanned Underwater Vehicle Training. Crescent Harbor, Dabob, Bangor, Bremerton, Keyport, Manchester, NAVY 3, NAVY 7: FLS2; M3	1.66 BEH	Up to 24 hours Average 8 hours per event	58

Table 118. Southern Resident killer whale DPS estimated maximum annual impacts by testing activity (BEH = behavioral harassment; TTS = temporary threshold shift).

Annual Take Estimate (Rounded to Nearest Integer)	TTS and BEH by Region	Activity Name, Location, Acoustic Sources (additional activity detail in NWTT EIS/OEIS Appendix A: Activity Descriptions)	TTS and BEH by Activity	Typical Duration	Events/Year
46 BEH 2 TTS	**Inland Waters** 0.42 BEH	Acoustic and Oceanographic Research. Dabob & Keyport: LF4; MF9	<0.01 BEH	Up to 14 days	3
		Cold Water Support. Carr Inlet, Dabob, & Keyport: HF6	<0.01 BEH	Assume 8 hours for all events, though they may continue for up to 40 hours, and infrequently some may operate intermittently for multiple consecutive months	4
		Pierside Sonar Testing. Bangor, Bremerton, Everett: MF1,2,3,9,10,12; HF3; ASW3	0.03 BEH	Up to 3 weeks total per ship, with each source run independently and not continuously during this time	88-99
		Mine Countermeasure and Neutralization Testing. Bremerton, Carr Inlet, Crescent Harbor, Dabob, Hood Canal, Everett, Keyport, Indian Island, NAVY 3: HF4	0.39 BEH	1–10 days, with intermittent use of countermeasure/ neutralization systems during this period	3
	Offshore 45.31 BEH 1.84 TTS	Acoustic and Oceanographic Research. Quinault: LF4; MF9	0.85 BEH	Up to 14 days	1
		Mine Countermeasure and Neutralization Testing. Offshore Area: HF4	1.04 BEH	1–10 days, with intermittent use of countermeasure/ neutralization systems during this period	3
		Mine Detection and Classification Testing. Quinault: LF4; HF4; BB1,2	0.40 TTS 2.20 BEH	Up to 24 days, with up to 12 hours of acoustic activity each day	1
		Unmanned Underwater Vehicle Testing. Quinault: HF5,9; VHF1; TORP1; FLS2; M3; SAS2	0.57 TTS 1.48 BEH	Typically 1–2 days, but endurance testing may last up to 35 days. Some propulsion systems (e.g., gliders) could operate	38-39

Annual Take Estimate (Rounded to Nearest Integer)	TTS and BEH by Region	Activity Name, Location, Acoustic Sources (additional activity detail in NWTT EIS/OEIS Appendix A: Activity Descriptions)	TTS and BEH by Activity	Typical Duration	Events/Year
				continuously for multiple months.	
		Anti-Submarine Warfare Testing. Offshore Area (may occur within 3NM from shore in QRS, would be >3NM from shore in the Offshore Area): MF1K,4,5,10,11,12; ASW1,2,3,5; TORP1	4.70 BEH	1-2 weeks, with 4-8 hours of active sonar use with intervals of non-activity in between	44
		Countermeasure Testing. Quinault: MF1; HF8; ASW3,4; TORP2	0.23 TTS 5.36 BEH	From 4 hours to 6 days depending on the countermeasure being tested Not all events will include the use of sonar and other transducers	14
		Torpedo (Non-Explosive) Testing. Offshore Area: LF4; MF1,3,4,5,6,9,10; HF1,5,6; ASW3,4; TORP1,2,3	0.15 TTS 9.08 BEH	Up to 2 weeks Typically, no more than 8 torpedoes are fired per day during daylight hours.	22
		At-Sea Sonar Testing. Offshore Area: MF3,4,5; HF1,5; ASW3; TORP1; M3	0.08 TTS 1.00 BEH	From 4 hours to 11 days Active sonar use is intermittent throughout the event	5
		Undersea Warfare Testing. Offshore Area: MF1,4,5,6,9; HF4; ASW3,4; TORP1,2	0.41 TTS 19.60 BEH	Up to 10 days Ships will not be conducting test constantly during the duration of the allotted time	1-12

Sei Whale

Sei whales could be present in the offshore portion of the action area, where they may be exposed to sounds from sonar and other transducers associated with training and testing activities

throughout the year. Sei whales are not expected to occur in the Inland Waters or Western Behm Canal portions of the action area. The Navy's quantitative analysis estimates 189 exposures resulting in behavioral harassment and 263 exposures resulting in TTS every seven years of the proposed action. The Navy's analysis estimates 32 exposures resulting in behavioral harassment and 49 exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 119). Overall, the majority of predicted impacts on sei whales from sonar and other transducers are from testing activities (i.e., 50 percent of behavioral harassment and nearly 72 percent of TTS). All predicted impacts would occur in the offshore portion of the action area (Figure 94).

Table 119. Estimated impacts on sei whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	111	95	0	78	168	0
One-year Maximum	16	14	0	16	35	0



Figure 94. Sei whales impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Sperm Whale

Sperm whales could be present in the offshore area of the action area, where they may be exposed to sounds from sonar and other transducers associated with training and testing activities throughout the year. Sperm whales are not expected to occur in the Inland Waters or Western Behm Canal portions of the action area. The Navy's quantitative analysis estimates 4,986 exposures resulting in behavioral harassment and 31 exposures resulting in TTS every seven years of the proposed action. The Navy's analysis estimates 834 exposures resulting in behavioral harassment and 5 exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 120). Most of the predicted behavioral impacts on sperm whales from sonar and other transducers are from training activities (i.e., nearly 62 percent) while most

of the impacts resulting in TTS are from testing activities (i.e., 60 percent). All predicted impacts would occur in the offshore portion of the action area (Figure 95).

Table 120. Estimated impacts on sperm whales over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	3,562	12	0	1,424	19	0
One-year Maximum	510	2	0	324	3	0



Figure 95. Sperm whales impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

Guadalupe Fur Seal

Guadalupe fur seals are present within the coastal margins of the offshore portion of the action area during the warm season (summer and early autumn), where they may be exposed to sounds from sonar and other transducers associated with training and testing activities. The Navy's quantitative analysis estimates 8,184 exposures resulting in behavioral harassment and 85 exposures resulting in TTS every seven years of the proposed action. The Navy's analysis estimates 1,482 exposures resulting in behavioral harassment and 13 exposures resulting in TTS per year of maximum activity as a result of the proposed action (Table 121). Most of the predicted impacts on Guadalupe fur seals from sonar and other transducers are from testing

activities (i.e., nearly 60 percent of behavioral harassment and nearly 77 percent of TTS). All predicted impacts would occur in the offshore portion of the action area (Figure 96).

Table 121. Estimated impacts on Guadalupe fur seals over a seven-year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using sonar and other transducers in the NWTT action area.

Estimated Impacts by Effect						
Timeframe	*Training*			*Testing*		
	Behavioral	*TTS*	*PTS*	*Behavioral*	*TTS*	*PTS*
Seven-year Period	4,226	21	0	3,958	64	0
One-year Maximum	605	3	0	877	10	0

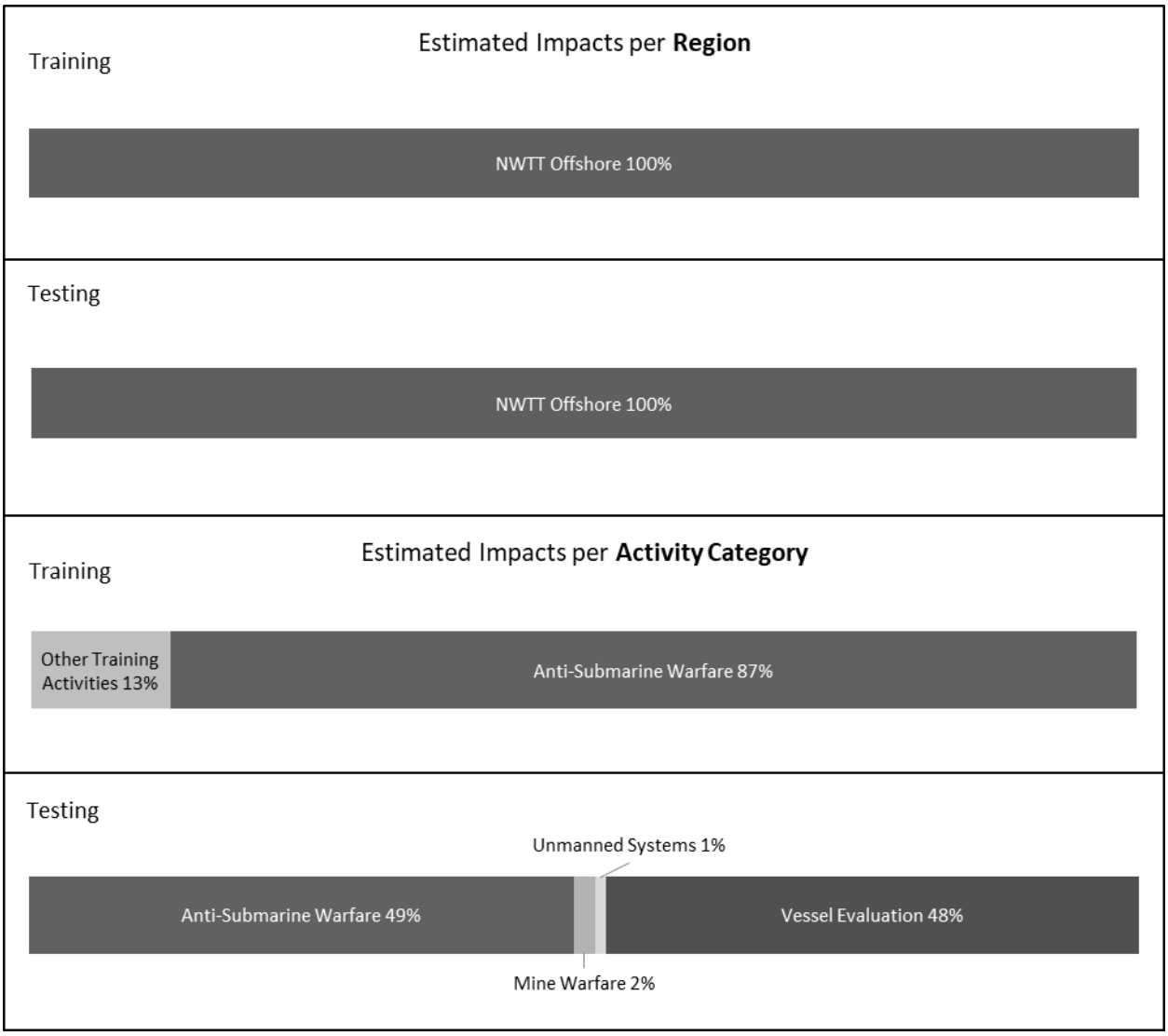


Figure 96. Guadalupe fur seal impacts estimated per year from sonar and other transducers used during training and testing under the proposed action by region and activity category.

8.2.1.1.10 Response Analysis

At the start of Section 8.2.1 we described the range of potential responses of ESA-listed marine mammals to sonar and other transducers associated with the proposed action. Given the above estimated exposures of ESA-listed marine mammals to sonar and other transducers associated with the proposed action, in this section we describe the likely responses of these species to this exposure. This includes behavioral responses and sound-induced hearing loss (i.e., TTS), as well as other possible responses (e.g., stress) that marine mammals may exhibit to exposure to sound fields from sonar and other transducers.

Our aim with this response analysis is to assess the potential responses that might reduce the fitness of individual ESA-listed marine mammals. In doing so, we consider and weigh evidence of adverse consequences, as well as evidence suggesting the absence of such consequences. In cases where data on the responses of the ESA-listed marine mammals to sonar and other transducers are not available, we rely on data from other closely-related species. In addition, we rely on information on the responses of ESA-listed marine mammals, as well as other related species, to anthropogenic sound sources other than military sonars (e.g., seismic air guns). We recognize that there can be species and sound-specific responses, and even within species, not all individual animals are likely to respond to all sounds in the same way. Nonetheless, by examining the range of responses that ESA-listed and other related marine mammals species exhibit to anthropogenic sounds, we incorporate uncertainty in our analysis that stems from intra- and inter-species response heterogeneity and make use of the best available science.

Hearing Threshold Shifts

Whether or not a hearing threshold shift will impact an individual animal's fitness depends on the duration, frequency, and magnitude of the shift. The frequencies affected by hearing loss will vary depending on the frequency of the fatiguing noise, with frequencies at and above the noise frequency most strongly affected. Recent literature on harbor porpoises showed that at higher sound pressure levels, hearing was most affected at frequencies half an octave above the center frequency of the fatiguing sound, while hearing was most affected at the center frequency of the fatiguing sound when the animals were exposed to lower sound pressure levels (Kastelein et al. 2020; Kastelein et al. 2019a; Kastelein et al. 2019c). As described previously, the Navy uses sonars operating at a wide range of frequencies (i.e., from low frequency sources to extremely high frequency sources), although most sources individually operate over a relatively narrow frequency band. Marine mammals that experience TTS from sonar sounds are likely to have reduced ability to detect biologically important sounds around the frequency band of the sonar until their hearing recovers. Some instances of hearing threshold shift are likely to occur at frequencies utilized by animals for acoustic cues. For example, during the period that a marine mammal has hearing loss, social calls from conspecifics could be more difficult to detect or interpret if they fell in the octave band of the sonar frequency. Killer whales are a primary predator of mysticetes and pinnipeds. Some hearing loss could make killer whale calls more difficult to detect until hearing recovers. Pinnipeds probably use sound and vibrations to find and capture prey underwater. It is unclear how or if mysticetes use sound for finding prey or feeding; therefore, it is unknown whether hearing loss would affect a mysticete's ability to locate prey or rate of feeding. Odontocetes use sound to find and capture prey underwater. Therefore, it could be more difficult for pinnipeds (e.g. Guadalupe fur seals) and odontocetes (e.g., sperm whales) to locate prey for a short period before their hearing recovers, assuming the TTS is within a frequency range used by these species for hunting.

The amount of hearing loss may range from slight to profound, depending on the ability of the individual to hear at the affected frequencies. Recovery from hearing loss begins almost

immediately after the noise exposure ceases and can take a few minutes to several days to fully recover, depending on the magnitude of the initial threshold shift. Instances of TTS resulting from Navy training and testing activities are expected to be minor to moderate (i.e., less than 20 dB of TTS directly after the exposure) and would recover within a matter of minutes to hours. Though there is uncertainty, this relatively short recovery time is supported by available information from the literature (e.g., Finneran 2015a). Exposures resulting in TTS are expected to be short term and of relatively low received level because of animal avoidance and the transient nature of most Navy sonar sources. Because TTS would likely be minor to moderate (less than 20 dB of TTS) and last for a short period of time (minutes to hours), costs would likely not be consequential to the animal long term. Behavioral research indicates that marine mammals most often will avoid sound sources at levels that would cause hearing loss, particularly more severe instances of TTS or PTS. Additionally, most Navy sonar sources are not stationary, minimizing the likelihood that an animal would remain in close proximity to the source for periods of time that could result in more severe instances of TTS (i.e., because marine mammals generally avoid loud sources of anthropogenic sound). Despite these factors that are expected to minimize the severity of TTS, we assume that some blue, fin, humpback, killer, sei, and sperm whales, and Guadalupe fur seals would experience TTS as the result of being exposed to sonar and other transducers from Navy training and testing activities (see Table 113 through Table 121 for estimates). As is the nature of TTS, such effects would be temporary and exposed individuals' hearing is expected to return to normal within minutes to days.

There is also the potential for repeat instances of TTS due to exposure to Navy sonar. In some exposure scenarios, it is possible that a particular animal will be exposed to sonar resulting in TTS and then, prior to being fully recovered, will be exposed again at a level resulting in TTS. Experimental studies have not explored such scenarios, so there is uncertainty as to how long recovery would take in these particular cases. Since we don't know what the condition of the animal's hearing is at the time of first exposure, it is possible that in some instances a minor TTS could exacerbate an already sensitive or vulnerable animal, thus increasing the risk of more severe effects. Given the relatively low usage of Navy sonar in the action area, and low predicted instances of TTS per individual in each ESA-listed population, the likelihood of repeat instances of TTS in a given day is very low.

Behavioral Responses

The Navy uses a behavioral response function to quantify the number of behavioral responses that could qualify as a significant behavioral disruption. Under the behavioral response function, a wide range of behavioral reactions may qualify as significant, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. While the risk functions were developed by identifying significant behavioral responses in the data sets (i.e., those that were moderate or high severity based on the Southall et al. (2007) severity scale), the estimates calculated using the behavioral response functions (as shown in

Table 113 through Table 121) do not differentiate between the different types of potential reactions, nor the significance of those potential reactions. These estimates also do not provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. Therefore, our analysis considers the available scientific evidence to determine the likely nature of modeled behavioral responses and potential fitness consequences for affected individuals.

The range of potential behavioral responses due to sonar exposure was presented earlier in this section. There are two general categories of information available regarding the likely responses of marine mammals to sonar exposure: 1) information from controlled exposure experiments, and 2) information from opportunistic observations during the operation of real world sonar. This research shows that marine mammals response to acoustic disturbance varies, depending on the characteristics of the sound source, the animal's experience with the sound source, and their behavioral state (e.g., migrating, breeding, feeding) at the time of the exposure.

As presented in a review by Southall et al. (2016), common responses to sonar during controlled exposure experiments include avoidance of the area of sonar exposure, cessation or modification of vocal behavior, and cessation of foraging. More minor reactions have also been observed including alerting to the sound source and startle responses. Southall et al. (2016) found that many, but not all responses of cetaceans to sonar observed so far have been relatively mild and/or brief. For example, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). In summarizing the response of blue whales to mid-frequency sonar, Goldbogen et al. (2013) states, "We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Additional controlled exposure experiments with mid-frequency sonar elicited behavioral responses from more than half of tagged blue whales feeding at depth, while no behavioral responses from blue whales feeding near the surface (Southall et al. 2019b). Friedlaender et al. (2016a) reported that deep-feeding blue whales responded more strongly to mid-frequency sonar than whales in other behavioral states. If individual ESA-listed cetaceans briefly respond to underwater sound from Navy training and testing (e.g., by slightly changing their behavior or temporarily relocating a short distance), the effects can be considered a behavioral response, but are unlikely to be significant to the animal unless that interruption is repeated many times. However, Southall et al. (2016) noted the short-term experiments designed to elicit behavioral responses from cetaceans due to sonar exposure were deliberately designed not to harm the affected animals.

Melcon et al. (2012) reported that baleen whales (i.e., blue whales) exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low frequency calls (D calls)

usually associated with feeding behavior. However, they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior and indicated that implications of the documented responses are unknown. Goldbogen et al. (2013) speculated that if the documented temporary behavioral responses interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since unconsumed prey would likely still be available in the environment in most cases following the cessation of acoustic exposure (i.e., sonar could cause scattering of prey, but would not be expected to injure or kill it). There would likely be an energetic cost associated with any temporary habitat displacement to find alternative locations for foraging, but unless disruptions occur over long durations or over subsequent days, we do not anticipate this movement to be consequential to the animal over the long term (Southall et al. 2007a).

While the Navy implements a series of mitigation measures to minimize high level sonar exposures during training and testing events, the responses of animals to real world Navy sonar could vary from the small scale, short-term controlled exposure experiments reviewed by Southall et al. (2016). Most of the studies reviewed by Southall et al. (2016) involved a single platform transmitting sonar or another sound source for a short period of time. The response of an animal to an initial exposure during such an event may be different than what could be expected if an animal is exposed multiple times or for a long period of time during an event. While some Navy activities involve the use of sonar from multiple platforms over days or weeks, the proposed NWTT action does not include any major training events and most activities only involve unit level training with a small number of assets involved. Additionally, while these studies can implement controls for some variables (e.g., the distance and movement of the source), they also introduce additional variables that do not normally occur in a real Navy training or testing activity, including the tagging of whales, intentionally following the tagged animals with multiple vessels, and continually approaching the animal to create a dose escalation.

Because of the limitations associated with controlled exposure experiments, it is also important to consider studies that opportunistically observed the response of cetaceans to real world Navy sonar. Passive acoustic monitoring and visual observational behavioral response studies have been conducted on Navy ranges, taking advantage of the existing seafloor hydrophones and real testing and training activity and associated sources to assess behavioral responses (Deakos and Richlen 2015; Henderson et al. 2019; Henderson et al. 2016; Manzano-Roth et al. 2016; Martin et al. 2015; Mccarthy et al. 2011; Mobley and Deakos 2015; Moretti et al. 2014; Tyack et al. 2011b). Collectively, these studies have indicated that responses vary, and include avoidance of the area of sonar exposure, cessation or modification of vocal behavior, changes in dive

behavior, and cessation of foraging. In addition, some aerial, visual, and acoustic monitoring is conducted before, during and after training events to ascertain whether behavioral responses occurred or could be observed during training and look for injured or stranded animals after training (Campbell et al. 2010; Farak et al. 2011; HDR 2011; Navy 2011b; Navy 2013a; Navy 2014b; Navy 2015; Norris et al. 2012; Smultea and Mobley 2009; Smultea et al. 2009; Trickey et al. 2015a). During all of these monitoring efforts, only a few behavioral responses have been observed, and no injured or dead animal was observed that was directly related to a training event (some dead animals were observed, but typically before the event, or appeared to have been deceased prior to the event; Smultea et al. 2011). However, it should be noted that passive acoustic studies are limited to observations of vocally-active cetaceans and visual studies are limited to what can be observed at the surface. These study types do have the benefit of occurring in the absence of some of the added contextual variables in the controlled exposure studies.

The limitations of opportunistic observations (e.g., limited to observations of vocally-active cetaceans or animals at the surface, limited ability to monitor animal activity long-term, limited ability to control other variable which could impact animal behavior [e.g., prey distribution]) result in some uncertainty as to the likely responses of ESA-listed cetaceans due to sonar exposure. Forney et al. (2017) noted that species that respond to noise (e.g., from military sonars) by avoiding an area are unlikely to be observed using traditional methods (e.g., lookouts or passive acoustic monitoring) because animals react at distances far greater than the detection range of these methods. They suggest that individuals that are observed must be considered relatively tolerant of anthropogenic noise.

In summary, the available information indicates a range of behavioral responses to sonar may occur for marine mammals, but most responses are expected to be brief, with the animal returning to baseline behavior shortly after the exposure is over. However, as noted by Forney et al. (2017), there is uncertainty due to the limitations of observing responses to sonar in the wild.

Masking (auditory interference)

The potential effects of masking were described earlier in this section. Some limited masking could occur due to the Navy's use of sonar and other transducers when animals are in close enough proximity. That is, if an animal is close enough to the source to experience TTS or a significant behavioral disruption, we anticipate some masking could occur. As stated previously, masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Masking from noise at the same frequency of communicative vocalizations may cause disruptions to communication, social interactions, and acoustically-mediated cooperative behaviors such as foraging or reproductive activities.

Because traditional military sonars typically have low duty cycles, the effects of such masking are expected to be limited. The typical duty cycle with most tactical anti-submarine warfare is about once per minute with most active sonar pulses lasting no more than a few seconds (Navy

2013b). This indicates biologically-relevant sounds for individuals in close proximity would only be masked intermittently for a short time.

Newer high duty cycle or continuous active sonars have more potential to mask vocalizations, particularly for sperm whales, but as explained above, these effects would only happen close to the source. These sonars transmit more frequently (greater than 80 percent duty cycle) than traditional sonars, but at a substantially lower source level. Similarly, high frequency acoustic sources such as pingers that operate at higher repetition, also operate at lower source levels. While the lower source levels of these systems limit the range of impact compared to more traditional systems, animals close to the sonar source could experience masking on a much longer time scale than those exposed to traditional sonars. This effect would only occur if the animals were to remain in close proximity to the source.

Non-auditory Physical or Physiological responses

The available research on the potential for sonar or other sources of anthropogenic noise to result in physiological responses (e.g., stress) is described earlier in this section. Relatively little information exists on the linkage between anthropogenic sound exposure and stress in marine mammals, and even less information exists on the ultimate consequences of sound-induced stress responses (either acute or chronic). Increased stress has been documented as a result of both acute (e.g., Romano et al. 2004) and chronic (e.g., Rolland et al. 2012) anthropogenic noise. As described previously, though there are unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, it is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

8.2.1.2 Explosives

As described in Section 5.1.2, explosives include, but are not limited to, bombs, missiles, rockets, naval gun shells, torpedoes, mines, demolition charges, and explosive sonobuoys. Explosive detonations involving the use of high-explosive munitions, including bombs, missiles, and naval gun shells, could occur in the air or near the water's surface. Explosive detonations associated with torpedoes and explosive sonobuoys would occur in the water column. Mines and demolition charges could be detonated in the water column or on the ocean bottom. The Navy would not conduct explosive training activities within 50 nautical miles from shore in the Marine Species Coastal Mitigation Area. A small number of explosive mine neutralization training activities would occur in existing mine warfare areas in Inland Waters. A new mine countermeasure and neutralization testing activity would occur closer to shore than other activities that involve the use of in-water explosives. This activity would typically occur greater than 3 nautical miles from shore in the Quinault Range Site (E4 would be used more than three nautical miles offshore and E7 would be used more than six nautical miles offshore), or greater than 12 nautical miles from shore elsewhere in the offshore area. Explosives would not be used in the Olympic Coast National Marine Sanctuary, Juan de Fuca Eddy Marine Species Mitigation

Area, and seasonally in the Stonewall and Heceta Bank Humpback Whale Mitigation Area and Point St. George Humpback Whale Mitigation Area.

Assessing whether an explosive detonation may disturb or injure a marine mammal involves understanding the characteristics of the explosive sources, the marine mammals that may be present near the sources, the physiological effects of a close explosive exposure, and the effects of impulsive sound on marine mammal hearing and behavior. Many other factors besides just the received level or pressure wave of an explosion such as the animal's physical condition and size; prior experience with the explosive sound; and proximity to the explosion may influence physiological effects and behavioral reactions.

The potential range of effects from explosions include death, physical injury or trauma, observable behavioral response, and stress response that may not be detectable. Injury can occur to organs or tissues of an animal. Permanent or temporary hearing loss may occur as well. Stress can help an animal cope with changing conditions, but too much stress can result in negative physiological effects. Behavioral responses range from brief distractions to avoidance of a sound source to prolonged flight. The sections below provide additional background on the potential effects of explosives on marine mammals. In our exposure and response analyses below, we use this information to discuss the likely effects of Navy NWTT explosive use on ESA-listed marine mammals.

8.2.1.2.1 Non-Auditory Injury

Explosive injury to marine mammals would consist of primary blast injury, which refers to those injuries that result from the compression of a body exposed to a blast wave and is usually observed as barotrauma of gas-containing structures (e.g., lung and gut) and structural damage to the auditory system (Corey et al. 1943; General 1991; Richmond et al. 1973). The near instantaneous high magnitude pressure change near an explosion can injure an animal where tissue material properties significantly differ from the surrounding environment, such as around air-filled cavities such as in the lungs or gastrointestinal tract. Large pressure changes at tissue-air interfaces in the lungs and gastrointestinal tract may cause tissue rupture, resulting in a range of injuries depending on degree of exposure. The lungs are typically the first site to show any damage, while the solid organs (e.g., liver, spleen, and kidney) are more resistant to blast injury (Ward and W. 1943). Recoverable injuries would include slight lung injury, such as capillary interstitial bleeding, and contusions to the gastrointestinal tract. More severe injuries, such as tissue lacerations, major hemorrhage, organ rupture, or air in the chest cavity (pneumothorax), would significantly reduce fitness and likely cause death in the wild. Rupture of the lung may also introduce air into the vascular system, producing air emboli that can cause a stroke or heart attack by restricting oxygen delivery to critical organs.

If an animal is exposed to an explosive blast underwater, the likelihood of injury depends on the charge size, the geometry of the exposure (distance to the charge, depth of the animal and the charge), and the size of the animal. In general, an animal would be less susceptible to injury near the water surface because the pressure wave reflected from the water surface would interfere

with the direct path pressure wave, reducing positive pressure exposure. Susceptibility would increase with depth, until normal lung collapse (due to increasing hydrostatic pressure) and increasing ambient pressures again reduce susceptibility. See Appendix D (Acoustic and Explosive Concepts) in the NWTT Final EIS/OEIS (Navy 2020f) for an overview of explosive propagation and an explanation of explosive effects on gas cavities.

The only known occurrence of mortality or injury to a marine mammal due to a Navy training or testing event involving explosives occurred in March 2011 in nearshore waters off San Diego, California, at the Silver Strand Training Complex. This area had been used for underwater demolitions training for at least three decades without prior known incident. On this occasion, however, a group of approximately 100-150 long-beaked common dolphins entered the mitigation zone surrounding an area where a time-delayed firing device had been initiated on an explosive with a NEW of 8.76 pounds (3.97 kilograms) placed at a depth of 48 feet (14.6 meters). Approximately one minute after detonation, three animals were observed dead at the surface. The Navy recovered those animals and transferred them to the local stranding network for necropsy. A fourth animal was discovered stranded and dead 42 nautical miles to the north of the detonation three days later. It is unknown exactly how close those four animals were to the detonation. Upon necropsy, all four animals were found to have sustained typical mammalian primary blast injuries (Danil and St. Leger 2011). Since that incident, the Navy has implemented additional mitigation measures to minimize the risk of such an event occurring again.

Relatively little is known about auditory system trauma in marine mammals resulting from explosive exposure, although it is assumed that auditory structures would be vulnerable to blast injuries. Auditory trauma was found in two humpback whales that died following the detonation of a 5,000 kilogram explosive used off Newfoundland during demolition of an offshore oil rig platform (Ketten et al. 1993), but the proximity of the whales to the detonation was unknown. Eardrum rupture was examined in submerged terrestrial mammals exposed to underwater explosions (Richmond et al. 1973; Yelverton et al. 1973). However, results may not be applicable to the anatomical adaptations for underwater hearing in marine mammals. In this discussion, primary blast injury to auditory tissues is considered gross structural tissue damage distinct from threshold shift or other auditory effects.

Controlled tests with a variety of lab animals (mice, rats, dogs, pigs, sheep and other species) are the best data sources on actual injury to mammals due to underwater exposure to explosions. In the early 1970s, the Lovelace Foundation for Medical Education and Research conducted a series of tests in an artificial pond at Kirtland Air Force Base, New Mexico to determine the effects of underwater explosions on mammals, with the goal of determining safe ranges for human divers. The resulting data were summarized in two reports (Richmond et al. 1973; Yelverton et al. 1973). Specific physiological observations for each test animal are documented in Richmond et al. (1973). Gas-containing internal organs, such as lungs and intestines, were the principle damage sites in submerged terrestrial mammals; this is consistent with earlier studies of mammal exposures to underwater explosions in which lungs were consistently the first areas to show damage, with less consistent damage observed in the gastrointestinal tract (Corey et al. 1943;

Ward and W. 1943). Results from all of these tests suggest two explosive metrics are predictive of explosive injury: peak pressure and impulse.

In the Lovelace studies, acoustic impulse was found to be the metric most related to degree of injury, and size of an animal's gas-containing cavities was thought to play a role in blast injury susceptibility. The lungs of most cetaceans are similar in proportion to overall body size as those of terrestrial mammals, so the magnitude of lung damage in the tests may approximate the magnitude of injury to cetaceans when scaled for body size. Within the marine mammals, mysticetes and deeper divers (e.g., Kogiidae, Physeteridae, Ziphiidae) tend to have lung to body size ratios that are smaller and more similar to terrestrial animal ratios than the shallow diving odontocetes (e.g., Phocoenidae, Delphinidae) and pinnipeds (Fahlman et al. 2014a; Piscitelli et al. 2010). The use of test data with smaller lung to body ratios to set injury thresholds may result in a more conservative estimate of potential for damaging effects (i.e., lower thresholds) for animals with larger lung to body ratios.

For these shallow exposures of small terrestrial mammals (masses ranging from 3.4 to 50 kilograms) to underwater detonations, Richmond et al. (1973) reported that no blast injuries were observed when exposures were less than six pounds per square inch per millisecond (psi-ms) (40 Pa-s), no instances of slight lung hemorrhage occurred below 20 psi-ms (140 Pa-s), and instances of no lung damage were observed in some exposures at higher levels up to 40 psi-ms (280 Pa-s). An impulse of 34 psi-ms (230 Pa -s) resulted in about 50 percent incidence of slight lung hemorrhage. About half of the animals had gastrointestinal tract contusions (with slight ulceration, i.e., some perforation of the mucosal layer) at exposures of 25-27 psi-ms (170-190 Pa-s). Lung injuries were found to be slightly more prevalent than gastrointestinal tract injuries for the same exposure.

The Lovelace subject animals were exposed near the water surface; therefore, depth effects were not discernible in this data set. In addition, this data set included only small terrestrial animals, whereas marine mammals may be several orders of magnitude larger and have respiratory structures adapted for the high pressures experienced at depth. Goertner (1982) examined how lung cavity size would affect susceptibility to blast injury by considering both cetacean size and depth in a bubble oscillation model of the lung. Animal depth relates to injury susceptibility in two ways: injury is related to the relative increase in explosive pressure over hydrostatic pressure, and lung collapse with depth reduces the potential for air cavity oscillatory damage. The period over which an impulse must be delivered to cause damage is assumed to be related to the natural oscillation period of an animal's lung, which depends on lung size.

Because gas-containing organs are more vulnerable to primary blast injury, adaptations for diving that allow for collapse of lung tissues with depth may make animals less vulnerable to lung injury with depth. Adaptations for diving include a flexible thoracic cavity, distensible veins that can fill space as air compresses, elastic lung tissue, and resilient tracheas with interlocking cartilaginous rings that provide strength and flexibility (Ridgway 1972). Older literature suggested complete lung collapse depths at approximately 70 meters for dolphins (Ridgway and

Howard 1979) and 20–50 meters for phocid seals (Falke et al. 1985; Kooyman et al. 1972). Follow-on work by Kooyman and Sinnett (1982), in which pulmonary shunting was studied in harbor seals and sea lions, suggested that complete lung collapse for these species would be about 170 meters and about 180 m, respectively. More recently, evidence in sea lions suggests that complete collapse might not occur until depths as great as 225 m; although the depth of collapse and depth of the dive are related, sea lions can affect the depth of lung collapse by varying the amount of air inhaled on a dive (Mcdonald and Ponganis 2012). This is an important consideration for all divers who can modulate lung volume and gas exchange prior to diving via the degree of inhalation and during diving via exhalation (Fahlman et al. 2009). Indeed, there are noted differences in pre-dive respiratory behavior with some cetaceans exhibiting pre-dive exhalation to reduce the lung volume [e.g., phocid seals (Kooyman et al. 1973)].

High instantaneous peak pressures can cause damaging tissue distortion. Goertner (1982) suggested a peak overpressure gastrointestinal tract injury criterion because the size of gas bubbles in the gastrointestinal tract are variable, and their oscillation period could be short relative to primary blast wave exposure duration. The potential for gastrointestinal tract injury, therefore, may not be adequately modeled by the single oscillation bubble methodology used to estimate lung injury due to impulse. Like impulse, however, high instantaneous pressures may damage many parts of the body, but damage to the gastrointestinal tract is used as an indicator of any peak pressure-induced injury due to its vulnerability.

Older military reports documenting exposure of human divers to blast exposure generally describe peak pressure exposures around 100 psi (237 dB re 1 µPa peak) to feel like slight pressure or stinging sensation on skin, with no enduring effects (Christian and Gaspin 1974). Around 200 psi, the shock wave felt like a blow to the head and chest. Data from the Lovelace Foundation experiments show instances of gastrointestinal tract contusions after exposures up to 1,147 psi peak pressure, while exposures of up to 588 psi peak pressure resulted in many instances of no observed gastrointestinal tract effects. The lowest exposure for which slight contusions to the gastrointestinal tract were reported was 237 dB re 1 µPa peak. As a vulnerable gas-containing organ, the gastrointestinal tract is vulnerable to both high peak pressure and high impulse, which may vary to differing extents due to blast exposure conditions (i.e., animal depth, distance from the charge). This likely explains the range of effects seen at similar peak pressure exposure levels and shows the utility of considering both peak pressure and impulse when analyzing the potential for injury due to explosives.

8.2.1.2.2 Hearing Loss and Auditory Injury

Exposure to intense sound may result in noise-induced hearing loss that persists after cessation of the noise exposure. Hearing loss may be temporary or permanent, depending on factors such as the exposure frequency, received SPL, temporal pattern, and duration. The frequencies affected by hearing loss may vary depending on the exposure frequency, with frequencies at and above

the exposure frequency most strongly affected. The amount of hearing loss may range from slight to profound, depending on the ability of the individual to hear at the affected frequencies.

Hearing loss has only been studied in a few species of marine mammals, although hearing studies with terrestrial mammals are also informative. There are no direct measurements of hearing loss in marine mammals due to exposure to explosive sources. The sound resulting from an explosive detonation is considered an impulsive sound and shares important qualities (i.e., short duration and fast rise time) with other impulsive sounds such as those produced by air guns. General research findings regarding TTS and PTS in marine mammals are discussed in Section 8.2.1.1.

8.2.1.2.3 Physiological Stress

Marine mammals naturally experience stress within their environment and as part of their life histories. The stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor. However, if the magnitude and duration of the stress response is too great or too long, then it can have negative consequences to the organism (e.g., decreased immune function, decreased reproduction).

There are no direct measurements of physiological stress in marine mammals due to exposure to explosive sources. General research findings regarding physiological stress in marine mammals due to exposure to sound and other stressors are discussed in detail in Section 8.2.1.1. Because there are many unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, it is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

8.2.1.2.4 Masking

Masking can also result from exposure to sound from Navy explosives. There are no direct observations of masking in marine mammals due to exposure to explosive sources. General research findings regarding masking in marine mammals due to exposure to sound and other stressors are discussed in Section 8.2.1.1. Due to the short duration of sound from explosives, the potential for explosives to result in masking that would be biologically significant is limited.

8.2.1.2.5 Behavioral Reactions

Impulsive signals such as explosives, particularly at close range, have a rapid rise time and higher instantaneous peak pressure than other signal types, making them more likely to cause startle responses or avoidance responses. In fact, any stimuli in the environment can cause a behavioral response in marine mammals, including noise from explosions. There are few direct observations of behavioral reactions from cetaceans due to exposure to explosive sounds. Lammers et al. (2017) recorded dolphin detections near naval mine neutralization exercises and found that although the immediate response (within 30 seconds of the explosion) was an increase in whistles relative to the 30 seconds before the explosion, there was a reduction in daytime acoustic activity during the day of and the day after the exercise within 6 km. However, the

nighttime activity did not seem to be different than that prior to the exercise, and two days after there appeared to be an increase in daytime acoustic activity, indicating a rapid return to the area by the dolphins (Lammers et al. 2017). Vallejo et al. (2017) report on boat-based line-transect surveys which were run over ten years in an area where an offshore wind farm was built; these surveys included the periods of preconstruction, construction, and post-construction. Harbor porpoise were observed throughout the area during all three phases, but were not detected within the footprint of the windfarm during the construction phase, and were overall less frequent throughout the action area. However, they returned after the construction was completed at a slightly higher level than in the preconstruction phase. Furthermore, there was no large-scale displacement of harbor porpoises during construction, and in fact their avoidance behavior only occurred out to about 18 km, in contrast to the approximately 25 km avoidance distance found in other windfarm construction and pile driving monitoring efforts.

At long distances the rise time increases as the signal duration lengthens (similar to a "ringing" sound), making the impulsive signal more similar to a non-impulsive signal. Data on behavioral responses to impulsive sound sources are limited across all cetacean groups, with only a few studies available for mysticetes and odontocetes. Most data have come from seismic surveys that occur over long durations (e.g., on the order of days to weeks), and typically utilize large multi-air gun arrays that fire repeatedly. While seismic data provide the best available science for assessing behavioral responses to impulsive sounds by cetaceans, it is likely that these responses represent a worst-case scenario as compared to responses to Navy impulsive sources such as explosives. Navy explosive activities typically consist of a single or multiple explosions occurring over a short period of time in a relatively small area whereas seismic surveys input impulsive sound from airguns into the water column over a long period of time and over a large area (e.g., following a transect).

For their quantitative effects analysis, the Navy assumes that significant behavioral responses to solitary explosions are not anticipated due to the short duration of acoustic exposure from such explosions, but this does not preclude the potential for responses within the range to TTS. There has been very little research conducted on this topic. Depending on numerous factors (e.g., proximity, attentional focus, charge weight of blast, and experience of the animal) the responses of individuals may vary and we would assume some animals would exhibit more of a reaction than others. The mitigation measures that would be implemented (such as exclusion zones) are expected to reduce the potential for significant behavioral responses to occur from exposure to solitary explosions.

8.2.1.2.6 Mysticetes

Baleen whales have shown a variety of responses to impulsive sound sources, including avoidance, attraction to the source, reduced surface intervals, altered swimming behavior, and changes in vocalization rates (Gordon et al. 2003; McCauley et al. 2000b; Richardson et al. 1985b; Southall et al. 2007a). Studies have been conducted on many baleen whale species,

including gray, humpback, blue, fin, and bowhead whales. For the purposes of this analysis, due to the limited amount of data available, it is assumed that these responses are representative of all baleen whale species. As was discussed for responses to sonar, the behavioral state of the whale seems to be an integral part of whether or not the animal responds and how they respond to impulsive sources, as does the location and movement of the sound source, more than the received level of the sound.

Migratory behavior seems to lead to a higher likelihood of response, with some species demonstrating more sensitivity than others do. For example, migrating gray whales showed avoidance responses to seismic vessels at received levels between 164 and 190 dB re 1 µPa (Malme et al. 1986; Malme et al. 1988). Similarly, migrating humpback whales showed avoidance behavior at ranges of 5 to 8 km from a seismic array during observational studies and controlled exposure experiments in one Australian study (McCauley et al. 1998) and up to 3 km from a source vessel moving directly across their migratory path (Dunlop et al. 2017), and in another Australian study decreased their dive times and reduced their swimming speeds (Dunlop et al. 2015). When comparing received levels and behavioral responses when using ramp-up versus a constant noise level of airguns, humpback whales did not change their dive behavior but did deviate from their predicted heading and decreased their swim speeds (Dunlop 2016). In addition, the whales demonstrated more course deviation during the constant source trials but reduced travel speeds more in the ramp-up trials. In either case there was no dose-response relationship with the received level of the air gun noise, and similar responses were observed in control trials with vessel movement but no airguns so some of the response was likely due to the presence of the vessel and not the received level of the airguns.

When looking at the relationships between proximity, received level, and behavioral response, Dunlop et al. (2017) used responses to two different air guns and found responses occurred more towards the smaller, closer source than to the larger source at the same received level, demonstrating the importance of proximity. Responses were found to be more likely when the source was within 3 km or above 140 dB re 1 µPa, although responses were variable and some animals did not respond at those values while others responded below them. In addition, responses were generally small, with course deviations of only around 500 m, and short term (Dunlop et al. 2017). McDonald et al. (1995b) tracked a blue whale with seafloor seismometers and reported that it stopped vocalizing and changed its travel direction at a range of 10 km from the seismic vessel (estimated received level 143 dB re 1 µPa peak-to-peak). Bowhead whales seem to be the most sensitive species, perhaps due to a higher overlap between bowhead whale distribution and seismic surveys in Arctic and sub-Arctic waters, as well as a recent history of being hunted. While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al. 1995b), some whales avoided vessels by more than 20 km at received levels as low as 120 dB re 1 µPa. Additionally, Malme et al. (1988) observed clear changes in diving and breathing patterns in bowheads at ranges up to 73 km from seismic vessels, with received levels as low as 125 dB re 1 µPa. Bowhead whales may also avoid the

area around seismic surveys, from 6 to 8 km (Gordon et al. 2003) out to 20 or 30 km (Richardson et al. 1999). However, work by Robertson (2014) supports the idea that behavioral responses are contextually dependent, and that during seismic operations bowhead whales may be less "available" for counting due to alterations in dive behavior but that they may not have left the area after all.

In contrast, noise from seismic surveys was not found to impact feeding behavior or exhalation rates in western gray whales while resting or diving off the coast of Russia (Gailey et al. 2007; Yazvenko et al. 2007). However, the increase in vessel traffic associated with the surveys and the proximity of the vessels to the whales did affect the orientation of the whales relative to the vessels and shortened their dive-surface intervals (Gailey et al. 2016). Todd et al. (1996) found no clear short-term behavioral responses by foraging humpbacks to explosions associated with construction operations in Newfoundland, but did see a trend of increased rates of net entanglement closer to the noise source, possibly indicating a reduction in net detection associated with the noise through masking or TTS. Distributions of fin and minke whales were modeled with a suite of environmental variables along with the occurrence or absence of seismic surveys, and no evidence of a decrease in sighting rates relative to seismic activity was found for either species (Vilela et al. 2016). Their distributions were driven entirely by environmental variables, particularly those linked to prey including warmer sea surface temperatures, higher chlorophyll-a values, and higher photosynthetically available radiation (a measure of primary productivity).

Vocal responses to seismic surveys have been observed in a number of baleen whale species, including a cessation of calling, a shift in frequency, increases in amplitude or call rate, or a combination of these strategies. Blue whale feeding/social calls were found to increase when seismic exploration was underway, with seismic pulses at average received SELs of 131 dB re 1 μPa^2s (Di Lorio and Clark 2010), a potentially compensatory response to increased noise level. Responses by fin whales to a 10-day seismic survey in the Mediterranean Sea included possible decreased 20-Hz call production and movement of animals from the area based on lower received levels and changes in bearings (Castellote et al. 2012). However, similarly distant seismic surveys elicited no apparent vocal response from fin whales in the mid-Atlantic Ocean; instead, Nieukirk et al. (2012) hypothesized that 20-Hz calls may have been masked from the receiver by distant seismic noise. Models of humpback whale song off Angola showed significant seasonal and diel variation, but also showed a decrease in the number of singers with increasing received levels of air gun pulses (Cerchio et al. 2014). Bowhead whale calling rates decreased significantly at sites near seismic surveys (41 to 45 km) where received levels were between 116-129 dB re 1 µPa, and did not decrease at sites further from the seismic surveys (greater than 104 km) where received levels were 99-108 dB re 1 µPa (Blackwell et al. 2013). In fact, bowhead whale calling rates increased at the lower received levels, began decreasing at around 127 dB re 1 μPa^2s cumulative SEL, and ceased altogether at received levels over 170 dB re 1 μPa^2s cumulative SEL (Blackwell et al. 2015).

Mysticetes seem to be the most sensitive taxonomic group of cetaceans to impulsive sound sources, with possible avoidance responses occurring out to 30 km and vocal changes occurring in response to sounds over 100 km away. However, responses appear to be behaviorally mediated, with most avoidance responses occurring during migration behavior and little observed response during feeding behavior. These response patterns are likely to hold true for Navy impulsive sources. However, Navy impulsive sources would largely be stationary and short term (i.e., instantaneous for explosives) as compared to sources in these studies, and so responses would likely occur in closer proximity or not at all.

8.2.1.2.7 Odontocetes

Few data are available on odontocete responses to impulsive sound sources, with only a few studies on responses to seismic surveys, pile driving and construction activity available. Based on the limited available information, odontocetes appear to be less sensitive to impulsive sound than mysticetes, with responses occurring at much closer distances. This may be due to the predominance of low-frequency sound associated with these sources that propagates long distances and overlaps with the range of best hearing for mysticetes but is below that range for odontocetes. The exception to this is the harbor porpoise, which has been shown to be highly sensitive to most sound sources, avoiding both stationary (e.g., pile driving) and moving (e.g., seismic survey vessels) impulsive sound sources out to approximately 20 km (e.g., Haelters et al. 2014; Pirotta et al. 2014). However, even this response is short-term, with porpoises returning to the area within hours after the cessation of the noise.

Madsen et al. (2006) and Miller et al. (2009) tagged and monitored eight sperm whales in the Gulf of Mexico exposed to seismic air gun surveys. Sound sources were from approximately 2 to 7 nautical miles away from the whales, and received levels were as high as 162 dB SPL re 1 µPa (Madsen et al. 2006). The whales showed no horizontal avoidance, however one whale rested at the water's surface for an extended period of time until air guns ceased firing (Miller et al. 2009). While the remaining whales continued to execute foraging dives throughout exposure, tag data suggested there may have been subtle effects of noise on foraging behavior (Miller et al. 2009). Similarly, Weir (2008) observed that seismic air gun surveys along the Angolan coast did not significantly reduce the encounter rate of sperm whales during the 10-month survey period, nor were avoidance behaviors to air gun impulsive sounds observed. In contrast, Atlantic spotted dolphins did show a significant, short-term avoidance response to air gun impulses within approximately 1 km of the source (Weir 2008). The dolphins were observed at greater distances from the vessel when the air gun was in use, and when the air gun was not in use they readily approached the vessel to bow ride.

Captive bottlenose dolphins sometimes vocalized or were reluctant to return to the test station after exposure to single impulses from a seismic water gun (Finneran et al. 2002). When exposed to multiple impulses from a seismic air gun, some dolphins turned their heads away from the sound source just before the impulse, showing that they could anticipate the timing of the

impulses and perhaps reduce the received level (Finneran 2015a). During construction (including the blasting of old bastions) of a bridge over a waterway commonly used by the Tampa Bay, Florida stock of bottlenose dolphins, the use of the area by females decreased while males displayed high site fidelity and continued using the area, perhaps indicating differential habitat uses between the sexes (Weaver 2015).

A study was conducted on the response of harbor porpoises to a seismic survey using aerial surveys and C-PODs (an autonomous recording device that counts odontocete clicks); the animals appeared to have left the area of the survey, and decreased their foraging activity within 5 to 10 km, as evidenced by both a decrease in vocalizations near the survey and an increase in vocalizations at a distance (Pirotta et al. 2014; Thompson et al. 2013). However, the animals returned within a day after the air gun operation ceased, and the decrease in occurrence over the survey period was small relative to the observed natural seasonal decrease compared to the previous year. A number of studies (Brandt et al. 2011; Dähne et al. 2014; Haelters et al. 2014; Thompson et al. 2010; Tougaard et al. 2005; Tougaard et al. 2009) also found strong avoidance responses by harbor porpoises out to 20 km during pile driving; however, all studies found that the animals returned to the area after the cessation of pile driving. Kastelein et al. (2013b) exposed a captive harbor porpoise to impact pile driving sounds, and found that above 136 dB re 1 µPa (zero-to-peak) the animal's respiration rates increased, and at higher levels it jumped more frequently. Bergstrom et al. (2014) found that although there was a high likelihood of acoustic disturbance during wind farm construction (including pile driving), the impact was short-term. Graham et al. (2017) assessed the occurrence of bottlenose dolphins and harbor porpoises over different area and time scales with and without impact and vibratory pile driving. While there were fewer hours with bottlenose dolphin detections and reduced detection durations within the pile driving area and increased detection durations outside the area, the effects sizes were small, and the reduced harbor porpoise encounter duration was attributed to seasonal changes outside the influence of the pile driving. However, received levels in this area were lower due to propagation effects than in the other areas described above, which may have led to the lack of or reduced response.

Odontocete behavioral responses to impulsive sound sources are likely species- and context-dependent, with most species demonstrating little to no apparent response. Responses might be expected within close proximity to a noise source, under specific behavioral conditions such as females with offspring, or for sensitive species such as harbor porpoises.

8.2.1.2.8 Pinnipeds

A review of behavioral reactions by pinnipeds to impulsive noise can be found in Richardson et al. (1995b) and Southall et al. (2007a). Blackwell et al. (2004) observed that ringed seals exhibited little or no reaction to pipe-driving noise with mean underwater levels of 157 dB re 1 µPa and in-air levels of 112 dB re 20 µPa, suggesting that the seals had habituated to the noise. In contrast, captive California sea lions avoided sounds from an underwater impulsive source at

levels of 165–170 dB re 1 µPa (Finneran et al. 2003). Harbor and grey seals were also observed to avoid a seismic air gun by rapidly swimming away, and ceased foraging during exposure, but returned to normal behavior afterwards (Gordon et al. 2003). In another study, few responses were observed by New Zealand fur seals to a towed air gun array operating at full power; rather, when responses were observed it seemed to be to the physical presence of the vessel and tow apparatus, and these only occurred when the vessel was within 200 meters and sometimes as close as five meters (Lalas and McConnell 2016). Captive Steller sea lions were exposed to a variety of tonal, sweep, impulsive and broadband sounds to determine what might work as a deterrent from fishing nets. The impulsive sound had a source level of 120 dB re 1 µPa at 1 m, and caused the animals to haul out and refuse to eat fish presented in a net (Akamatsu et al. 1996). Steller sea lions exposed to in-air explosive blasts increased their activity levels and often re-entered the water when hauled out (Demarchi et al. 2012). However, these responses were short-lived and within minutes, the animals had hauled out again, and there were no lasting behavioral impacts in the days following the blasts.

Experimentally, Götz & Janik (2011) tested underwater startle responses to a startling sound (sound with a rapid rise time and a 93 dB sensation level [the level above the animal's hearing threshold at that frequency]) and a nonstartling sound (sound with the same level, but with a slower rise time) in wild-captured gray seals. The animals exposed to the startling treatment avoided a known food source, whereas animals exposed to the nonstartling treatment did not react or habituated during the exposure period. The results of this study highlight the importance of the characteristics of the acoustic signal in an animal's response of habituation.

Pinnipeds may be the least sensitive taxonomic group to most noise sources, although some species may be more sensitive than others. Pinnipeds are likely to only respond to loud impulsive sound sources at close ranges by startling, jumping into the water when hauled out, or even cease foraging, but only for brief periods before returning to their previous behavior (e.g., Southall et al. (2007a)). Pinnipeds may even experience TTS before exhibiting a behavioral response Southall et al. (2007a).

8.2.1.2.9 Impact Range to Effects from Explosives

Section 2.2.2 presented information on the criteria and thresholds used to estimate impacts to marine mammals from explosives. Additional information on these criteria is described in the technical report *Criteria and Thresholds for U.S Navy Acoustic and Explosive Impact to Marine Mammals and Sea Turtles* (Navy 2017a). In this section we present information on calculated range to effects for various explosive sources used by the Navy as part of the proposed action.

The tables below (Table 122 through Table 129) provide range to effects for explosives sources to the criteria and thresholds described in Section 2.2.2 as they were used as inputs into NAEMO. The range to effects are shown for a range of explosive bins from E1 (up to 0.25 pounds NEW) to E11 (greater than 500 pounds to 650 pounds NEW). Ranges are determined by modeling the distance that noise from an explosion will need to propagate to reach exposure level thresholds specific to a hearing group that will cause a non-auditory injury, PTS, TTS and

significant behavioral disruption. For events with multiple explosions, sound from successive explosions can be expected to accumulate and increase the range to the onset of an impact based on SEL thresholds. Modeled ranges to TTS and PTS based on peak pressure for a single explosion generally exceed the modeled ranges based on SEL even when accumulated for multiple explosions. Peak pressure based ranges are estimated using the best available science; however, data on peak pressure at far distances from explosions are very limited. For additional information on how ranges to impacts from explosions were estimated, see the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing Ranges* (Navy 2018d).

Ranges to mortality, based on animal mass, are shown in Table 122 shows the minimum, average, and maximum ranges due to varying propagation conditions to non-auditory injury as a function of animal mass and explosive bin. Ranges to gastrointestinal tract injury typically exceed ranges to slight lung injury; therefore, the maximum range to effect is not mass dependent. Animals within these water volumes would be expected to receive minor injuries at the outer ranges, increasing to more substantial injuries, and finally mortality as an animal approaches the detonation point. SEL-based and peak based range to effects (PTS, TTS, and behavioral for SEL only) for low-frequency cetaceans are shown by bin and cluster size in Table 124 and Table 125, respectively. For mid-frequency cetaceans, SEL-based and peak based range to effects are shown in Table 126 and Table 127, respectively. Similarly for otariids, SEL-based and peak based range to effects are shown in Table 128 and Table 129, respectively.

Table 122. Ranges to Mortality (in meters) for All Marine Mammal Hearing Groups as a Function of Animal Mass.

Bin[2]	*Range to Mortality (meters) for Various Animal Mass Intervals (kg)*[1]					
	10 kg	*250 kg*	*1,000 kg*	*5,000 kg*	*25,000 kg*	*72,000 kg*
E1	3 (2–3)	1 (0–3)	0 (0 –0)	0 (0–0)	0 (0–0)	0 (0–0)
E2	4 (3–5)	2 (1–3)	1 (0–1)	0 (0–0)	0 (0–0)	0 (0–0)
E3	10 (9–20)	5 (3–20)	2 (1–5)	0 (0–3)	0 (0–1)	0 (0–1)
E4	13 (11–19)	7 (4–13)	3 (2–4)	2 (1–3)	1 (1–1)	1 (0–1)
E5	13 (11–15)	7 (4–11)	3 (3–4)	2 (1–3)	1 (1–1)	1 (0–1)
E7	49 (40–80)	27 (15–60)	13 (10–20)	9 (5–12)	4 (4–6)	3 (2–4)
E8	65 (60–75)	34 (22–55)	17 (14–20)	11 (9–13)	6 (5–6)	5 (4–5)
E10	43 (40–50)	25 (16–40)	13 (11–16)	9 (7–11)	5 (4–6)	4 (3–4)
E11	185 (90–230)	90 (30–170)	40 (30–50)	28 (23–30)	15 (13–16)	11 (9–13)

[1]Average distance to mortality (meters) is depicted above the minimum and maximum distances (in parentheses), which are in parentheses for each animal mass interval.
[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 - 5), E5 (> 5 - 10), E7 (> 20 - 60), E8 (> 60 - 100), E10 (> 250 - 500), E11 (> 500 - 650)
Note: kg = kilogram

Table 123. Ranges to non-auditory Injury (in meters) for all marine mammal hearing groups.

Bin[2]	*Range to Non-Auditory Injury (meters)* [1]
E1	12 (11–13)
E2	16 (15–16)
E3	25 (25–45)
E4	31 (23–50)
E5	40 (40–40)
E7	104 (80–190)
E8	149 (130–210)
E10	153 (100–400)
E11	419 (350–725)

[1] Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.
Notes: Modeled ranges based on peak pressure for a single explosion generally exceed the modeled ranges based on impulse (related to animal mass and depth).
[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 - 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 - 100), E10 (> 250 - 500), E11 (> 500 - 650)

Table 124. Sound Exposure Level (SEL)-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Low-Frequency Cetaceans.

Range to Effects for Explosives Bin: Low-frequency cetaceans[1]					
Bin[2]	***Source Depth (meters)***	***Cluster Size[3]***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***	***Range to Behavioral (meters)[1]***
E1	0.1	1	52 (50—55)	221 (120—250)	354 (160—420)
E2	0.1	1	66 (55—70)	276 (140—320)	432 (180—525)
E3	10 Inland Waters	1	330 (160—550)	1,583 (160—4,025)	2,085 (160—7,525)
	18.25 offshore area	1	198 (180—220)	1,019 (490—2,275)	1,715 (625—4,025)
E4	10	2	462 (400—600)	3,743 (2,025—7,025)	6,292 (2,525—13,275)
	30	2	527 (330—950)	3,253 (1,775—4,775)	5,540 (2,275—8,275)
	70	2	490 (380—775)	3,026 (1,525—4,775)	5,274 (2,275—7,775)
	90	2	401 (360—500)	3,041 (1,275—4,525)	5,399 (1,775—9,275)
E5	0.1	1	174 (100—260)	633 (220—850)	865 (270—1,275)
		8	403 (170—500)	1,111 (320—1,775)	1,504 (430—2,525)
E7	10	1	1,375 (875—2,525)	7,724 (3,025—15,025)	11,787 (4,525—25,275)
	30	1	1,334 (675—2,025)	7,258 (2,775—11,025)	11,644 (4,525—24,275)
E8	45.75	1	1,227 (575—2,525)	3,921 (1,025—17,275)	7,961 (1,275—48,525)
E10	0.1	1	546 (200—700)	1,522 (440—5,275)	3,234 (850—30,525)
E11	91.4	1	2,537 (950—5,525)	11,249 (1,775—50,775)	37,926 (6,025—94,775)
	200	1	2,541 (1,525—4,775)	7,407 (2,275—43,275)	42,916 (6,275—51,275)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

[3] Significant behavioral reactions beyond the TTS range are only estimated in instances where explosive cluster size is greater than one (U.S. Department of the Navy 2018b).

Table 125. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Low Frequency Cetaceans.

Range to Effects for Explosives Bin: Low-frequency cetaceans[1]			
Bin[2]	***Source Depth (meters)***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***
E1	0.1	133 (90—150)	234 (110—270)
E2	0.1	165 (100—180)	288 (120—340)
E3	10 Inland Waters	450 (160—1,000)	907 (160—3,275)
	18.25 offshore area	355 (260—825)	664 (390—1,775)
E4	10	402 (370—430)	833 (650—1,275)
	30	582 (300—975)	938 (470—2,025)
	70	571 (370—1,275)	891 (550—1,775)
	90	437 (370—750)	933 (650—1,525)
E5	0.1	410 (150—500)	683 (210—900)
E7	10	1,121 (750—2,025)	2,248 (1,025—4,775)
	30	1,307 (525—2,275)	1,829 (775—3,775)
E8	45.75	1,486 (575—3,525)	2,130 (800—5,775)
E10	0.1	925 (280—1,275)	1,243 (350—1,775)
E11	91.4	2,845 (950—7,525)	3,662 (1,025—9,025)
	200	3,284 (1,525—6,025)	4,586 (1,775—8,275)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

Table 126. Sound Exposure Level (SEL)-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Mid-Frequency Cetaceans.

Range to Effects for Explosives Bin: Mid-frequency cetaceans[1]					
Bin[2]	***Source Depth (meters)***	***Cluster Size[3]***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***	***Range to Behavioral (meters)[1]***
E1	0.1	1	25 (25—25)	118 (110—120)	203 (190—210)
E2	0.1	1	30 (30—30)	146 (140—150)	246 (230—250)
E3	10 Inland Waters	1	61 (50—100)	512 (160—750)	928 (160—2,025)
	18.25 offshore area	1	40 (35—40)	199 (180—280)	368 (310—800)
E4	10	2	73 (70—75)	445 (400—575)	765 (600—1,275)
	30	2	71 (65—90)	554 (320—1,025)	850 (525—1,775)
	70	2	63 (60—85)	382 (320—675)	815 (525—1,275)
	90	2	59 (55—85)	411 (310—900)	870 (525—1,275)
E5	0.1	1	79 (75—80)	360 (350—370)	575 (525—600)
		8	198 (190—200)	804 (675—875)	1,126 (825—1,275)
E7	10	1	121 (110—130)	742 (575—1,275)	1,272 (875—2,275)
	30	1	111 (100—130)	826 (500—1,775)	1,327 (925—2,275)
E8	45.75	1	133 (120—170)	817 (575—1,525)	1,298 (925—2,525)
E10	0.1	1	273 (260—280)	956 (775—1,025)	1,370 (900—1,775)
E11	91.4	1	242 (220—310)	1,547 (1,025—3,025)	2,387 (1,275—4,025)
	200	1	209 (200—300)	1,424 (1,025—2,025)	2,354 (1,525—3,775)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift.

[3] Significant behavioral reactions beyond the TTS range are only estimated in instances where explosive cluster size is greater than one (U.S. Department of the Navy 2018b).

Table 127. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Mid-Frequency Cetaceans.

Range to Effects for Explosives Bin: Mid-frequency cetaceans[1]			
Bin[2]	***Source Depth (meters)***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***
E1	0.1	44 (40—45)	86 (80—90)
E2	0.1	59 (55—60)	106 (100—110)
E3	10 Inland Waters	122 (100—230)	245 (160—410)
	18.25 offshore area	100 (100—100)	190 (180—280)
E4	10	120 (120—120)	247 (240—260)
	30	136 (120—220)	365 (230—750)
	70	129 (120—200)	257 (230—440)
	90	126 (120—190)	247 (230—380)
E5	0.1	160 (150—170)	295 (280—300)
E7	10	309 (300—370)	592 (525—825)
	30	483 (290—850)	840 (525—1,775)
E8	45.75	561 (350—1,025)	1,056 (625—2,275)
E10	0.1	557 (490—600)	878 (625—1,025)
E11	91.4	1,187 (650—2,525)	2,272 (1,025—4,275)
	200	683 (650—950)	1,972 (1,025—4,025)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

Table 128. Sound Exposure Level (SEL) Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Otariids.

Range to Effects for Explosives Bin: Otariids[1]					
Bin[2]	***Source Depth (meters)***	***Cluster Size[3]***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***	***Range to Behavioral (meters)[1]***
E1	0.1	1	7 (7—8)	34 (30—35)	58 (55—60)
E2	0.1	1	9 (9—10)	43 (40—45)	72 (70—75)
E3	10 Inland Waters	1	21 (18—25)	135 (120—210)	250 (160—370)
	18.25 offshore area	1	15 (15—15)	91 (85—95)	155 (150—160)
E4	10	2	30 (30—30)	175 (170—180)	312 (300—350)
	30	2	25 (25—25)	176 (160—250)	400 (290—750)
	70	2	26 (25—35)	148 (140—200)	291 (250—400)
	90	2	26 (25—35)	139 (130—190)	271 (250—360)
E5	0.1	1	25 (24—25)	111 (110—120)	188 (180—190)
		8	62 (60—65)	284 (270—290)	462 (430—490)
E7	10	1	60 (60—60)	318 (300—360)	575 (500—775)
	30	1	53 (50—65)	376 (290—700)	742 (500—1,025)
E8	45.75	1	55 (55—55)	387 (310—750)	763 (525—1,275)
E10	0.1	1	87 (85—90)	397 (370—410)	599 (525—675)
E11	91.4	1	100 (100—100)	775 (550—1,275)	1,531 (900—3,025)
	200	1	94 (90—100)	554 (525—700)	1,146 (900—1,525)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift.

[3] Significant behavioral reactions beyond the TTS range are only estimated in instances where explosive cluster size is greater than one (U.S. Department of the Navy 2018b).

Table 129. Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Otariids.

Range to Effects for Explosives Bin: Otariids[1]			
Bin[2]	***Source Depth (meters)***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***
E1	0.1	37 (35—40)	69 (65—70)
E2	0.1	48 (45—50)	88 (80—90)
E3	10 Inland Waters	99 (85—170)	197 (150—370)
	18.25 offshore area	80 (80—85)	154 (150—200)
E4	10	100 (100—100)	190 (190—190)
	30	105 (100—140)	262 (190—675)
	70	106 (100—160)	206 (190—350)
	90	103 (100—150)	197 (190—320)
E5	0.1	128 (120—130)	243 (230—250)
E7	10	255 (250—260)	471 (440—500)
	30	419 (240—1,025)	722 (440—1,025)
E8	45.75	434 (280—975)	913 (525—2,025)
E10	0.1	476 (450—490)	739 (600—875)
E11	91.4	934 (525—1,775)	1,912 (1,000—3,775)
	200	553 (525—800)	1,516 (1,000—3,525)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances (in parentheses) due to varying propagation environments in parentheses.
[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)
Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

8.2.1.2.10 Exposure Analysis – Marine Mammal Exposure to Explosives

The numbers of potential impacts from the quantitative analysis estimated for individual species of marine mammals from exposure to explosive energy and sound for training activities are presented below for each species. Results are presented for a maximum explosive use year. With the exception of a small number of explosive mine countermeasure and neutralization testing activities, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore.

Blue Whales

The quantitative analysis, using the maximum number of explosives per year under the proposed action, estimates no impacts to blue whales from training activities. The analysis estimates three behavioral reactions are predicted for blue whales every seven years of the proposed action, and one behavioral reaction per year of maximum activity from testing activities are predicted for blue whales in the offshore portion of the action area (Table 130 and Figure 97).

Table 130. Estimated impacts on blue whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area.

Estimated Impacts by Effect								
Timeframe	*Training*				*Testing*			
	Behavioral	*TTS*	*PTS*	*Injury*	*Behavioral*	*TTS*	*PTS*	*Injury*
Seven-year Period	0	0	0	0	3	0	0	0
One-year Maximum	0	0	0	0	1	0	0	0

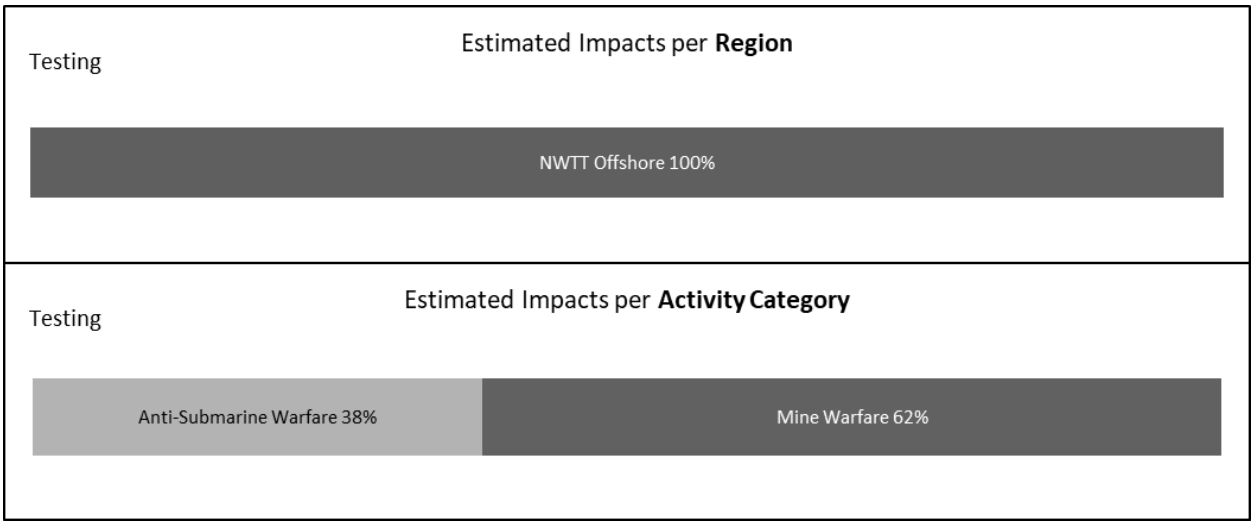


Figure 97. Estimated impacts on blue whales per year from explosives used during NWTT by region and activity category.

Fin Whales

The quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts to fin whales from training activities (Table 131). For testing

activities, 39 exposures resulting in behavioral reactions and nine exposures resulting in TTS are predicted for fin whales every seven years of the proposed action, and six exposures resulting in behavioral reactions and two exposures resulting in TTS per year of maximum activity are predicted for fin whales in the offshore region of the action area (Table 131 and Figure 98). No exposures from explosives were estimated to impact fin whales from the Northeast Pacific stock.

Table 131. Estimated impacts on fin whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area.

Estimated Impacts by Effect								
Timeline	*Training*				*Testing*			
	Behavioral	*TTS*	*PTS*	*Injury*	*Behavioral*	*TTS*	*PTS*	*Injury*
Seven-year Period	0	0	0	0	39	9	0	0
One-year Maximum	0	0	0	0	6	2	0	0

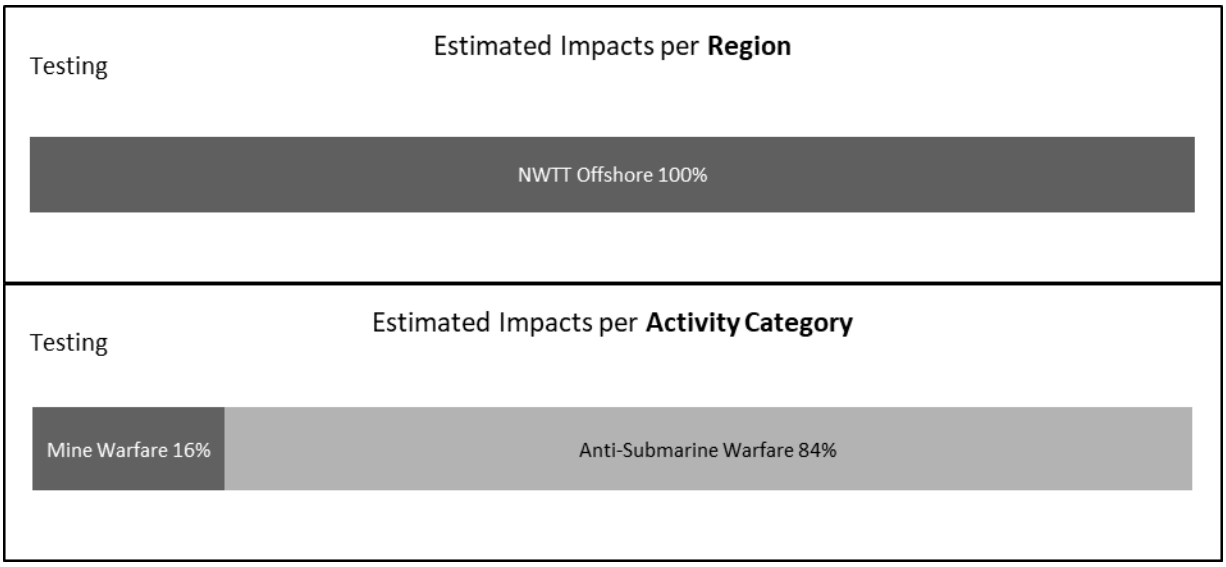


Figure 98. Estimated impacts on fin whales per year from explosives used during NWTT by region and activity category.

Humpback Whale – Central America DPS and Mexico DPS

The quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts for training activities for both the Mexico and Central America DPSs of humpback whales (Table 132). For testing activities, five exposures resulting in

behavioral harassment and five exposures resulting in TTS are predicted for the humpback whale Mexico DPS every seven years of the proposed action, and one exposure resulting in behavioral harassment and one exposure resulting in TTS are predicted for the humpback whale Mexico DPS per year of maximum activity in the offshore region of the action area (Table 132 and Figure 99). Based on the results of the Navy's quantitative analysis and the proposed procedural mitigation measures, the likelihood of Central America DPS humpback whales being exposed to explosive sound and energy in the offshore portion of the action area is extremely unlikely, and thus considered discountable.

Table 132. Estimated impacts on the humpback whale Mexico DPS every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area.

Estimated Impacts by Effect								
Timeframe	*Training*				*Testing*			
	Behavioral	*TTS*	*PTS*	*Injury*	*Behavioral*	*TTS*	*PTS*	*Injury*
Seven-year Period	0	0	0	0	5	5	0	0
One-year Maximum	0	0	0	0	1	1	0	0

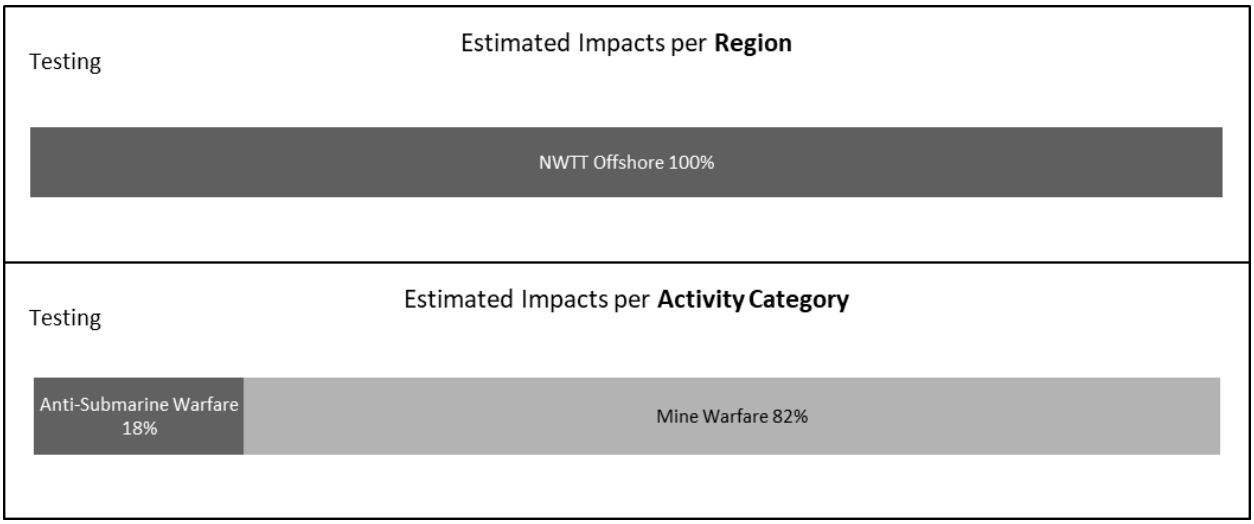


Figure 99. Estimated impacts on the humpback whale Mexico DPS per year from explosives used during NWTT by region and activity category.

Southern Resident Killer Whale DPS

The Navy's quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts to Southern Resident killer whales from explosives used as part of NWTT Phase III activities. Explosive use within the Inland Waters portion of the action area is limited to a small number of low NEW charges (bin E3 [< 0.5–2.5 lb.] or less) that would be used during Mine Neutralization – EOD training events at Crescent Harbor EOD Training Range and Hood Canal EOD Training Range. Southern Resident killer whales are very rare in Hood Canal (e.g. have not been documented since 1995), although they have been sighted in waters near the Crescent Harbor EOD Range. Given the limited range to effects of these relatively small explosives (from Table 125 and Table 126 above), the small number proposed (six bin E3 explosions per year) in inland waters, implementation of procedural mitigation (as described in Section 3.5.1), and results of the Navy's quantitative analysis, the likelihood of exposure of this DPS to explosive sound and energy in inland waters is extremely unlikely, and thus considered discountable.

With the exception of a small number of explosive mine countermeasure and neutralization testing activities, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles (57.5 miles) from shore. While Southern Resident killer whales DPS killer whales may be present in the offshore portion throughout the year, they spend nearly all of their time closer to shore (i.e., within 22 miles) than the locations where most in-water explosions would be detonated during NWTT activities (Hanson et al. 2017b).

Mine countermeasure and neutralization testing is a new activity in Phase III that would occur closer to shore than other in-water explosive activities analyzed in the NWTT Phase II biological opinion. This activity would occur greater than three nautical miles from shore in the Quinault Range Site (E4 would be used more than three nautical miles offshore and E7 would be used more than six nautical miles offshore) or greater than 12 nautical miles from shore elsewhere in the offshore area. Explosive mine countermeasure and neutralization testing activities, therefore, could occur in waters inhabited by the Southern Resident killer whale DPS. Offshore mine countermeasure and neutralization testing activities would occur in water depths ranging from 72 to no more than 1,000 feet(typically occurring at around 300 feet), and would involve up to 36 ordnance in bin E4 (2.5 to 5 lbs. NEW) and up to 5 ordnance in bin E7 (20 to 60 lbs. NEW) maximum within a year. Over the course of every seven-year period, the Navy would limit the use of explosives to 108 E4 and 15 E7 explosives. Observation of the mitigation zone both before and during testing events (as described in Section 3.5.1 *Procedural Mitigation*) would likely reduce the potential for injurious exposures in the offshore area. For personnel safety, the Navy has proposed to conduct offshore explosive mine countermeasure and neutralization testing activities only during daylight hours and in Beaufort sea states of three or less. This measure should improve the effectiveness of procedural mitigation using lookouts during this activity. In addition to a Lookout positioned on a vessel, mine countermeasure neutralization testing events involving bin E7 explosives would also have a Lookout positioned in the aircraft asset, which should improve the effectiveness of this mitigation when implementing the larger mitigation

zone (i.e., 2,100 yards (1,920 meters)) for the larger weight explosives. In addition, the sea state requirement increases the likelihood that this activity would occur during the summer months when Southern Resident killer whales are expected to spend more time in the inland portion of the action area, and less time offshore. In summary, based on the results of the Navy's quantitative analysis estimating no impacts, the proposed procedural mitigation measures, the small number of explosive activity events proposed in offshore areas that overlap spatially with Southern Resident killer whale offshore distribution, and the requirement to conduct offshore mine countermeasure and neutralization testing only during daylight hours and in Beaufort sea states of three or less, the likelihood of this DPS being exposed to explosive sound and energy in the offshore portion of the action area is extremely unlikely, and thus considered discountable.

Sei Whales

The Navy's quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts for training activities for sei whales. For testing activities, eight exposures resulting in behavioral harassment and three exposures resulting from TTS are predicted for sei whales every seven years of the proposed action, and one exposure resulting in behavioral harassment and one exposure resulting in TTS are predicted for sei whales per year of maximum activity in the offshore region of the action area (Table 133 and Figure 100).

Table 133. Estimated impacts on sei whales every seven year period and over a year of maximum activity as a result of the proposed Navy training and testing activities using explosives in the NWTT action area.

Estimated Impacts by Effect								
Timeframe	*Training*				*Testing*			
	Behavioral	*TTS*	*PTS*	*Injury*	*Behavioral*	*TTS*	*PTS*	*Injury*
Seven-year Period	0	0	0	0	8	3	0	0
One-year Maximum	0	0	0	0	1	1	0	0

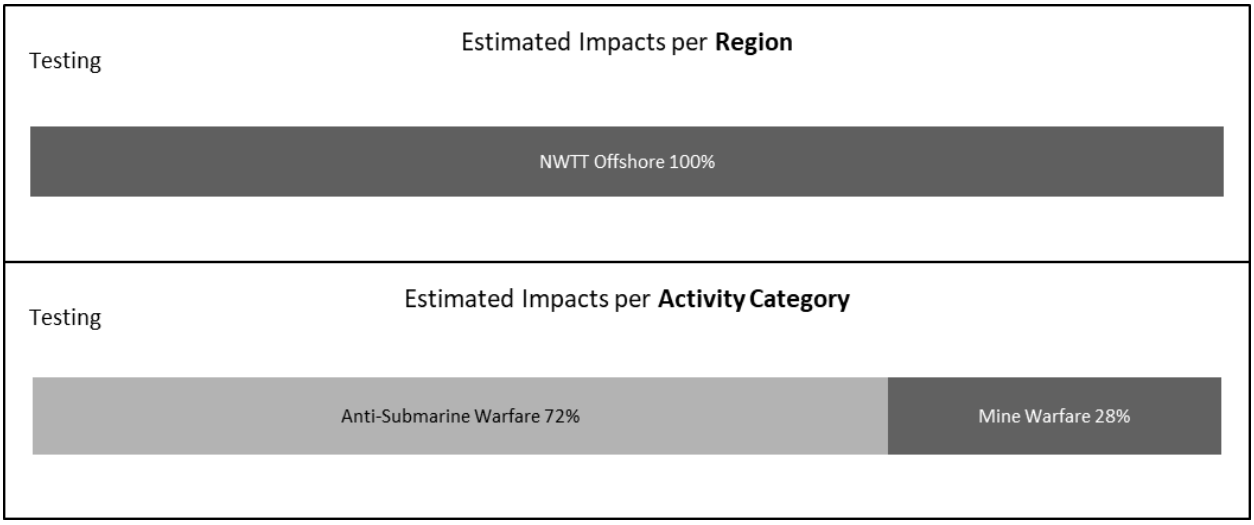


Figure 100. Estimated impacts on sei whales per year from explosives used during NWTT by region and activity category.

Sperm Whales

The quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts to sperm whales due to training or testing activities. Based on the results of the Navy's quantitative analysis and the proposed procedural mitigation measures, the likelihood of sperm whales being exposed to explosive sound and energy in the offshore portion of the action area is extremely unlikely, and thus considered discountable.

Guadalupe Fur Seals

The quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts to Guadalupe fur seals due to training or testing activities. Based on the results of the Navy's quantitative analysis and the proposed procedural mitigation measures, the likelihood of Guadalupe fur seals being exposed to explosive sound and energy in the offshore portion of the action area is extremely unlikely, and thus considered discountable.

8.2.1.2.11 Response Analysis

Above, we described the range of potential responses of ESA-listed marine mammals to explosives associated with the proposed action. Given the above estimated exposures of ESA-listed marine mammals to explosives, in this section we describe the likely responses of these species to this exposure. This includes behavioral response, sound-induced hearing loss (i.e., TTS), as well as other possible responses (e.g., stress) that marine mammals may exhibit as a result of exposure to Navy explosives. As with our response analysis for the effects of sonars, our aim with this response analysis is to assess the potential responses to explosives that might reduce the fitness of individual ESA-listed marine mammals. In doing so, we consider and weigh evidence of adverse consequences, as well as evidence suggesting the absence of such consequences.

Hearing Threshold Shifts

The response of ESA-listed marine mammals from exposure to explosives resulting in TTS is expected to be similar to the response of ESA-listed marine mammals experiencing hearing loss due to sonar or other transducers. The exception is that because active sonar is transmitted at a specified frequency, animals experiencing TTS from sonar will only experience threshold shifts around that particular frequency. In contrast, explosives are a broadband source, so if an animal experiences TTS from explosives, a greater frequency band will be affected. Because a greater frequency band will be affected due to explosives, there is an increased chance that the hearing impairment will affect frequencies utilized by animals for acoustic cues. The exposure analysis indicates that the following numbers of exposures to explosives are expected to result in TTS of ESA-listed marine mammals per year as a result of NWTT activities: two for fin whales, one for Mexico DPS humpback whales, and one for sei whales. No other ESA-listed marine mammals are expected to experience TTS from Navy explosives in the action area.

Behavioral response

There are no direct observations of behavioral reactions from marine mammals due to exposure to explosive sounds. General research findings regarding potential behavioral reactions from marine mammals due to exposure to impulsive sounds, such as those associated with explosions, are discussed in detail earlier in this section. Behavioral reactions from explosive sounds could be similar to reactions studied for other impulsive sounds such as those produced by seismic air guns (e.g., startle reactions, avoidance of the sound source), but there are important differences in how seismic surveys using air guns are conducted compared with explosive use by the Navy. Seismic surveys using air guns are typically conducted over transects and successive air gun blasts occurring over a sustained period of time. In contrast, Navy explosive use typically involves a single detonation or series of detonations conducted over a short period of time. The available information on the response of humpback and sei whales to explosives indicates animals may alert to the sound source, may alter foraging behavior, or exhibit avoidance behavior. These responses are expected to be temporary with behavior returning to a baseline state shortly after the activity using explosives ends. The exposure analysis indicates that the following number of exposures to explosives are expected to result in significant behavioral disruptions of ESA-listed marine mammals per year as a result of NWTT activities: one for blue whales, six for fin whales, one for Mexico DPS humpback whales, and one for sei whales. No other ESA-listed marine mammals are expected to experience a significant behavioral disruption from Navy explosives in the action area.

Non-auditory Physical or Physiological responses

The available research on the potential for explosives or other sources of anthropogenic noise to result in physiological responses (e.g., stress) is described earlier in this section. Relatively little information exists on the linkage between anthropogenic sound exposure and stress in marine mammals, and even less information exists on the ultimate consequences of sound-induced stress

responses (either acute or chronic). However, increased stress has been documented as a result of both acute (e.g., Romano et al. 2004) and chronic (e.g., Rolland et al. 2012) anthropogenic noise. As described previously, though there are unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, it is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

Masking

Some limited masking could occur due to the Navy's use of explosives when animals are in close enough proximity. That is, if an animal is close enough to the source to experience TTS or a significant behavioral disruption, we anticipate some masking could occur. Masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Given that Navy explosive use typically involves a single detonation or series of detonations conducted over a short period of time, if masking occurs it would likely be a very short-term effect, one which we do not anticipate would result in the reduced fitness of individual ESA-listed marine mammals.

8.2.1.3 Anticipated Consequences of Acoustic Stressors on Individual Marine Mammals Exposed

In the exposure and response analyses above we established that a range of impacts including TTS, behavioral response, and physiological stress are likely to occur due to exposure to Navy sonar and explosives during NWTT activities. In this, section we assess the likely consequences of the responses to the individual ESA-listed marine mammals that have been exposed. We determined that the potential effects of masking from sonar are limited because of the duty cycles of most military sonars, the transient nature of sonar use, and the short duration of explosive sound effects. As such, we have concluded that there is little to no risk to marine mammals associated with exposure and response to the effects of masking.

Efforts have been made to link short-term effects to individuals due to anthropogenic stressors with long-term consequences to marine mammal populations using population models. Population models are well known from many fields in biology including fisheries and wildlife management. These models accept inputs for the population size and changes in vital rates of the population such as the mean values for survival age, lifetime reproductive success, and recruitment of new individuals into the population. Unfortunately, for acoustic and explosive impacts on cetacean populations, many of the inputs required by population models are not known. Nowacek et al. (2016) reviewed new technologies, including passive acoustic monitoring, tagging, and the use of unmanned aerial vehicles, which can improve scientists' abilities to study these model inputs and link behavioral changes to individual life functions and ultimately population-level effects. The Population Consequences of Acoustic Disturbance model (NRC 2005) proposes a conceptual framework for determining how changes in the vital rates of individuals (i.e., a biologically significant consequence to the individual) translates into biologically significant consequences to the population.

In 2009, the U.S. Office of Naval Research set up a working group to transform the Population Consequences of Acoustic Disturbance framework into a mathematical model and include other stressors potentially causing disturbance in addition to noise. The model, now called Population Consequences of Disturbance, has been used for case studies involving bottlenose dolphins, North Atlantic right whales, beaked whales, southern elephant seals, California sea lions, blue whales, humpback whales, and harbor porpoise (Costa et al. 2016a; Costa et al. 2016b; Harwood et al. 2014; Hatch et al. 2012; New et al. 2014; New et al. 2013a; New et al. 2013b; Pirotta et al. 2018). However, the Population Consequences of Disturbance model is still in the preliminary stages of development. Costa et al. (2016b) emphasized taking into account the size of an animal's home range, whether populations are resident and non-migratory or if they migrate over long areas and share their feeding or breeding areas with other populations. These factors, coupled with the extent, location, and duration of a disturbance can lead to markedly different impact results. Farmer et al. (2018) developed a bioenergetics framework to examine the impact of foraging disruption on body reserves of individual sperm whales. The authors examined rates of daily foraging disruption to predict the number of days to terminal starvation for various life stages, assuming exposure to seismic surveys. Mothers with calves were found to be most vulnerable to disruptions.

The long-term consequences due to individual behavioral reactions and short-term instances of physiological stress are especially difficult to predict because individual experience over time can create complex contingencies, especially for long-lived animals like cetaceans. Of critical importance in discussion on the potential consequences of such effects is the health of the individual animals disturbed, and the trajectory of the population those individuals comprise. The consequences of disturbance, particularly repeated effects, would be more significant if the affected animal were already in poor condition as such animals would be less likely to compensate for additional energy expenditures or lost foraging or reproductive opportunities. Short-term costs may be recouped during the life of an otherwise healthy individual. These factors are taken into consideration when assessing risk of long-term consequences to individuals exposed to the effects of Navy sonars and other transducers as part of the proposed action.

To consider the potential consequences of temporary hearing impacts, behavioral response, and stress to affected animals, we also consider the context of the exposure and response scenario including the following: 1) the duration of the exposure and associated response, 2) whether or not repeated exposures would be expected, 3) the behavioral state of the animal at the time of the response, and 4) the health of the animal at the time of the response.

Since marine mammals depend on acoustic cues for vital biological functions (e.g., orientation, communication, finding prey, avoiding predators), fitness consequences could occur to individual animals from hearing threshold shifts that last for a long time, occur at a frequency utilized by the animal for acoustic cues, and are of a profound magnitude. A hearing threshold shift of limited duration, occurring in a frequency range that does not coincide with that used for

vocalization or recognition of important acoustic cues would likely have no effect on an animal's fitness.

Based on our review of the available literature (discussed above), we expect instances of TTS from Navy sonar to be short-term and of relatively low severity because of animal avoidance and the transient nature of most Navy sonar sources. Because active sonar is transmitted at a specified frequency, animals experiencing TTS from sonar would only experience threshold shifts around that particular frequency. In contrast, explosives are a broadband source, so if an animal experiences TTS from explosives, a greater frequency band would be affected. Because a greater frequency band would be affected due to explosives, there is increased chance that the hearing impairment will affect frequencies utilized by animals for acoustic cues. The exposure analysis estimates two annual exposures to explosives resulting in TTS of fin whales, one annual exposure to explosives resulting in TTS of humpback whales, and one annual exposure to explosives resulting in TTS of sei whales. These numbers represent the estimated TTS exposures during a maximum year of NWTT explosive activity levels, and estimates in some years would be lower. No other ESA-listed marine mammals are expected to experience TTS from Navy explosives in the action area. Given these low exposure numbers, it is unlikely that an individual whale would experience TTS from Navy explosives more than once per year, or possibly per lifetime. Thus, adverse effects on acoustic cues resulting from exposure to TTS from explosives would likely be limited in scope and duration for individual whales.

The available literature on marine mammal behavioral responses indicate that most responses that have been observed to sonar exposure are of mild to moderate severity, often lasting for the duration of the exposure. Some more severe reactions have been observed, but these have mostly been in cetacean species known to be particularly sensitive to acoustic disturbance (e.g., beaked whales; Southall et al. 2016), which are not listed under the ESA. Based on information available to date, the marine mammal species considered in this opinion are not thought to be particularly sensitive to acoustic disturbance. However, it is worth noting that the controlled exposure experiments reviewed by Southall et al. (2016) were deliberately designed to demonstrate the onset of response and not to produce adverse or permanent effects. Additionally, the limitations of opportunistic observations (e.g., limited to observations of vocally-active marine mammals or animals at the surface, limited ability to monitor animal activity long-term, limited ability to control other variables which could impact animal behavior [e.g., prey distribution]) result in some uncertainty as to the severity and duration of likely responses of ESA-listed marine mammals due to sonar exposure. Forney et al. (2017) noted that species that respond to noise (e.g., from military sonars) by avoiding an area are unlikely to be observed using traditional methods (e.g., Lookouts or passive acoustic monitoring) because animals react at distances far greater than the detection range of these methods. They suggest that individuals that are observed must be considered relatively tolerant of anthropogenic noise.

The duration and magnitude of the proposed activity is important to consider in determining the likely severity, duration, and potential consequences of exposure and associated response to

Navy sonar and explosives. As noted in Southall et al. (2007a), substantive behavioral reactions to noise exposure (such as disruption of critical life functions, displacement, or avoidance of important habitat) are considered more likely to be significant if they last more than 24 hours, or recur on subsequent days. Several categories of training exercises are expected to result in hundreds of hours of sonar activity involving multiple platforms (i.e., surface vessels, submarines, and aircraft) utilizing sonar, as well as the use of explosives. These exercises range in duration from two days to over ten, and therefore have the potential to result in sustained and/or repeat exposure. However, there is no Navy activity in the proposed action that is both long in duration (more than a day) and concentrated in the same location (e.g., within a few square miles), so there is a low likelihood that animals and Navy activities would co-occur for extended periods of time or repetitively over the duration of an activity.

While it is difficult to predict exactly what a marine mammal may be doing at the time of exposure, we can make some predictions based on time of year and the location of the animal at the time of exposure, where such information is available. Humpback whales are known to feed in and around waters overlapping with the offshore portion of the action area (Bettridge et al. 2015b). Anthropogenic noise can have negative impacts on marine mammals if they result in the animals leaving the area, potentially away from a food source.

Currently, the Navy has proposed Geographic Mitigation Areas encompassing regions where humpback whales are known to feed, including Stonewall and Heceta Bank, and Point St. George. The Navy has proposed seasonal closures in these areas, restricting the use of mid-frequency active sonars and explosives during certain months of the year. For additional details about these Geographic Mitigation Areas, see the NWTT FSEIS (Navy 2020f).

Also important to consider is an animal's prior experience with a sound source. The majority of ESA-listed marine mammals exposed to sound from NWTT activities have likely been exposed to such sources previously as these activities have been occurring in the action area for decades. Harris et al. (2017a) suggested that processes such as habituation, sensitization, or learning from past encounters may lead to stronger or weaker reactions than those of a naïve animal. For example, Baird et al. (2017) found no large-scale avoidance by false killer whales of areas with relatively high mid-frequency active sonar use in the Pacific Missile Range Facility in Hawaii. The authors suggested that since sonar had been used at Pacific Missile Range Facility for over 30 years, it was likely that animals in this area had been exposed to sonar multiple times on previous occasions. The authors suggested that more naïve populations may be more likely to exhibit avoidance responses if exposed to sonar.

When considering the potential consequences of exposure and response to Navy sonar and explosives, we must also take into account the health of the individual animal affected. Individuals that are in good health, with sufficient energy reserves, are likely to be much more resilient when faced with acoustic stressors than an animal in poor condition. As described in Harris et al. (2017a), one approach to understanding the potential importance of a behavioral

response is to consider an animal's energy budget. Cetacean behavioral research has indicated that many species including humpback whales (Sivle et al. 2016), blue whales (Goldbogen et al. 2013), and sperm whales (Isojunno et al. 2016) may disrupt foraging when exposed to anthropogenic noise. If the animals are not able to make up for lost foraging opportunities due to such exposure, this could have consequences on the affected animal's available energy supply. For individuals in good health, with sufficient energy reserves, such a reduction could likely be compensated for at a later time, provided the animal is not subject to sustained disruption. However, for individuals in a compromised state, a reduction in available energy has a higher likelihood of being consequential, depending on the duration of the disruption (i.e., long duration disruptions would have a higher likelihood of being consequential).

Quantifying the fitness consequences of sub-lethal impacts is exceedingly difficult for marine mammals because of the limitations of studying these species (e.g., due to the costs and logistical challenges of studying animals that spend the majority of time underwater). Harris et al. (2017a) summarized the research efforts conducted to date that have attempted to understand the ways in which behavioral responses may result in long-term consequences to individuals and populations. Efforts have been made to try and quantify the potential consequences of such responses, and frameworks have been developed for this assessment (e.g., Population Consequences of Disturbance). However, models that have been developed to date to address this question require many input parameters and, for most species, there are insufficient data for parameterization (Harris et al. 2017a). A key limitation in these models is that we often do not have empirical data to link sub-lethal behavioral responses to effects on animal vital rates.

Behavioral responses may impact health through a variety of different mechanisms, but most Population Consequences of Disturbance models focus on how such responses affect an animal's energy budget (Farmer et al. 2018; King et al. 2015; NAS 2017; New et al. 2014; Villegas-Amtmann et al. 2017). Responses that relate to foraging behavior, such as those that may indicate reduced foraging efficiency or involve the complete cessation of foraging, may result in an energetic loss to animals (Miller et al. 2009). Other behavioral responses, such as avoidance, may have energetic costs associated with traveling (Bejder et al. 2019; NAS 2017). Important in considering whether or not energetic losses, whether due to reduced foraging or increased traveling, will affect an individual's fitness is considering the duration of exposure and associated response. Nearly all studies and experts agree that infrequent exposures of a single day or less are unlikely to impact an individual's overall energy budget and that long duration and repetitive disruptions would be necessary to result in consequential impacts on an animal (Farmer et al. 2018; Harris et al. 2017b; King et al. 2015; NAS 2017; New et al. 2014; Southall et al. 2007a; Villegas-Amtmann et al. 2015).

We also recognize that aside from affecting health via an energetic cost, a behavioral response could result in more direct impacts to health and/or fitness. For example, if a marine mammal hears Navy sonar or an explosion and avoids the area, this may cause it to travel to an area with other threats such as vessel traffic or fishing gear. However, we find such possibilities (i.e., that a

behavioral response would lead directly to a ship strike) to be extremely unlikely and not reasonably certain to occur due to the low densities of the ESA-listed marine mammals in the action area and the size and relative ease of detection of these species by shipboard observers. Therefore, we focus our risk analysis on the energetic costs associated with a behavioral response.

We would expect many of the anticipated exposures and potential responses of ESA-listed marine mammals to sonar and other transducers and explosives to have little effect on the exposed animals. Based on the controlled exposure experiments and opportunistic research presented above, responses are expected to be short term, with the animal returning to normal behavioral patterns shortly after the exposure is over. However, there is some uncertainty due to the limitations of the controlled exposure experiments and observational studies used to inform our analysis. Additionally, Southall et al. (2016) suggested that even minor, sub-lethal behavioral changes may still have significant energetic and physiological consequences given sustained or repeated exposure. Quantifying the fitness consequences of sub-lethal impacts from acoustic stressors is exceedingly difficult for cetaceans and we do not currently have data to conduct a quantitative analysis on the likely consequences of such sub-lethal impacts. While we are unable to conduct a quantitative analysis on how sub-lethal behavioral effects and temporary hearing impacts may impact animal vital rates (and therefore fitness), based on the best available information, we expect an increased likelihood of consequential effects when exposures and associated effects are long-term and repeated, occur in locations where the animals are conducting critical activities, and when the animal affected is in a compromised state.

During exposure, affected animals may be engaged in any number of activities including, but not limited to, migration, foraging, nursing, or resting. If marine mammals exhibited a behavioral response to Navy sonar, these activities would be disrupted and it may pose some energetic cost. However, as noted previously, responses to Navy sonar are anticipated to be short term and instances of hearing impairment are expected to be mild or moderate. Based on best available information that indicates marine mammals resume normal behavior quickly after the cessation of sound exposure (e.g., Goldbogen et al. 2013; Melcon et al. 2012), we anticipate that exposed animals will be able to return normal behavioral patterns after this short duration activity ceases. Goldbogen et al. (2013) suggested that if the documented temporary behavioral responses interrupted behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual animal could not compensate for this lost resting/nursing or feeding opportunity by either moving to another location, by stopping the activity until shortly after cessation of acoustic exposure, or by resting/nursing or feeding at a later time. There is no indication this is the case. There would likely be an energetic cost associated with any temporary disruption of humpback whale resting/nursing activities or other marine mammal feeding activities to find alternative locations for these to occur. However, unless such disruptions occur over long durations or over subsequent days, we do not anticipate these movement to be consequential to the animal's fitness

over the long-term (Southall et al. 2007a). While activities could be conducted for up to ten days, there is no Navy activity in the proposed action that is both long in duration (more than a day) and concentrated in the same location.

Based on the estimated abundance of the ESA-listed marine mammals that are expected to occur in the action area, and the estimated maximum annual number of instances of behavioral disruption (i.e., TTS or significant behavioral response) expected from sonar and explosives (i.e., estimates based on Navy modeling using maximum annual activity level), most individuals of these species would be exposed, and respond, to Navy sonar, on average, less than once per year (Table 134). ESA-listed marine mammal annual abundance numbers were obtained from the Navy's August 2020 technical report for the Marine Species Density Database Phase III for the Northwest Training and Testing Study Area (Navy 2019f), except for Southern Resident killer whales. For those seasonal species exhibiting densities in the action area that vary throughout the year, the species density most representative of each species (covering the greatest area in the density distribution maps provided by Navy 2019f) in each portion of the action area for each season was used to calculate the estimated abundance throughout the entire action area. The lower abundance values, if applicable, were then used to obtain conservative estimates of the annual number of behavioral disruptions per animal. The highest number of behavioral disruptions per animal is anticipated to be of the Southern Resident killer whale DPS (i.e., 0.689 disruptions per animal). For all other species, less than 0.67 behavioral disruptions are anticipated per animal annually. This indicates that multiple exposures of the same individual within a year would likely be rare, and some (or many, depending on the species or DPS) individuals within the population would not experience a single behavioral disruption per year due to Navy sonar and explosives.

Table 134. Estimated average behavioral disruptions (i.e., TTS or significant behavioral response) from Navy sonar (and other transducers) and explosives per animal of each species/stock/DPS in the action area. These estimates are based on a year of estimated maximum behavioral disruptions during a year of maximum sonar and explosive activity levels.

Species	Size of Listing Unit Likely to be Found in Action Area	Annual Behavioral Disruptions from Active Sonar and Explosives	Annual Disruptions per Animal
Blue Whale – Eastern North Pacific Stock	455	10	0.022
Fin Whale – California/Oregon/Washington Stock	788	137	0.174
Humpback Whale – Central America and Mexico DPSs	759	93	0.123
Killer Whale – Southern Resident DPS	74	51	0.689

Species	Size of Listing Unit Likely to be Found in Action Area	Annual Behavioral Disruptions from Active Sonar and Explosives	Annual Disruptions per Animal
Sei Whale – Eastern North Pacific Stock	166	83	0.500
Sperm Whale – California/Oregon/Washington Stock	1,253	839	0.670
Guadalupe Fur Seal – Mexico Stock[1]	2,733	1,495	0.547

We recognize that the calculation of the number of disruptions per animal is based on Navy modeling and is a rough approximation of what will occur during Navy training and testing activities in the action area. Therefore, some individuals from each species could experience a few more or less disruptions annually than what is presented. However, due to the limitations on acoustic exposure modeling capabilities, we are unable to identify which individual from each population will be exposed to and affected by a particular training or testing event in the action area. For this reason, we are not able to predict exactly how many times each animal in the action area will be exposed to and affected by Navy sonar and explosives annually. The estimates presented in Table 135 are based on conservative assumptions, and are provided to indicate the relative magnitude of likely exposures on an annual basis.

In summary, we anticipate some animals in the action area could experience more than one behavioral disruption per year, but animals would be exposed periodically and based on the available literature that indicates infrequent exposures are unlikely to impact an individual's overall energy budget (Farmer et al. 2018; Harris et al. 2017b; King et al. 2015; NAS 2017; New et al. 2014; Southall et al. 2007a; Villegas-Amtmann et al. 2015), we do not expect this level of exposure to impact the fitness of exposed animals. Further, we anticipate that any instances of TTS will be of minimum severity and short duration. This conclusion is based on literature indicating that even following relatively prolonged periods of sound exposure resulting in TTS, recovery occurs quickly (Finneran 2015a). The brief amount of time marine mammals are expected to experience TTS is unlikely to significantly impair their ability to communicate, forage, or breed and is not expected to have long-term fitness consequences for the individuals affected. Additionally, we do not anticipate these species will experience long duration or repeat exposures within a short period of time due to the species' wide ranging life history and that long duration (i.e., more than one day) Navy activities also occur over large geographic areas (i.e., both the animal and the activity are moving within the action area, most likely not in the same direction). This decreases the likelihood that animals and Navy activities will co-occur for extended periods of time or repetitively over the duration of an activity. Although there is an increased chance that TTS resulting from explosives would affect frequencies utilized by animals for acoustic cues (as compared to TTS from sonar), the Navy's quantitative model predicts very few instances of TTS from explosives. Since it is unlikely that an individual marine mammal

would experience TTS from Navy explosives on multiple occasions, adverse effects on acoustic cues resulting from such exposures would likely be limited in scope and duration for individual whales.

For the reasons above, we do not anticipate that instances of behavioral response or TTS from Navy activities involving sonar (and other transducers) and explosives would result in long-term fitness consequences to individual ESA-listed marine mammals in the action area. The Navy's continued implementation of procedural mitigation to avoid or reduce potential impacts on marine mammals from active sonar and explosives further decreases the likelihood of long-term fitness consequences resulting from these activities (see Section 3.5.1 *Procedural Mitigation* for details).

8.2.1.4 *Effects of Explosives on Southern Resident Killer Whales via Impacts to their Prey*

As discussed above in the status of the species (Section 6.2.4), limitations in Southern Resident killer whale prey availability is considered one of the primary threats affecting the survival and recovery of this endangered species. The injury and mortality of fish that would result from NWTT stressors could have indirect effects on Southern Resident killer whales by reducing prey availability. A reduction in the availability of their prey may cause killer whales to forage for longer periods, travel to alternate locations, or abandon foraging efforts. Nutritional stress may also be a factor affecting adult reproductive success and calf survival. We analyzed the effects of the proposed action on Southern Resident killer whales via impacts to their prey in two steps. First, we estimated the reduction in prey available to the Southern Resident population caused by the Navy's proposed action. Next, we considered information to help put the reductions in context and evaluate whether the effects on their prey are likely to result in adverse effects to individual Southern Resident killer whales. In Section 8.3.1.2 below, we again discuss the effects of the proposed action on prey availability, this time in the context of the prey essential feature of Southern Resident killer whale designated and proposed critical habitat.

Other than explosives, we found that none of the stressors associated with the proposed action were likely to adversely affect ESA-listed salmonids (see Section 8.1.3 above for details). We anticipate the effects of these other stressors on non-listed salmonids in the action area would be similar to the effects on ESA-listed salmonids considered in this consultation. Therefore, adverse effects to Southern Resident killer whale primary prey species (i.e., listed and non-listed salmonids) are only anticipated to result from NWTT activities involving explosives.

Studies on the Southern Resident killer whale diet indicate that Chinook salmon is their primary prey, both in inland waters and offshore, and during all months of the year (Ford et al. 2010b; Ford et al. 2016; Hanson In prep; Hanson et al. 2010; Hilborn et al. 2012; NMFS 2019e). Non-salmonid fish species (e.g., halibut, herring, lingcod, rockfish, sole) have also been found in fecal samples, but these prey species combined likely account for less than two or three percent of the Southern Resident killer whale diet (Ford et al. 2010b). Therefore, based on the available information, non-salmonid fish species are not expected to constitute a primary component of the

Southern Resident killer whale diet. While Southern Resident killer whales may feed opportunistically on other salmonid species (i.e., coho salmon, chum salmon, steelhead), we have no information from the limited studies conducted on Southern Resident killer whale diet to indicate that species other than Chinook salmon are a major component of their offshore diet. Southern Resident killer whales travel considerable distances to locate aggregations of Chinook salmon, and the seasonal timing of the return of this high-energy preferred prey to different river systems likely influences their movements along the Northwest Pacific coast (NMFS 2019e). Since Chinook salmon are their primary prey, and many populations of Chinook salmon have declined in recent decades, we focus our analysis of the indirect effects of offshore explosives on Southern Resident killer whales via impacts to their prey on the Chinook salmon populations affected by the proposed action.

In inland waters, coho salmon have been found to account for nearly 15 percent of the Southern Resident killer whale diet, with increased consumption of coho salmon (more than 40 percent of the diet) in late summer during seasonal downward shifts in Chinook salmon abundance (Ford et al. 2016). Steelhead have also shown up in Southern Resident killer whale fecal samples taken in inland waters (Ford et al. 2016; Hanson et al. 2010). Hanson et al. (2010) noted that steelhead might be a significant Southern Resident killer whale prey item during the spring in the San Juan Islands when Chinook salmon are only just beginning to arrive. However, Puget Sound steelhead abundance in or around Crescent Harbor is likely very low, and from our analysis below (Section 8.2.3.1) only two steelhead mortalities or injuries are estimated per year as a result of explosions in Crescent Harbor. Ford et al. (2016) reported that steelhead made up less than two percent of DNA sequences from potential prey species sequenced from Southern Resident killer whales fecal samples. Similarly, less than three percent each of chum salmon and sockeye salmon were observed in fecal DNA samples collected in the summer months (May through September) in inland waters. Unpublished data from NMFS Northwest Fisheries Science Center suggest that, in addition to Chinook salmon, chum salmon are an important component of the Southern Resident killer whale inland diet from October through December (PFMC 2020). If Southern Resident killer whales are feeding on chum during these months, they would likely be targeting the large, adult fall-run populations from the non-listed Puget Sound/Georgia Strait ESU. Thus, for our inland analysis we focus on the effects of explosives on Southern Resident killer whale prey on the following salmonid populations: Puget Sound Chinook salmon ESU, Puget Sound/Georgia Strait coho salmon ESU, and Puget Sound/Georgia Strait chum salmon ESU.

While Weitkamp (2010) reported recoveries of tagged Chinook salmon from other stocks or ESUs (e.g., Fraser River, Vancouver Island, Washington Coast, and Columbia River) within Puget Sound, percent recovered from the Sound were typically very low (i.e., less than 15 percent). Chinook salmon from stocks other than the Puget Sound ESU are likely very rare (if they occur at all) in the areas affected by NWTT inland explosives. Given the small number of detonations proposed annually, and the relatively small charges (E0 and E3) that would be used,

any impacts to Chinook salmon from stocks/ESUs other than the Puget Sound Chinook salmon ESU from inland explosives would likely be minor, resulting in no significant secondary effects on Southern Resident killer whales.

Chinook salmon populations that may occur within the Southern Resident killer whale range but are not likely to overlap spatially with proposed NWTT detonations were also not included in this analysis. These include Chinook salmon stocks originating from Southeast Alaska and British Columbia, Canada. From Weitkamp (2010), none of the Chinook salmon originating from Alaska, Northern British Columbia or the West Coast of Vancouver Island were recovered within the NWTT action area. While Chinook salmon originating from the Fraser River were recovered in the offshore portion of the NWTT action area, these represented a very small fraction (less than five percent) of the total recoveries of this stock (Weitkamp 2010). As such, it is extremely unlikely that Chinook salmon originating from the Fraser River would be exposed to NWTT explosives in the offshore portion of the action area.

Thus, based on the best available information, we focus our analysis of the effects of explosives on Southern Resident killer whales via impacts to their prey on the following prey fish populations that could be exposed to NWTT explosives: all ESA-listed Chinook salmon ESUs; non-listed Chinook salmon ESUs originating from Washington, Oregon, and California; Puget Sound/Georgia Strait coho salmon ESU; and Puget Sound/Georgia Strait chum salmon ESU.

With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore, and, therefore, outside of the areas where higher densities of many Chinook salmon or coho salmon ESUs would typically be found. Mine countermeasure and neutralization testing involving the use of explosives would occur in waters three nautical miles or greater from shore within the Quinault Range Site (E4 would be used beyond three nautical miles from shore and E7 would be used beyond six nautical miles from shore in Quinault) outside the Olympic Coast National Marine Sanctuary or 12 nautical miles or greater from shore elsewhere in the NWTT offshore area. As discussed in Section 2.4.2 (*Fish Density Estimates*), the following Chinook ESUs would also be impacted by NWTT explosive activities beyond 50 nautical miles from shore: Snake River fall; Snake River spring/summer; Upper Columbia River spring; Upper Willamette River; Upper Columbia River fall; and Deschutes River.

In the inland portion of the action area, NWTT activities involving the use of explosives would only occur within two Navy EOD Training Range locations (Hood Canal and Crescent Harbor). The Navy has proposed to conduct up to six events per year (up to three E3 events and up to three E0 events at each site per year) involving mine neutralization activities in both the Hood Canal and Crescent Harbor EOD sites. The total ordnance proposed per year from these 12

events would be six E3 and 36 E0 charges. One E3 charge and up to six E0 charges would be used per event.

For our Southern Resident killer whale prey exposure analysis, we present results from our analysis of the anticipated effects of NWTT explosives on ESA-listed Chinook salmon (from Section 8.2.3 below), along with an exposure analysis for non-listed Chinook salmon ESUs (originating from Washington, Oregon and California), Puget Sound/Georgia Strait coho salmon ESU, and Puget Sound/Georgia Strait chum salmon ESU. We then discuss how the estimated impacts (i.e., mortalities, injuries, etc.) of NWTT explosives on these salmon populations would, in turn, affect Southern Resident killer whales (see *Southern Resident Killer Whale Response to Effects on Prey* below). For details on the approach used for our fish exposure analysis and anticipated fish responses to the effects of NWTT explosives see Section 8.2.3 below.

8.2.1.4.1 Prey Exposure Analysis

The use of Navy explosives would likely result in the mortality and injury of Chinook salmon and coho salmon from populations preyed upon by Southern Resident killer whales. From our analysis on the effects of explosives on ESA-listed fish below (Section 8.2.3), we estimated the maximum number (based on the proposed maximum annual explosive activity level) of salmonid mortalities and injuries as a result of Navy explosives proposed for NWTT Phase III. The results are summarized for ESA-listed Chinook salmon in Table 135 below by ESU, life stage, and hatchery (fin intact/clipped) versus natural. For Puget Sound Chinook salmon, anticipated effects are shown separately for explosions in offshore waters and explosions in inland waters (Hood Canal and Crescent Harbor EOD range sites). For all other Chinook salmon ESUs effects from Navy explosives are only anticipated to occur in the offshore portion of the action area. Table 136 shows the estimated annual number of mortalities and injuries of non-listed Chinook salmon, coho salmon, and chum salmon ESUs (included in effects analysis for Southern Resident killer whales prey) resulting from offshore NWTT explosive activities during a maximum year of explosive activity levels. For non-listed ESUs, we only estimated impacts from offshore explosives on adult salmon, as information was not available to estimate impacts on non-listed juvenile salmon.

Table 135. Estimated annual number of mortalities and injuries of ESA-listed Chinook salmon ESUs included in effects analysis for Southern Resident killer whales prey resulting from offshore NWTT explosive activities during a maximum year of explosive activity levels.

			Hatchery fish w/adipose fin intact		Hatchery fish w/adipose clip		Natural fish	
Species	Life stage	ESU - Status	Mortality	Injury	Mortality	Injury	Mortality	Injury
	Adult		-	-	0	40	0	4

			Hatchery fish w/adipose fin intact		Hatchery fish w/adipose clip		Natural fish	
Species	**Life stage**	**ESU - Status**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook salmon	Juvenile	Sac River winter run - E	-	-	10	1,199	10	1,171
	Adult	Central valley spring run - T	-	-	0	41	1	67
	Juvenile		-	-	106	13,005	38	4,649
	Adult	California coastal - T	-	-	-	-	1	127
	Juvenile		-	-	-	-	63	7,662
	Adult	Snake River fall - T	4	476	3	440	3	364
	Juvenile		116	14,197	101	12,319	28	3,436
	Adult	Snake River spring/summer - T	0	14	-	77	3	416
	Juvenile		31	3,845	180	22,089	41	4,997
	Adult[1]	Lower Columbia River - T	2	273	-	-	2	208
	Juvenile		44	5,421	1,441	176,612	540	66,159
	Adult[1]	Upper Willamette River - T	9	1,033	-	-	3	335
	Juvenile		0	1	210	25,759	54	6,629
	Adult	Upper Columbia River spring - E	1	109	2	202	1	93
	Juvenile		15	1,828	25	3,084	19	2,325
	Adult[1]	Puget Sound – T Offshore Effects	1	68	-	-	1	98
	Juvenile		221	27,075	1,103	135,159	92	11,302
	Adult	Puget Sound – T Inland Effects	-	-	1	84	1*	26
	Juvenile		92	20,652	109	6,748	134	24,745

[1] Includes mortality and injury estimates for fin-clipped and intact adipose fin hatchery fish for the life stage of this ESU.

*Estimated mortality or injury of one adult Puget Sound Chinook salmon in inland waters (Crescent Harbor) may be of natural or hatchery origin.

Table 136. Estimated annual number of mortalities and injuries of non-listed Chinook salmon, coho salmon, and chum salmon ESUs included in effects analysis for Southern Resident killer whales prey resulting from offshore NWTT explosive activities during a maximum year of explosive activity levels.

		Hatchery fish		Natural fish	
Species	**ESU**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook salmon	Central Valley Fall-run	6	693	10	1,229
	Sacramento River Late Fall-run	0	0	1	143
	Klamath River Chinook salmon Fall-run	3	408	14	1,655
	Oregon Coast	0	0	6	738
	Upper Columbia River Fall-run	5	527	70	8,599
	Middle Columbia River Spring-run	0	0	5	560
	Washington coast	1	101	1	157
	Southern Oregon/Northern California Coast (SONCC)	3	341	2	261
	Deschutes River	0	6	2	298
Coho salmon	Puget Sound/Georgia Strait	40	4,889	49	6,023
Chum salmon	Puget Sound/Georgia Strait	0	35	1	169

As discussed above, while Chinook salmon was our primary focus for analyzing effects of explosives on Southern Resident killer whale prey availability, limited diet data suggest that Puget Sound/Georgia Strait coho salmon and chum salmon may also be an important component

of the inland diet, particularly during certain times of year when Chinook salmon availability is limited (Ford et al. 2016). We do not have area specific density information for coho or chum salmon at the Crescent Harbor or Hood Canal EOD sites. In Crescent Harbor, adult coho salmon and chum salmon density is likely much lower than would be expected in some other areas and habitat types in Puget Sound (e.g., river mouths, deltas, and estuaries of the Skagit and Stillaguamish Rivers) because there are no documented spawning rivers for these ESUs in the immediate vicinity of this EOD range. While adult coho and chum salmon are unlikely to stage in large numbers in or near Crescent Harbor, it is possible that very low numbers of migrating adults could occur within the range to effects for underwater detonations for very short periods of time (though Crescent Harbor is not a direct migratory route from the ocean to any coho salmon or chum salmon spawning rivers).

Given the very low densities of adults that may occur in the area, the short amount of time any individuals would be likely to spend in Crescent Harbor, and the low number of explosion events that will occur throughout the year, we do not expect more than one adult Puget Sound/Georgia Strait ESU coho salmon and one adult Puget Sound/Georgia Strait ESU chum salmon may be injured or killed each year from Navy explosive activities at Crescent Harbor. Higher densities of juvenile coho and chum salmon, as compared to adults, may be expected in or near the Crescent Harbor and Hood Canal EOD range sites, although we have no specific density information for these areas with which to quantify juvenile exposures.

There are several populations of Puget Sound/Georgia Strait coho salmon throughout Hood Canal including the Skokomish River watershed, Mid Hood Canal region, and Southern Hood Canal area (Ford 2011a). Hood Canal coho salmon populations typically spawn mid-September to late October (Commission 2003). Juvenile coho salmon emigrating from or adults immigrating to these watersheds would likely pass by the Hood Canal EOD site. Therefore, although we cannot quantify exposures, we would expect higher densities of adult coho salmon around the Hood Canal EOD site as compared to Crescent Harbor. Similarly, there are several populations of fall-run Puget Sound/Georgia Strait chum salmon throughout Hood Canal including Northeast Hood Canal, Southeast Hood Canal, and West Hood Canal. We focus on fall-run chum since, as noted above, chum salmon may be an important component of the Southern Resident killer whale inland diet from October through December. Hood Canal fall-run chum salmon spawn predominantly from November through December. Juvenile chum salmon emigrating from or adults immigrating to these watersheds would likely pass by the Hood Canal EOD site. Therefore, although we cannot quantify exposures, we would expect higher densities of adult chum salmon around the Hood Canal EOD site as compared to Crescent Harbor.

While a larger number of adult coho and chum salmon would likely be exposed to the effects (e.g., mortality and injury) of Navy explosives in Hood Canal, there are no confirmed sightings of Southern Resident killer whales inside Hood Canal (NMFS 2019e). If, based on Ford et al. (2016), Puget Sound/Georgia Strait coho salmon are an important inland prey item in the late

summer, it is more likely that Southern Resident killer whales are feeding on coho salmon stocks of this ESU from areas other than Hood Canal (e.g., stocks originating in the Eastern Olympic Peninsula, Georgia Strait, and British Columbia mainland). This is consistent with the fecal sampling locations in the Ford et al. (2016) study, which were located in the Strait of Juan de Fuca, Georgia Strait, and around the San Juan Islands (i.e., no sampling in Puget Sound). In addition, Ford et al. (2016) found that coho salmon only accounted for a large proportion of the Southern Resident killer whale diet in the late summer. As a mitigation measure to minimize impacts to salmon in Hood Canal, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (> 0.5-2.5 pounds), and only conduct training with one ounce or less (E0) charges during August, September, and October. This mitigation should further minimize the impact of explosives on adult coho salmon in Hood Canal during the time of year when Southern Resident killer whales may be feeding on this population. Similarly, since Southern Resident killer whales are rare inside Hood Canal, we would expect that they are predominantly feeding on fall-run Puget Sound/Georgia Strait adult chum salmon from populations outside of Hood Canal, and therefore, not within the range to effects from Navy explosives.

Although we could not quantify, we expect larger numbers of juvenile coho and chum salmon (as compared to adults) would be exposed to the effects of explosives at both the Crescent Harbor and Hood Canal EOD range sites and offshore. However, as with Chinook salmon, we would expect Southern Resident killer whales to prefer large, adult coho and chum salmon, which provide a much high energy content compared to juveniles. In addition, we anticipate the number of juvenile coho and chum salmon exposed to the effects of Navy explosives to make up a very small proportion of the total juvenile population of the large and expansive Puget Sound/Georgia Strait ESUs of these species. These non-listed ESUs cover a wide range, which includes coho salmon from drainages of Puget Sound, the eastern Olympic Peninsula, and the Georgia Strait from the eastern side of Vancouver Island and the British Columbia mainland.

8.2.1.4.2 Southern Resident Killer Whale Response to Effects on Prey

Southern Resident killer whales are large mammals requiring abundant food sources to sustain metabolic processes throughout the year (NMFS 2019e). Prey availability changes seasonally, and Southern Resident killer whales appear to depend on different prey species/populations and habitats throughout the year. The seasonal timing of salmon returns to different river systems likely influences their movements (NMFS 2019e). Whales may travel significant distances to locate prey aggregations sufficient to support their numbers. Diet data indicate that Chinook salmon is the primary prey of Southern Resident killer whales across all seasons and areas (Ford et al. 2016; Hanson In prep; Hanson et al. 2010; NMFS 2019e). Chinook salmon have the highest value of total energy content compared to other salmonids because of their larger body size and higher energy density (kcal/kg) (O'Neill et al. 2014). The mortality of adult Chinook salmon can have immediate impacts on the availability of prey for Southern Resident killer

whales, as well as the reproductive potential of the salmon stock and population. In addition, some proportion of injuries resulting from explosives would likely have fitness consequences for individual fish exposed, which could affect their survival or future reproductive potential. Although diet data in offshore waters is sparse, juvenile salmon are not anticipated to be a preferred target prey item for Southern Resident killer whales, which likely seek out adult Chinook salmon (Hilborn et al. 2012) due to the higher nutritional value of the larger fish (O'Neill et al. 2014). While juvenile Chinook salmon mortality may not result in the same immediate impacts on prey availability as adult mortality, the loss of juveniles represents the loss of future adult prey availability, as well as future reproductive potential within the population.

The daily prey energy requirements for individual females and males range from 41,376 to 269,458 kcal/day and 41,376 to 217,775 kcal/day, respectively (Noren 2011). The daily prey energy requirements can be converted to the number of fish required each year if the caloric densities of the fish (kcal/fish) consumed are known. Noren (2011) estimated the daily consumption rate of a population with 82 individuals over the age of one that consumes solely Chinook salmon would consume 289,131–347,000 fish/year by assuming the caloric density of Chinook was 16,386 kcal/fish (i.e., the average value for adults from Fraser River). Williams et al. (2011b) and Chasco et al. (2017a) modeled annual SRKW prey requirements and found that the whole population requires approximately 211,000 to 364,100 and 190,000 to 260,000 Chinook salmon per year, respectively.

From Table 135 and Table 136 above, the estimated total number of adult Chinook salmon killed annually (all ESUs, including ESA-listed and non-listed) as a result of the proposed action, based on a maximum annual activity level of explosive use, is 168 fish. We also estimated up to 90 mortalities of adult Puget Sound/Georgia Strait coho salmon and two mortalities of adult Puget Sound/Georgia Strait chum salmon based on a maximum annual activity level of explosive use (based on offshore and inland impacts combined). In total, this anticipated number of mortalities in a maximum year represents a small fraction of the total number of adult salmon needed per year by an individual Southern Resident killer whale to maintain their energy requirements. Spread out over the Southern Resident killer whale DPS, the impacts of adult salmon mortalities as a result of the proposed action on the nutritional needs of the population would likely be extremely small.

A considerably larger total estimated number of adult Chinook salmon would likely be injured than killed as a result of NWTT explosives (i.e., approximately 12,647 fish annually during a maximum year of explosive activity, assuming no individual fish is injured more than once per year). More than one-half of the estimated injuries (7,654) are from the non-listed Upper Columbia River fall ESU. We also estimated 10,912 injuries of the non-listed adult Puget Sound/Georgia Strait coho salmon ESU and 204 injuries of the non-listed adult Puget Sound/Georgia Strait chum salmon ESU during a maximum year of explosive activity levels. We anticipate that while some injuries from explosives would be relatively minor, others may impact important life functions including predator avoidance, foraging, and migrations. The duration

and magnitude of such impacts would likely depend on a number of factors including the severity of the injury, the baseline health of the individual prior to the injury, the size of the fish, and the timing of the injury (i.e., seasonal timing and life cycle timing). Sublethally injured adult salmon would still be available as potential prey for killer whales and, depending on the nature of the injury, could be more easily captured by killer whales, as well as other predators. Although we cannot quantify, based on the available information we expect that some proportion of Chinook salmon injuries from explosives would likely result in fitness consequences, thus affecting the survival and reproductive potential of the individual fish affected. For a detailed discussion of the anticipated response of fish to injuries from explosives see Section 8.2.3.2.

Compared to adult impacts, a larger total number of juvenile Chinook salmon would likely be affected by explosives as a result of the proposed action (Table 135 and Table 136). This reflects the higher densities of juveniles, as compared to adults, in the areas where explosives would occur. Since Southern Resident killer whales generally target and feed on the larger, adult salmon, impacts to juveniles would likely result in only a minor reduction in the immediate availability of prey for Southern Resident killer whales. However, juvenile salmon mortality and injury could have greater impacts on the future availability of prey for Southern Resident killer whales, as well as the future reproductive potential of the population.

To further help evaluate whether the anticipated effects on Chinook salmon ESUs are likely to result in adverse effects to Southern Resident killer whales, we calculated the percent of each affected Chinook salmon ESU that would be killed or injured by NWTT detonations (Table 137). For injuries, it should be noted that since some individual fish could be exposed more than once per year (and other fish not exposed at all), the numbers in Table 137 are an approximation of the proportion of the population that may be injured annually. For the Puget Sound ESU, the calculated percent includes mortalities and injuries resulting from both offshore and inland explosives (for all other Chinook salmon ESUs only offshore effects are anticipated). Percentages were derived from population abundances presented in NMFS (2020c). As was done for our salmonid density approach (Section 2.4.2), a correction factor was applied for adults that multiplied the average total number of years at sea for each ESU by the total estimated number of returning spawners.

As shown in Table 137, an extremely small percentage of each Chinook salmon ESU would likely be killed from NWTT explosive activities during a maximum year of explosive activity levels (i.e., less than 0.02 percent across all ESUs, life stages, and hatchery/natural origin). Similarly, for the non-listed Puget Sound/Georgia Strait coho salmon and chum salmon ESUs, less than 0.01 percent of each population would likely be killed from NWTT explosive activities during a maximum year of explosive activity levels (i.e., percent across both ESUs, life stages, and hatchery/natural origin). While the loss of an individual fish represents the loss of future reproductive potential, given the small anticipated mortality rates, it is unlikely that salmon mortality from NWTT explosives would have an appreciable impact on the future reproductive potential of the salmon populations preyed upon by Southern Resident killer whales.

The percentages of each Chinook salmon population that would likely be injured during a maximum year of NWTT explosive activity levels range from less than 0.1 percent to nearly 1.3 percent depending on the particular ESUs and life stage (Table 137). We also estimate that about 1.1 percent of the Puget Sound/Georgia Strait coho salmon ESU and less than 0.02 percent of the Puget Sound/Georgia Strait chum salmon ESU would likely be injured from NWTT explosives during a maximum year of explosive activity levels. As noted above, while we expect some of these sublethal injuries would be relatively minor in nature, others would likely result in fitness consequences, thus affecting the survival and reproductive potential of the individual fish affected.

It is important to note that our range to effects for fish injury are based on the more conservative acoustic injury criterion of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely resulted in conservatively high estimates of salmonid fish injury from Navy explosives.

The relative importance of the different Chinook salmon ESUs (shown in Table 137) to the overall diet and fitness of Southern Resident killer whale's is highly variable, and there is likely a seasonal component as well (i.e., some ESUs are more important during particular times of year). As described in Section 6.2.4, NMFS and the WDFW developed a framework to identify Chinook salmon stocks of particular importance to Southern Resident killer whales (WDFW 2018). Based on this conceptual model, some of the highest priority Chinook salmon stocks for Southern Resident killer whales originate from Puget Sound, the Columbia River, and the Snake River. From our effects analysis, several of the highest priority Chinook salmon ESUs also had some of the highest estimated injury rates as a proportion of the population size. For the Snake River fall run, Snake River spring/summer, Upper Columbia fall run and Upper Columbia River spring run Chinook salmon ESUs, we estimate adult injury rates of between 1.0 and 1.2 percent during a maximum year of NWTT explosive activity. For the Puget Sound Chinook salmon ESU, which is impacted by both offshore and inland explosives, we estimate a juvenile injury rates of nearly 1.2 percent for natural origin fish during a maximum year of NWTT explosive activity, although adult and hatchery injury rates were somewhat lower (Table 137). We also estimate an adult injury rate of 1.1. percent for the Puget Sound/Georgia Strait coho salmon ESU, which, as noted above, may be particularly important to the Southern Resident killer whale diet in the late summer during seasonal downward shifts in Chinook salmon abundance (Ford et al. 2016).

Table 137. Anticipated effects of NWTT explosives on Chinook salmon ESUs expressed as a percent of the population (i.e., hatchery[1] and natural populations) killed and injured annually during a maximum year of explosive activity levels.

Species	Life stage	ESU	Hatchery Fish Percent of Population Killed	Hatchery Fish Percent of Population Injured	Natural Fish Percent of Population Killed	Natural Fish Percent of Population Injured
ESA-listed Chinook salmon	Adult	Sac River winter run - E	0.0000%	0.5974%	0.0000%	0.6349%
	Juvenile		0.0050%	0.5995%	0.0051%	0.5994%
	Adult	Central valley spring run - T	0.0000%	0.6013%	0.0089%	0.5992%
	Juvenile		0.0049%	0.5995%	0.0049%	0.5995%
	Adult	California coastal - T	0.0000%	0.0000%	0.0047%	0.6018%
	Juvenile		0.0000%	0.0000%	0.0049%	0.5995%
	Adult	Snake River fall - T	0.0080%	1.0507%	0.0096%	1.1726%
	Juvenile		0.0041%	0.4960%	0.0040%	0.4959%
	Adult	Snake River spring/summer - T	0.0000%	1.0802%	0.0089%	1.0833%
	Juvenile		0.0040%	0.4960%	0.0041%	0.4960%
	Adult	Lower Columbia River - T	0.0017%	0.2358%	0.0023%	0.2353%
	Juvenile		0.0046%	0.5633%	0.0046%	0.5633%
	Adult	Upper Willamette River - T	0.0095%	1.0940%	0.0090%	1.0945%
	Juvenile		0.0045%	0.5470%	0.0045%	0.5470%
	Adult	Upper Columbia River spring - E	0.0104%	1.0810%	0.0089%	1.0833%
	Juvenile		0.0040%	0.4960%	0.0041%	0.4959%
	Adult	Puget Sound - T	0.0043%	0.3260%	0.0021%	0.1842%
	Juvenile		0.0035%	0.4353%	0.0074%	1.1876%
Non-listed Chinook salmon	Adult	Central Valley Fall-run	0.0049%	0.5995%	0.0049%	0.5995%
	Adult	Sacramento River Late Fall-run	0.0000%	0.0000%	0.0049%	0.5995%
	Adult	Klamath River	0.0049%	0.5995%	0.0049%	0.5995%

Species	Life stage	ESU	Hatchery Fish Percent of Population Killed	Hatchery Fish Percent of Population Injured	Natural Fish Percent of Population Killed	Natural Fish Percent of Population Injured
		Chinook salmon Fall-run				
	Adult	Oregon Coast	0.0000%	0.0000%	0.0011%	0.1406%
	Adult	Upper Columbia River Fall-run	0.0103%	1.0843%	0.0088%	1.0835%
	Adult	Middle Columbia River Spring-run	0.0000%	0.0000%	0.0020%	0.2391%
	Adult	Washington coast	0.0012%	0.1460%	0.0012%	0.1460%
	Adult	Southern Oregon/Northern California Coast	0.0104%	1.2755%	0.0104%	1.2755%
	Adult	Deschutes	0.0000%	1.1696%	0.0072%	1.0800%

[1] Hatchery includes fish with adipose fin intact and fish with adipose clip.

8.2.1.4.3 Summary - Effects of Explosives on Southern Resident Killer Whales via Impacts to their Prey

The use of explosives as part of the proposed action would likely result in the mortality and injury of fish species preyed upon by Southern Resident killer whales. Southern Resident killer whale diet studies indicate that Chinook salmon is their primary prey both in inland waters and offshore, and during all months of the year (Ford et al. 2010b; Ford et al. 2016; Hanson In prep; Hanson et al. 2010; Hilborn et al. 2012; NMFS 2019e). In inland waters, coho salmon have also been found to account for a large proportion of the Southern Resident killer whale diet, particularly in late summer during seasonal downward shifts in Chinook salmon abundance (Ford et al. 2016). Fall-run chum salmon may also be an important component of the inland diet from October through December. While Southern Resident killer whales likely feed opportunistically on other salmonid species (i.e., steelhead and sockeye salmon), we have no information from the limited available diet studies to indicate that populations other than

Chinook salmon (all ESUs) and Puget Sound/Georgia Strait coho and chum salmon (in inland waters only) are a primary component of their diet. Therefore, while NWTT explosives may impact other salmonid fish species (i.e., besides Chinook salmon and Puget Sound/Georgia Strait coho and chum salmon), such impacts are not likely to adversely affect Southern Resident killer whales since these fish species are not likely a major component of their diet. Non-salmonid fish species (e.g., halibut, herring, lingcod, rockfish, sole) have also been found in fecal samples, but these prey species combined likely account for less than two to three percent of the Southern Resident killer whale diet (Ford et al. 2010b). Therefore, while NWTT explosives may impact non-salmonid fish species, such impacts are not likely to adversely affect Southern Resident killer whales since these fish species are not likely a major component of their diet.

With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore, outside areas where several populations of Chinook, coho, or chum salmon would typically be found. Mine countermeasure and neutralization testing involving the use of explosives would occur outside the Olympic Coast National Marine Sanctuary in waters three nautical miles or greater from shore within the Quinault Range Site or in waters 12 nautical miles or greater from shore elsewhere in the NWTT offshore area. Additionally, mine countermeasure and neutralization testing using explosives within bin E7 will be no closer to shore than six nautical miles in the Quinault Range Site.

The mortality of Chinook salmon and Puget Sound/Georgia Strait coho and chum salmon could have immediate impacts on Southern Resident killer whale prey availability, as well as immediate and future impacts on the reproductive potential of the population. Based on our quantitative analysis, the estimated annual number of adult Chinook salmon (all ESUs) and Puget Sound/Georgia Strait coho and chum salmon killed during a maximum year of NWTT explosive activity levels represents an extremely small fraction of the total number of salmon needed per year by an individual Southern Resident killer whale to maintain their energy requirements. Since the effects of reduced prey availability would likely be spread out over the population as a whole (or some proportion of the population), the impacts of adult Chinook salmon, chum salmon, and coho salmon mortalities as a result of NWTT explosives on the nutritional needs of the Southern Resident killer whale DPS are anticipated to be minor. While the loss of an individual fish represents the loss of future reproductive potential, given the small anticipated mortality rates (i.e., less than 0.02 percent of each population), it is unlikely that salmon mortality from NWTT explosives would have an appreciable impact on the future reproductive potential of the salmon populations preyed upon by Southern Resident killer whales.

From our quantitative analysis, we expect a significantly larger number of salmon would be injured (as compared to killed) as a result of NWTT explosives. The proportion of each salmon population that would likely be injured during a maximum year of NWTT explosive activity

levels range from around 0.1 percent to nearly 1.3 percent, depending on the particular ESUs and life stage. From our effects analysis, several of the highest priority Chinook salmon ESUs for Southern Resident killer whale conservation, also had some of the highest estimated injury rates as a proportion of the population size among the ESUs analyzed. These include the Snake River fall run ESU, Snake River spring/summer ESU, Upper Columbia River fall run ESU and Upper Columbia River spring run ESU (all with estimated adult injury rates of between 1.7 and 1.9 percent), and Puget Sound Chinook salmon ESU (estimated adult injury rates of 0.4 percent for hatchery origin fish and 1.2 percent for natural origin fish). We also estimate that about one percent of the adult Puget Sound/Georgia Strait coho salmon ESU, which may be particularly important to the Southern Resident killer whale diet in the late summer (Ford et al. 2016), would also likely be injured during a maximum year of NWTT explosive activity.

We anticipate that while some injuries from explosives would be relatively minor, others may impact important life functions including predator avoidance, foraging, and migrations. The duration and magnitude of such impacts would likely depend on a number of factors including the severity of the injury, the baseline health of the individual prior to the injury, the size of the fish, and the timing of the injury (i.e., seasonal timing and life cycle timing). Although we cannot quantify based on the available information, we expect that some proportion of salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected. Given the estimated number and proportional rate of injuries anticipated, particularly of high priority Chinook salmon ESUs, salmonid injuries resulting from the proposed action would likely have impacts on Southern Resident killer whale prey availability, as well as impacts on the future reproductive potential of important salmon prey populations. Reduced prey availability that results in whales spending more time and energy searching for prey can lead to nutritional stress. Therefore, we conclude that Southern Resident killer whales would likely be adversely affected due to the anticipated reduction in the availability of their salmonid prey as a result of the proposed action.

As noted previously (Section 2.3.2), our range to effects for fish injury are based on the more conservative acoustic injury criterion of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors, and others described in Section 8.2.3.1 below, likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

We also note that since the estimated impacts presented above are based on a maximum annual activity level of offshore explosive use, in an average year the anticipated impacts would be somewhat lower. For the mine countermeasure neutralization activity, the annual average number of explosives used would be less than half that used in a maximum year. The Navy would also continue to implement procedural mitigation measures to avoid or reduce potential impacts on marine mammals from activities involving explosives. The Navy has proposed

mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months of October through June (see Section 3.5.1.9 for details). Since this timeframe includes months when peak densities of many Chinook salmon ESUs would be expected in the action area, this mitigation measure would likely result in reduced impacts to several of the Chinook salmon ESUs preyed upon by Southern Resident killer whales. Southern Residents are also more likely to occur in the offshore portion of the action area from October through June as compared to July through September, when they are more often seen in the inland waters. Mitigation measures proposed by the Navy would likely reduce the impacts of inland explosives on salmonid populations preyed upon by Southern Resident killer whales in the inland portion of their range. As a mitigation measure to minimize impacts to salmon in Hood Canal, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (less than 0.5-2.5 pounds), and only conduct training with one ounce or less (E0) charges during August, September, and October. The Navy will also continue to implement procedural mitigation to avoid or reduce potential impacts on marine mammals from explosive mine countermeasure and neutralization activities. The use of Lookouts should reduce the likelihood that Southern Resident killer whales would be actively feeding on salmon in areas where underwater explosives are being used.

8.2.1.5 Vessel Strike

Vessel strikes from commercial, recreational, and military vessels are known to affect large whales and have resulted in serious injury and occasional fatalities to cetaceans (Berman-Kowalewski et al. 2010; Calambokidis 2012; Douglas et al. 2008; Laggner 2009; Lammers et al. 2003). Records of collisions date back to the early 17th century, and the worldwide number of collisions appears to have increased steadily during recent decades (Laist et al. 2001; Ritter 2012).

Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals often, but not always (e.g., McKenna et al. 2015), engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Amaral and Carlson 2005b; Au and Green 2000; Bain et al. 2006a; Bauer 1986; Bejder et al. 1999; Bejder and Lusseau. 2008; Bejder et al. 2009; Bryant et al. 1984; Corkeron 1995; Erbe 2002; Félix 2001; Goodwin and Cotton 2004; Lemon et al. 2006; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2003; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002d; Wursig et al. 1998). Several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). Water disturbance may also be a factor. These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators. Avoidance behavior is expected to be even stronger when the Navy is conducting training or testing activities (e.g., when active sonar or explosives are in use).

The most vulnerable marine mammals are those that spend extended periods of time at the surface in order to restore oxygen levels within their tissues after deep dives (e.g., sperm whales). In addition, some baleen whales seem generally unresponsive to vessel sound, making them more susceptible to vessel collisions (Nowacek et al. 2004). These species are primarily large, slow moving whales.

Some researchers have suggested the relative risk of a vessel strike can be assessed as a function of animal density and the magnitude of vessel traffic (e.g., Fonnesbeck et al. 2008; Vanderlaan et al. 2008). Differences among vessel types also influence the probability of a vessel strike. The ability of any ship to detect a marine mammal and avoid a collision depends on a variety of factors, including environmental conditions, ship design, size, speed, and personnel, as well as the behavior of the animal. Vessel speed, size, and mass are all important factors in determining if injury or death of a marine mammal is likely due to a vessel strike. For large vessels, speed and angle of approach can influence the severity of a strike. For example, Vanderlaan and Taggart (2007) found that between vessel speeds of 8.6 and 15 knots, the probability that a vessel strike is lethal increases from 0.21 to 0.79. Large whales also do not have to be at the water's surface to be struck. Silber et al. (2010) found when a whale is below the surface (about one to two times the vessel draft), under certain circumstances (vessel speed and location of the whale relative to the ship's centerline), there is likely to be a pronounced propeller suction effect. This suction effect may draw the whale into the hull of the ship, increasing the probability of propeller strikes.

Large Navy vessels (greater than 18 meters in length) within the offshore portion of the action area operate differently from commercial vessels in ways important to the prevention of whale collisions. For example, the average speed of large Navy ships ranges between 10 and 15 knots, and submarines generally operate at speeds in the range of 8 and 13 knots, while a few specialized vessels can travel at faster speeds. By comparison, this is slower than most commercial vessels where normal design speed for a container ship is typically 24 knots (Bonney and Leach 2010). Even given the advent of "slow steaming" by commercial vessels in recent years due to fuel prices (Barnard 2016; Maloni et al. 2013), this generally reduces the design speed by only a few knots, given that 21 knots would be considered slow, 18 knots is considered "extra slow," and 15 knots is considered "super slow" (Bonney and Leach 2010). Small Navy craft (less than 50 feet in length), have much more variable speeds (0–50 knots or more, depending on the mission). While these speeds are considered averages and representative of most events, some Navy vessels need to operate outside of these parameters during certain situations. Differences between most Navy ships and commercial ships also include the following disparities:

- The Navy has several standard operating procedures for vessel safety that could result in a secondary benefit to marine mammals through a reduction in the potential for vessel strike, as described in Section 3.4.1 *Vessel Safety*.

- Many Navy ships have their bridges positioned closer to the bow, offering good visibility ahead of the ship.
- There are often aircraft associated with the training or testing activity, which can detect marine mammals in the vicinity or ahead of a vessel's present course.
- Navy ships are generally much more maneuverable than commercial merchant vessels if marine mammals are spotted and it becomes necessary to change direction.
- Navy ships operate at the slowest speed possible consistent with either transit needs, or training or testing need. While minimum speed is intended as a fuel conservation measure particular to a certain ship class, secondary benefits include being better able to spot and avoid objects in the water, including marine mammals.
- In many cases, Navy ships will likely move randomly or with a specific pattern within a sub-area of the action area for a period of time, from one day to two weeks, as compared to straight line point-to-point commercial shipping.
- Navy overall crew size is much larger than merchant ships, allowing for more potential observers on the bridge.
- When submerged, submarines are generally slow moving (to avoid detection), and therefore marine mammals at depth with a submarine are likely able to avoid collision with the submarine. When a submarine is transiting on the surface, there are Lookouts serving the same function as they do on surface ships.
- The Navy will implement mitigation to avoid potential impacts from vessel strikes on marine mammals. Mitigation includes training Lookouts and watch personnel with the Marine Species Awareness Training (which provides information on sighting cues, visual observation tools and techniques, and sighting notification procedures) (see Section 3.5.1.1), and requiring vessels to maneuver to maintain a specified distance from marine mammals during vessel movements (see Section 0).

Data from the ports of Vancouver, British Columbia; Seattle, Washington; and Tacoma, Washington indicated there were in excess of 7,000 commercial vessel transits in 2017 associated with visits to just those ports (The Northwest Seaport Alliance 2018; Vancouver Fraser Port Authority 2017). This number of vessel transits does not account for other vessel traffic in the Strait of Juan de Fuca or Puget Sound resulting from commercial ferries[20], tourist vessels, or recreational vessels. Additional commercial traffic in the action area also includes vessels transiting offshore along the Pacific coast, bypassing ports in Canada and Washington; traffic associated with ports to the south along the coast of Washington and in Oregon; and vessel traffic in Southeast Alaska (Nuka Research & Planning Group 2012). This level of

[20] 23 Washington state ferries make almost 450 transits per day, equivalent to approximately 164,000 transits per year which contribute to underwater ambient noise in the Inland Waters Washington State Department of Transportation. 2018. Washington State Ferries Federal Transit Administration Asset Management Plan, Olympia, WA..

commercial vessel traffic for the ports of Vancouver, Seattle, and Tacoma is approximately the same as in 2015.

In the action area, the existing marine environment is dominated by non-Navy vessel traffic given the Navy has, in total, the following homeported operational vessels: 2 aircraft carriers, 6 destroyers, 14 submarines, and 22 smaller security vessels with a combined annual total of 241 Navy vessel transits. Activities involving military vessel movement would be widely dispersed throughout the action area. For details on Navy vessel types used and vessel activity levels in the NWTT action area see Section 5.3.1.

8.2.1.5.1 Exposure Analysis

Navy vessel strike records have been kept since 1995, and since 1995 there have been two recorded strikes of large whales by Navy vessels (or vessels being operated on behalf of the Navy) in the action area. Neither strike was associated with training or testing activities. The first strike occurred in 2012 by a Navy destroyer off the southern coast of Oregon while in transit to San Diego. The whale was suspected to be a minke whale due to the appearance and size (25 feet, dark with white belly), however the Navy could not rule out the possibility that it was a juvenile fin whale. The whale was observed swimming after the strike and no blood or injury was sighted. The second strike occurred in 2016 by a U.S. Coast Guard cutter operating on behalf of the Navy as part of a Maritime Security Operation escort vessel in the Strait of Juan de Fuca. The whale was positively identified as a humpback whale. It was observed for 10 minutes post-collision and appeared normal at the surface. There was no blood observed in the water and the whale subsequently swam away.

In order to account for the potential risk from any vessel movement within the NWTT action area within the seven-year LOA period in particular, the Navy requested incidental takes based on probabilities derived from a Poisson distribution using: 1) ship strike data between 2009-2018 in the action area (the time period from when current mitigation measures to reduce the likelihood of vessel strikes were instituted until the Navy conducted the analysis for the Navy's application), 2) historical at-sea days in the action area from 2009-2018, and 3) estimated potential at-sea days for the period from 2020 to 2027 covered by the requested regulations. This distribution predicted the probabilities of a specific number of strikes (n=0, 1, 2, etc.) over the period from 2018 to 2025. The analysis for the period of 2020 to 2027 is described in detail in Chapter 6.6 (Vessel Strike Analysis) of the Navy's rulemaking/LOA application (Navy 2019e).

For the same reasons listed above, describing why a Navy vessel strike is less likely than a commercial vessel strike, it is highly unlikely that a Navy vessel would strike a whale, dolphin, porpoise, or pinniped without detecting it and, accordingly, NMFS is confident that the Navy's reported strikes are accurate and appropriate for use in the analysis. Specifically, Navy ships have multiple Lookouts, including on the forward part of the ship that can visually detect a hit animal, in the unlikely event ship personnel do not feel the strike. Unlike the situation for non-Navy ships engaged in commercial activities, NMFS and the Navy have no evidence that the

Navy has struck a whale and not detected it. While there are new activities involving the use of vessels, such as Propulsion Testing, the projected Navy vessel use has not significantly changed over time and is not projected to significantly change. Navy's strict internal procedures and mitigation requirements include reporting of any vessel strikes of marine mammals, and the Navy's discipline, extensive training (not only for detecting marine mammals, but for detecting and reporting any potential navigational obstruction), and strict chain of command give NMFS a high level of confidence that all strikes actually get reported.

The Navy used the two reported whale strikes (mentioned above) in their calculations to determine the number of large whale strikes likely to result from their activities and evaluated data beginning in 2009. The Navy's Marine Species Awareness Training was first used in 2006 and was fully integrated across the Navy in 2009, which is why the Navy uses 2009 as the date to begin the analysis. The adoption of additional mitigation measures to address ship strike also began in 2009, and will remain in place along with additional mitigation measures during the seven years of the NMFS Permit Division's rule. The probability analysis concluded that there was a 26 percent chance that zero whales would be struck by Navy vessels over the seven-year period, and a 35, 24, 11, and 4 percent chance that one, two, three, or four whales, respectively, would be struck over the seven-year period (with a 74 percent chance that at least one whale would be struck over the seven-year period). The Navy estimates, and NMFS agrees with reasonable certainty, that up to three large whale vessel strikes resulting in serious injury or mortality would occur as a result of NWTT activities within the action area every seven years.

Smaller ESA-listed marine mammals, including Southern Resident killer whales and Guadalupe fur seals, could also potentially be impacted by vessel strike in general. In 2005, a Southern Resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. In 2006, an adult male Southern Resident killer whale, L98, was killed in a collision with a tug boat. In fall 2016 another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that the whale died of blunt force trauma to the head: "the animal had injuries consistent with blunt trauma to the dorsal side, and a hematoma indicating that it was alive at the time of injury and would have survived the initial trauma for a period of time prior to death" (Fisheries and Oceans Canada 2019). The injuries described are consistent with those incurred during a vessel strike. To date, there have been no known Navy vessel strikes involving a Southern Resident killer whale. We could not find any information regarding vessel strikes (by Navy or non-Navy vessels) of Guadalupe fur seals in the action area.

In addition to the reasons listed above that make large whale strikes by Navy vessels a rare event (more maneuverable ships, larger crew, etc.), there are additional reasons that make Navy vessel strike of a Southern Resident killer whale or Guadalupe fur seal extremely unlikely. Their much smaller size and greater maneuverability make these species less susceptible to a vessel strike in general. This is supported by worldwide ship strike records that show strikes of smaller, more agile marine mammals from the shipping sector and larger vessels are extremely rare. Although

Southern Resident killer whales have been struck in the past (three since 2005), these strikes did not involve Navy vessels, which make up an extremely small proportion of vessel traffic in the action area. Given this rate of past vessel strikes, the small number of Navy vessels in the action area compared to other vessels, and the fact that a Navy vessel strike involving a Southern Resident killer whale has not been recorded in the past, it is extremely unlikely that future strikes of Southern Resident killer whales would involve Navy vessels. In addition, the Navy will implement procedural mitigation to avoid or reduce the potential for vessel strikes of marine mammals (see Table 36 for details), as well as additional requirements for activities within the Puget Sound and Strait of Juan de Fuca Mitigation Area (see Table 42 for details). In summary, for the reasons discussed above, the likelihood of a Southern Resident killer whale or Guadalupe fur seal Navy vessel strike as a result of the proposed action is extremely unlikely, and thus considered discountable. Based on this information, NMFS concurs with the Navy's analysis that estimated incidental take by vessel strike of three large whales only (i.e., no Southern Resident killer whales or Guadalupe fur seals) over the course of the seven-year MMPA regulations from training and testing activities.

Taking into account the available information regarding how many of any given stock/DPS would be struck and therefore should be authorized for take, NMFS considered three factors in addition to those considered in the Navy's request: 1) the relative likelihood of hitting one stock/DPS versus another based on available strike data from all vessel types as denoted in the SARs, 2) whether the Navy has ever definitively struck an individual from a particular species or stock/DPS in the action area, and if so, how many times, and 3) whether there are records that an individual from a particular species or stock/DPS has been struck by any vessel in the action area, and if so, how many times (based on ship strike records provided by the NMFS West Coast Region in February 2020). To address number one above, NMFS compiled information from NMFS' SARs on detected annual rates of large whale serious injury or mortality (M/SI) from vessel collisions (Table 138). The annual rates of large whale serious injury or mortality from vessel collisions from the SARs help inform the relative susceptibility of large whale species to vessel strike in the action area as recorded systematically over the last five years (the period used for the SARs). However, we note that the SARs present strike data from the stock/DPS's entire range, which is much larger than the action area, and available ship strike records show that the majority of strikes that occur off the U.S. west coast occur in southern California. We summed the annual rates of serious injury or mortality from vessel collisions as reported in the SARs, then divided each species' annual rate by this sum to get the proportion of strikes for each species/stock. To inform the likelihood of striking a particular species of large whale, we multiplied the proportion of striking each species by the probability of striking at least one whale (i.e., 74 percent, as described by the Navy's probability analysis above). We note that these probabilities vary from year to year as the average annual mortality for a given five-year window in the SAR changes, however, over the years and through changing SARs, stocks tend to consistently maintain a relatively higher or relatively lower likelihood of being struck (and we include the annual averages from 2017 SARs in Table 138 to illustrate).

Table 138. Summary of factors considered in determining the number of individuals in each stock potentially struck by a vessel (M/SI – Mortality/Serious Injury).

Species	Stock	Annual rate of M/SI from vessel collision (observed from 2017 SARs)	Annual rate of M/SI from vessel collision (observed from 2019 Draft SARs)	Percent likelihood of hitting species/stock once (from 2019 Draft SARs)[1]	Total known strikes in OR, WA, northern CA (from 2000 to present)[2]	Total known Navy strikes in the action area	Rockwood et al. (2017) modeled vessel strikes[6]	Proposed takes (from the 3 total)	Annual proposed Take
Blue whale	Eastern North Pacific	0	0.4	3.7%	-	-	18	0	0
Fin whale	Northeast Pacific and California/Oregon/Washington	0.2 and 1.8	0.4 and 1.6	3.7% and 14.8%	10[3]	-	43	2	0.29
Sei whale	Eastern North Pacific	0	0.2	1.85%	-	-	-	0	0
Humpback whale	California/Oregon/Washington (Mexico and Central America DPSs)	1.1	2.1	19.425%	4[4]	1[5]	22	2	0.29
Sperm whale	California/Oregon/Washington	0.2	0	0	3	-	-	1	0.14

[1]The percentages do not add to 100 because non-ESA-listed species are not included in this table.

[2] Only one ship strike was reported in CA in the action area (Humboldt and Del Norte Counties). This strike occurred in 2004 in Humboldt County and was not identified to species.

[3] A total of 10 fin whale strikes are reported in the regional stranding database, however no information on stock is provided. As these two stocks of fin whales are known to overlap spatially and temporally in the action area, the 10 reported strikes could come from either stock or a combination of both stocks.

[4] A total of 4 humpback whale strikes are reported in the regional stranding database, however no information on DPS is provided. As these two DPSs of humpback whales are known to overlap spatially and temporally in the action area, the 4 reported strikes could come from either DPS or a combination of both DPSs.

[5] One humpback whale was reported as struck by a U.S. Coast Guard cutter operating on behalf of the Navy, however it was not possible for the Navy to determine which DPS this whale came from. As these two DPSs of humpback whales are known to overlap spatially and temporally in the action area, this whale could have come from either DPS.

[6] Rockwood et al. modeled likely annual vessel strikes off the West Coast for these three species only.

The probabilities calculated as described above are then considered in combination with the information indicating the species that the Navy has definitively hit in the NWTT action area since 1995 (since they started tracking consistently), and the species that are known to have been struck by any vessel (through regional strandings data) in the NWTT action area. We also note that Rockwood et al. (2017) modeled the likely vessel strike of blue whales, fin whales, and humpback whales on the U.S. West Coast, and those numbers help inform the relative likelihood that the Navy will hit those stocks/DPSs.

For each indicated stock/DPS, Table 138 includes the percent likelihood of hitting an individual whale once based on SAR data, total strikes from Navy vessels (from 1995), total strikes from any vessel (from 2000 from regional stranding data) and modeled vessel strikes from Rockwood et al. (2017). The last column indicates the annual serious injury or mortality due to vessel strike proposed for authorization in the MMPA rule.

Accordingly, stocks that have no record of having been struck by any vessel are considered unlikely to be struck by the Navy in the seven-year period of the rule. Stocks that have never been struck by the Navy, have rarely been struck by other vessels, and have a low likelihood of being struck based on the SAR calculation and a low relative abundance (Eastern North Pacific stock of blue whales and Eastern North Pacific stock of sei whales) are also considered unlikely to be struck by the Navy during the seven-year rule.

The two stocks/DPSs of humpback whales (California/Oregon/Washington and Central North Pacific stocks, former includes the Central America and Mexico DPSs) and two stocks of fin whales (California/Oregon/Washington and Northeast Pacific) are known to overlap spatially and temporally in the action area, and it is not possible to distinguish the difference between individuals of these stocks based on visual sightings in the field. The Navy has previously struck a humpback whale in the action area, and it is the second most common species struck by any vessel in the Study Area based on stranding data. Based on the SAR data, the two stocks of humpback whales also have the highest likelihood of being struck. Though the Navy has not definitively struck a fin whale in the action area (noting that the Navy could not rule out that the minke whale strike could have been a juvenile fin whale), fin whales are the most common species struck by any vessel in the Study Area based on stranding data. Based on the SAR data the California/Oregon/Washington stock has the third highest likelihood of being struck. Based on all of these factors, it is considered reasonably certain that humpback whales (from either the California/Oregon/Washington or Central North Pacific stocks) could be struck twice and fin whales (from either the California/Oregon/Washington or Northeast Pacific stocks) could be struck twice during the seven-year rule.

Based on the SAR data, sperm whales have a very low likelihood of being struck. However, three sperm whales have been struck by non-Navy vessels in the action area (in 2002, 2007, and 2012). Therefore, we consider it reasonably certain that a sperm whale would be struck by the Navy once during the seven-year rule period.

8.2.1.5.2 Response Analysis

Vessel collisions with large whales can result in death or serious injury of the animal. Wounds resulting from ship strike may include massive trauma, hemorrhaging, broken bones, or propeller lacerations (Knowlton and Kraus 2001). Superficial strikes may not kill or result in the death of the animal. The severity of injuries typically depends on the size and speed of the vessel (Conn and Silber 2013; Knowlton and Kraus 2001; Laist et al. 2001; Vanderlaan and Taggart 2007). Impact forces increase with speed, as does the probability of a strike at a given distance (Gende et al. 2011; Silber et al. 2010).

An examination of all known ship strikes from all shipping sources (civilian and military) indicates vessel speed is a principal factor in whether a vessel strike results in death or serious injury (Knowlton and Kraus 2001; Laist et al. 2001; Pace and Silber 2005; Vanderlaan and Taggart 2007). In assessing records in which vessel speed was known, Laist et al. (2001) found a direct relationship between the occurrence of a whale strike and the speed of the vessel involved in the collision. The authors concluded that most deaths occurred when a vessel was traveling in excess of 13 knots.

Jensen and Silber (2003) detailed 292 records of known or probable ship strikes (inclusive of military and non-military vessels) of all large whale species from 1975 to 2002. Of these, vessel speed at the time of collision was reported for 58 cases. Of these cases, 39 (or 67 percent) resulted in serious injury or death (19 of those resulted in serious injury as determined by blood in the water, propeller gashes or severed tailstock, and fractured skull, jaw, vertebrae, hemorrhaging, massive bruising or other injuries noted during necropsy and 20 resulted in death). Operating speeds of vessels that struck various species of large whales ranged from 2 to 51 knots. The majority (79 percent) of these strikes occurred at speeds of 13 knots or greater. The average speed that resulted in serious injury or death was 18.6 knots. Pace and Silber (2005) found that the probability of death or serious injury increased rapidly with increasing vessel speed. Specifically, the predicted probability of serious injury or death increased from 45 to 75 percent as vessel speed increased from 10 to 14 knots, and exceeded 90 percent at 17 knots. Higher speeds during collisions result in greater force of impact and also appear to increase the chance of severe injuries or death. While modeling studies have suggested that hydrodynamic forces pulling whales toward the vessel hull increase with increasing speed (Clyne et al. 1999; Knowlton et al. 1995), this is inconsistent with Silber et al. (2010), which demonstrated that there is no such relationship (*i.e.*, hydrodynamic forces are independent of speed).

In a separate study, Vanderlaan and Taggart (2007) analyzed the probability of lethal mortality of large whales at a given speed, showing that the greatest rate of change in the probability of a lethal injury to a large whale as a function of vessel speed occurs between 8.6 and 15 knots. The chances of a lethal injury decline from approximately 80 percent at 15 knots to approximately 20 percent at 8.6 knots. At speeds below 11.8 knots, the chances of lethal injury drop below 50 percent, while the probability asymptotically increases toward 100 percent above 15 knots. The

Jensen and Silber (2003) report notes that the database represents a minimum number of collisions, because the vast majority probably goes undetected or unreported. In contrast, Navy vessels are likely to detect any strike that does occur due to the number of Lookouts and other personnel onboard, and they are required to report all ship strikes involving marine mammals (Navy Memorandum for the Record; May 14, 2018).

Our exposure analysis considered vessel strike of marine mammals comprehensively, as a result of all Navy vessel movement within the action area, as opposed to in the context of specific training or testing exercises. For this reason, we are not able to predict the speed or size of Navy vessels that are expected to result in ship strikes of large whales. Because of these unknowns, we assume that all incidences of ESA-listed large whale vessel strike associated with Navy training and testing activities in the action area will result in mortality to the affected animal.

8.2.2 Leatherback Sea Turtles

In this section we address stressors that are likely to adversely affect leatherback sea turtles. Of all the potential stressors resulting from the proposed NWTT activities (see Section 5 *Potential Stressors*), we determined that only stressors associated with the use of explosives would likely result in adverse effects to leatherback sea turtles. Previously, in Section 8.1.2, we discussed those stressors associated with the proposed action that we determined were not likely to adversely affect leatherback sea turtles.

8.2.2.1 Explosives

Additional information on explosives as a potential stressor associated with the proposed action can be found in Section 5.1.2. For a discussion of the criteria and thresholds used to predict impacts from explosives on sea turtles see Section 2.2.3.

Explosives that may be used as part of the proposed action include bombs, missiles, rockets, naval gun shells, torpedoes, mines, demolition charges, and explosive sonobuoys (Navy 2019c). Explosions in the water or near the water surface can introduce loud, impulsive, broadband sounds into the marine environment. Unlike other acoustic stressors, explosions release energy at a high rate, producing a shock wave that can result in both sublethal and lethal effects on marine animals. Potential impacts include mortality, injury, hearing loss due to threshold shift (permanent or temporary), masking of other biologically relevant sounds, physiological stress, and changes in behavior. Based on what is known about potential sea turtle impacts from explosives studies and other activities that use explosives (e.g. oil and gas exploration), we assume underwater explosives can result in mortality, injury, and impairment of sea turtles that are exposed. Lethal injuries result from massive trauma or combined trauma to internal organs as a result of close proximity to the point of detonation. Types of lethal injuries include massive lung hemorrhage, gastrointestinal tract injuries (contusions, ulcerations, and ruptures), and concussive brain damage, cranial and skeletal (shell) fractures, hemorrhage, or massive inner ear trauma (Ketten 1995). Examples of nonlethal injuries include eardrum rupture, bruising, and immobilization of severely stunned animals. Stunned animals beneath the water may drown or become vulnerable to other impacts while they are immobilized. Minor organ injuries and

contusions can also occur as a result of underwater explosions; however, some sea turtles would be expected to recover over time through normal healing processes. Still, delayed complications arising from nonlethal injuries may ultimately result in the death of the animal because of potential increased risks from secondary infection, predation, or disease, and a reduced foraging capacity.

Explosive detonations involving the use of high-explosive munitions, including bombs, missiles, and naval gun shells, could occur in the air or near the water's surface. Explosive detonations associated with torpedoes and explosive sonobuoys would occur in the water column; and mines and demolition charges could be detonated in the water column or on the ocean bottom (Navy 2018b; Navy 2019c). Explosives associated with mine countermeasure and neutralization testing in the offshore area would be detonated in the water column only (no bottom explosives).

8.2.2.1.1 Impact Range to Effects from Explosives

As part of their quantitative analysis, the Navy modeled the distance that noise from an explosion would need to propagate to reach exposure level thresholds that would cause TTS, PTS, non-auditory injury, and mortality in sea turtles. Table 139 through Table 143 show the minimum, average, and maximum ranges to onset of physiological and behavioral effects based on the criteria and thresholds described in Section 2.2.3.3. Modeled ranges to TTS and PTS based on peak pressure for a single explosion generally exceed the modeled ranges based on SEL even when accumulated for multiple explosions. Ranges are provided for a representative source depth and cluster size (the number of rounds fired [or buoys dropped] within a very short duration) for each bin. For events with multiple explosions, Sound from successive explosions can be expected to accumulate and increase the range to the onset of an impact based on SEL thresholds. For more details on how range to effects were estimated refer to the Navy's *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (U.S. Department of the Navy 2018b).

Table 139. Ranges to non-auditory injury[1] (in meters) for sea turtles exposed to explosives as a function of animal mass.

Bin[2]	*Ranges to Non-Auditory Injury (m)[1]*	
	Animal Mass of 250 kg	*Animal Mass of 1000 kg*
E1	12 (11–13)	12 (11–13)
E2	16 (15–16)	16 (15–16)
E3	25 (25–45)	25 (25–45)
E4	31 (30–50)	31 (30–50)
E5	40 (40–40)	40 (40–40)
E7	79 (75–120)	79 (75–120)
E8	93 (90–110)	93 (90–110)
E10	155 (150–160)	155 (150–160)
E11	247 (190–270)	174 (170–260)

[1] Average distance (m) to non-auditory injury is depicted above the minimum and maximum distances which are in parentheses. The ranges depicted are the further of the ranges for gastrointestinal tract injury or slight lung injury for an explosive bin and animal mass interval combination.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 - 5), E5 (> 5 - 10), E7 (> 20 - 60), E8 (> 60 - 100), E10 (> 250 - 500), E11 (> 500 - 650)

Table 140. Ranges to mortality (in meters) for sea turtles exposed to explosives as a function of animal mass[1]

Bin[2]	*Ranges to Mortality (m)*	
	Animal Mass of 250 kg[1]	*Animal Mass of 1000 kg*[1]
E1	1 (1–1)	0 (0–0)
E2	2 (2–3)	1 (1–1)
E3	6 (6–10)	2 (2–5)
E4	8 (7–9)	4 (4–5)
E5	8 (7–8)	4 (3–4)
E7	29 (25–35)	16 (14–20)
E8	40 (40–40)	21 (21–21)
E10	27 (25–30)	16 (16–17)
E11	96 (70–100)	49 (45–50)

[1] Average distance (m) to mortality is depicted above the minimum and maximum distances which are in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 - 5), E5 (> 5 - 10), E7 (> 20 - 60), E8 (> 60 - 100), E10 (> 250 - 500), E11 (> 500 - 650)

Table 141. Peak pressure based ranges to TTS and PTS (in meters) for sea turtles exposed to explosives

Range to Effects for Explosives Bin: Sea turtles[1]			
Bin[2]	***Source Depth (meters)***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***
E1	0.1	37 (35—40)	69 (65—70)
E2	0.1	48 (45—50)	88 (80—90)
E3[3]	18.25 offshore area	80 (80—85)	154 (150—200)
E4	10	100 (100—100)	190 (190—190)
	30	105 (100—140)	262 (190—675)
	70	106 (100—160)	206 (190—350)
	90	103 (100—150)	197 (190—320)
E5	0.1	128 (120—130)	243 (230—250)
E7	10	255 (250—260)	471 (440—500)
	30	419 (240—1,025)	722 (440—1,025)
E8	45.75	434 (280—975)	956 (525—2,025)
E10	0.1	481 (470—490)	863 (850—875)
E11	91.4	929 (525—1,775)	2,122 (1,000—3,775)
	200	563 (525—800)	1,606 (1,000—3,525)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

[3] Distances for bin E3 are provided for the offshore area only since sea turtle analyses were based on occurrence in the offshore area only.

Table 142. Sound exposure level-based ranges (in meters) to TTS and PTS for sea turtles exposed to explosives

Range to Effects for Explosives Bin: Sea turtles[1]				
Bin[2]	***Source Depth (meters)***	***Cluster Size***	***Range to PTS (meters)[1]***	***Range to TTS (meters)[1]***
E1	0.1	1	0 (0—0)	0 (0—0)
E2	0.1	1	0 (0—0)	1 (1—1)
E3[3]	18.25 offshore area	1	3 (3—3)	17 (16—17)
E4	10	2	7 (7—7)	51 (50—55)
	30	2	7 (7—7)	47 (45—55)
	70	2	7 (7—7)	37 (35—50)
	90	2	7 (7—7)	36 (35—45)
E5	0.1	1	1 (1—1)	7 (7—8)
		8	3 (3—4)	18 (17—21)
E7	10	1	40 (40—40)	232 (190—290)
	30	1	30 (30—30)	254 (190—420)
E8	45.75	1	40 (40—55)	283 (260—400)
E10	0.1	1	14 (13—21)	87 (60—440)
E11	91.4	1	155 (150—200)	1,108 (775—2,275)
	200	1	111 (110—120)	872 (800—925)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

[3] Distances for bin E3 are provided for the offshore area only since sea turtle analyses were based on occurrence in the offshore area only.

Table 143. Ranges to behavioral response for sea turtles exposed to multiple explosions within any given event

Range to Effects for Explosives Bin: Sea turtles[1]			
Bin[2]	***Source Depth (meters)***	***Cluster Size***	***Range to Behavioral (meters)***[1]
E1	0.1	1	4,265 (4,025—4,775)
E2	0.1	1	4,744 (4,275—5,025)
E3	18.25 offshore area[4]	1	13,585 (3,025—31,525)
E4	10	2	16,692 (5,525—36,275)
	30	2	16,400 (8,275—31,275)
	70	2	15,122 (8,775—31,775)
	90	2	14,793 (8,025—33,025)
E5	0.1	1	7,011 (6,025—8,275)
		8	7,011 (6,025—8,275)
E7	10	1	29,798 (9,775—77,775)
	30	1	31,345 (16,025—61,775)
E8	45.75	1	58,351 (13,525—100,000)
E10	0.1	1	34,037 (9,525—64,775)
E11	91.4	1	84,776 (31,275—100,000)
	200	1	99,914 (93,775—100,000)

[1] Distances in meters (m). Average distance is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

[2] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

[3] Significant behavioral reactions beyond the TTS range are only estimated in instances where explosive cluster size is greater than one (U.S. Department of the Navy 2018b).

[4] Distances for bin E3 are provided for the offshore area only since sea turtle analyses were based on occurrence in the offshore area only.

8.2.2.1.2 Exposure and Response Analysis

In this subsection we summarize the results from the Navy's quantitative acoustics effects model and discuss the anticipated responses (i.e., numbers of individuals taken, types of take anticipated) based on the leatherback sea turtle exposure levels predicted by the model. The

NAEMO model takes into account (1) criteria and thresholds used to predict impacts from explosives, (2) the density and spatial distribution of leatherback sea turtles, and (3) the influence of environmental parameters (e.g., temperature, depth, salinity) on sound propagation and explosive energy when estimating the received sound level and pressure on the animals. For details on the approach used to evaluate the effects of explosives on leatherback sea turtles and model inputs refer to Section 2.2.3 (criteria and thresholds) and Section 2.4.1 (leatherback densities) of this opinion, and the Navy's technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (Navy 2018d).

The quantitative analysis, using a maximum year of training and testing activities, estimates that no leatherback sea turtle mortalities or non-auditory injuries would occur as a result of NWTT explosive activities. The mortality threshold is based on the exposure level expected to result in extensive lung hemorrhage. The data used to derive the threshold equations for onset of mortality are from Richmond et al. (1973). The injury threshold is based on the exposure level expected to result in onset of a slight lung injury and/or contusions to the gastrointestinal tract. The data and theory used to derive these threshold are from Richmond et al. (1973) and Goertner (1982). In addition to minor lung injuries or gastrointestinal tract contusions from the blast wave, it is possible that sea turtles may be physically injured due to fragmentation of exploding munitions. However, given that fragments would quickly decelerate in water, and that injury due to the blast wave would extend much further than any risk from fragmentation, sea turtles that may experience injury from fragmentation are also assumed to experience injury due to the blast wave. As such, the estimates produced by NAEMO modeling for non-auditory injuries are assumed to encompass any sea turtles that may also be injured due to fragmentation.

During a maximum year of training and testing activities, the quantitative analysis also estimated no PTS or TTS exposures of leatherback sea turtles. The quantitative analysis estimated four annul exposures to leatherback turtles from explosions during NWTT activities (all from testing) at or exceeding 175 dB re 1 µPa SPL (rms). The quantitative analysis estimated 26 such exposures to leatherback turtles from explosions during NWTT testing activities over any given seven-year period of the proposed action. It is assumed that some portion of these exposures would result in behavioral harassment responses. As discussed above for marine mammals, significant leatherback sea turtle behavioral responses to solitary explosions (i.e., cluster size equals one) are not anticipated due to the short duration of acoustic exposure from such explosions. NAEMO exposure estimates represent the total number of exposures, and not necessarily the number of individuals exposed as a single individual may be exposed multiple times over the course of a year.

NAEMO estimates are further refined by considering animal avoidance of sound-producing activities and implementation of mitigation (see Section 3.5.1 for details). Procedural mitigation measures include delaying or ceasing applicable detonations when a sea turtle is observed in a mitigation zone. The mitigation zones for explosives encompass the estimated ranges to

mortality for a given explosive. Therefore, the impact analysis quantifies the potential for mitigation to reduce the risk of mortality due to exposure to explosives (i.e., mitigated to the level of injury). The impact analysis does not analyze the potential for mitigation to further reduce the risk of PTS, TTS or behavioral effects, even though mitigation could also reduce the likelihood of these effects.

The Navy will also implement mitigation measures for in-water explosives within mitigation areas (see Section 3.5.2 for details). The benefits of mitigation areas have not been factored into the quantitative analysis, but rather are discussed here qualitatively. Leatherback sea turtle critical habitat overlaps part of the Marine Species Coastal Mitigation Area. The Navy will not conduct explosive training and testing activities within 50 nautical miles from shore with the exception of explosive mine countermeasure and neutralization testing activities. Leatherback sea turtle critical habitat also encompasses the Olympic Coast National Marine Sanctuary Mitigation Area. Within the Olympic Coast Mitigation Area, the Navy will not conduct any explosive training and testing activities. Although developed for humpback whales, seasonal restrictions on the use of explosives in the Stonewall and Heceta Bank and Point St. George Humpback Whale Mitigation Areas could benefit leatherback turtles if they transit through these mitigation areas during the applicable season.

There is very limited data available regarding the behavioral responses of sea turtles to anthropogenic sound sources. Sea turtle behavioral responses to an explosion could include a startle response, leaving an area, avoiding an area, diving, or a disruption of activity (e.g., feeding or resting). Because sea turtles exhibit avoidance behaviors to air gun exposure at levels above 175 dB rms (re 1 µPa), responses to explosive detonations could be similar. Exposure to multiple detonations over a short period may cause a sea turtle to exhibit behavioral reactions such as interruption of feeding or avoiding the area. However, exposure to a single blast during an event, which is the most probable scenario during Navy activities, would more likely result in a short-term startle response. Sea turtles would presumably return to normal behaviors quickly after exposure to a single blast, assuming the exposure did not result in injury. Additionally, significant behavioral responses that result in disruption of important life functions are more likely to occur from multiple exposures within a longer period of time. We do not expect this to occur as a result of the Navy's use of explosives during their training and testing exercises. Most explosions occur in more discrete areas and would not likely persist for long enough periods of time to result in a significant, long-term behavioral response with fitness consequences. Therefore, the anticipated impacts on fitness and survival are minor and short-term for the small number of leatherback sea turtles that would be exposed at levels that could elicit a behavioral response. Leatherback sea turtles that experience a strong behavioral response are also expected to experience a physiological stress response. Whereas stress is an adaptive response that does not normally place an animal at risk, distress involves a chronic stress response resulting in a negative biological consequence to the individual. Stress responses from this stressor are expected to be short-term in nature given that in most cases sea turtles would not experience

repeated exposure to explosives. As such, we do not anticipate stress responses would be chronic, involve distress, or have negative long-term impacts on any individual leatherback sea turtle's fitness. In summary, while a small number of leatherback sea turtles (i.e., up to four estimated exposures per year) would experience behavioral harassment and physiological stress responses from exposure to explosives, these responses alone are not expected to have any long-term impacts, nor to affect the fitness of individual sea turtles.

8.2.3 Fishes – Effects of Explosive Stressors

Of all the potential stressors resulting from the proposed NWTT activities (see Section 5 *Potential Stressors*), we determined that only stressors associated with the use of explosives would likely result in adverse effects to ESA-listed fishes. Previously, in Section 8.1.38.1.2, we discussed those stressors associated with the proposed action that we determined were not likely to adversely affect ESA-listed fishes. The effects of explosions on fish have been studied and reviewed by numerous authors (Hawkins and Johnstone 1978; Popper 2008; Popper et al. 2007a; Popper et al. 2014b; Tavolga and Wodinsky 1963). This section discusses the effects of explosive stressors from the proposed action on ESA-listed chinook salmon (all ESUs listed in Section 6.2), steelhead (All ESUs and DPSs listed in Section 6.2), coho salmon (all ESUs listed in Section 6.2), Columbia River ESU and Hood Canal Summer-Run ESU of chum salmon, Snake River ESU of sockeye salmon, Puget Sound/Georgia Basin DPS of bocaccio and yelloweye rockfish, Southern DPS eulachon, and Southern DPS green sturgeon (see Section 5.1.2 for a general discussion of explosives as a potential stressor).

8.2.3.1 Exposure Analysis – Fishes

Training and testing activities using explosive ordnances that could affect ESA-listed fish will occur in the offshore portion of the action area and in inland waters. The general categories of the explosives that are likely to adversely affect ESA-listed fishes in the proposed action area, such as size and number of detonations, are presented in Table 144 and Table 145 below.

Table 144. Maximum annual offshore explosive sources that could be used in-water or at the water's surface.

BIN	Net Explosive Weight (lb)	Location in the NWTT Offshore[1]	In-water or at surface?	Activity	Training Annual	Testing Annual
E1	0.1–0.25	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	120	--
				Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft	--	8
E2	> 0.25–0.5	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	130	--

E3	> 0.5-2.5	> 50 NM	Surface	Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (SUS)	--	72
E4	>2.5–5	< 1,000 feet depth	In-water	Mine Countermeasure and Neutralization	--	36
E5	> 5–10	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	112	--
E7	> 20– 60	< 1,000 feet depth	In-water	Mine Countermeasure and Neutralization	--	5
E8	> 60– 100	> 50 NM	Surface	Torpedo (Explosive) Testing (MK-46/54)	--	4
E10	> 250– 500	> 50 NM	Surface	Missile Exercise Air-to-Surface	2	--
			Surface	Bombing Exercise Air-to-Surface	2	--
E11	> 500–650	> 50 NM	Surface	Torpedo (Explosive) Testing (MK-48)	--	4
[1] All explosive activities are required to occur outside the Olympic Coast National Marine Sanctuary						

Table 145. Maximum annual inland explosive sources that could be used in-water or at the water's surface.

BIN	Net Explosive Weight (lb)	Location in Inland Waters	In-water or at surface?	Activity	Training Annual	Testing Annual
E0	< 0.1	Crescent Harbor	In-water	Mine Neutralization – Explosive Ordnance Disposal (EOD)	18	--
		Hood Canal			18	
E3	> 0.5–2.5	Crescent Harbor	In-water	Mine Neutralization – Explosive Ordnance Disposal (EOD)	3	--
		Hood Canal			3	

Under the Proposed Action (Section 3), training activities using explosives consist of anti-submarine warfare, mine warfare, and surface warfare training in the offshore area and the NWTT inland area, and the amount of explosives used could fluctuate between years. Testing activities using explosives consist of anti-submarine warfare, mine warfare, and surface warfare testing in the offshore Area, and the amount of explosives used could fluctuate between years. There are no training or testing activities that involve the use of explosives in the Western Behm Canal; therefore, there would be no impacts on fishes that occur in these areas. In addition, testing activities in the Inland Waters would not involve the use of explosives. Explosives are used less frequently under testing activities than under training activities. Overall training and

testing activities would occur in the same general locations and in a similar manner as previously analyzed in NMFS (2015a), with one exception.

A new explosive mine countermeasure and neutralization testing activities would occur closer to shore than other activities previously analyzed in NMFS (2015a) (i.e., shoreward of 50 nautical miles) that involved the use of in-water explosives. This activity would typically occur greater than three nautical miles from shore in the Quinault Range Site, or greater than 12 nautical miles from shore elsewhere in the offshore area. Mine countermeasure and neutralization testing using explosives within bin E7 will be no closer to shore than six nautical miles in the Quinault Range Site (Navy 2019a). These activities involving explosives would occur approximately two times per year and typically in water depths shallower than 1,000 feet. E4 explosives would occur in the water column 20 feet or greater below the water surface; E7 explosives would occur in the water column 100 feet or greater below the water surface. Moored E7 mines associated with this activity would be at least 75 feet from the seafloor, and no explosive mines would be detonated on the ocean bottom (Navy 2020a). Mine neutralizers (bin E4) would be used with inert targets either on the seafloor and in the water column, 20 feet or greater below the water surface. The E4 explosives used for this activity would have a directional force aimed at the target. The maximum annual number of explosives proposed by the Navy for this activity is 36 bin E4 and 5 bin E7 explosives. The Navy's proposed action also includes a cap on the seven year total number of offshore explosives for this activity as follows: 108 E4 and 15 E7 explosives.

The Navy calculated ranges to effects for fishes with and without swim bladders using NMFS' exposure criteria as discussed in Section 2.3.2 and presented in Table 9 and Table 146. These exposure criteria were used to predict the distances to the onset of mortality, injury, and TTS within fishes within the NWTT action area. Using those criteria, the Navy developed ranges to effects for fish mortality, injury, and TTS (See Table 147 and Table 148 below). Page: 703 Ranges may vary greatly depending on factors such as the cluster size, location, depth, and season of the event, all of which are accounted for in NAEMO. Fishes within these ranges could be exposed above the effect threshold levels and a portion may experience those effects. The NAEMO model which predicted these ranges cannot account for the highly non-linear effects of cavitation and surface blowoff; therefore, some estimated ranges may be overly conservative. Also, the ranges are the distance where the threshold is not exceeded at any depth where animals could be present (the entire water column for fishes). Thus, portions of the water column within the ranges shown would not exceed mortality, injury, or TTS thresholds (i.e., the range does not represent a cylinder of effect in the water column). In some instances, a significant portion of the water column within the ranges shown may not exceed threshold (S. Kotecki, Navy, personal communication to NMFS, October 4, 2020).

As noted previously (Section 2.3.2), our range to effects for fish injury are based on the more conservative acoustic injury criterion of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are

based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors also likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

Table 146. Sound exposure criteria used to calculate ranges to effects.

Fish Hearing Group	Onset of Mortality	Onset of Injury	TTS
	SPL_{peak}	SPL_{peak}	(SEL_{cum})
Fishes without a swim bladder	229	220	NC
Fishes with a swim bladder not involved in hearing	229	> 207	> 186

Notes: SEL_{cum} = Cumulative sound exposure level (decibel referenced to 1 micropascal squared seconds [dB re 1 µPa^2-s]), SPL_{peak} = Peak sound pressure level (decibel referenced to 1 micropascal [dB re 1 µPa]), > indicates that the given effect would occur above the reported threshold. Notes: TTS = Temporary Threshold Shift. NC = no criteria, > indicates that the given effect would occur above the reported threshold.

Table 147. Ranges to mortality and injury from explosives for fishes with and without a swim bladder.

Bin[1]	*Range to Effects (meters)*		
	Onset of Mortality	*Onset of Injury*	
	229 SPL_{peak} Average (Min-Max)	*220 SPL_{peak} (fishes without a swim bladder) Average (Min-Max)*	*> 207 SPLpeak (fishes with a swim bladder) Average (Min-Max)*
E0[2] Inland Waters (Hood Canal)	49 (35–75)	125 (90–220)	< 427 (160—875)
E0[2] Inland Waters (Crescent Harbor)	35 (35–45)	100 (95–150)	< 661 (260—2,275)
E1	50 (45–50)	124 (120–140)	< 556 (450—950)
E2	64 (60–65)	163 (150–170)	< 595 (575—600)
E3 Inland Waters (Hood Canal)	162 (120–260)	358 (160–675)	< 1,210 (160—2,525)
E3 Inland Waters (Crescent Harbor)	129 (120–180)	400 (230–1,525)	< 1,674 (800—4,525)
E3 offshore area	111 (110–120)	322 (270–600)	< 1,473 (1,000—3,025)
E4	150 (140–370)	466 (350–1,025)	< 1,829 (1,025—3,525)

E5	177 (170–180)	447 (430–460)	< 1,478 (1,275—1,525)
E7	424 (320–1,025)	1,142 (775–2,275)	< 4,259 (2,025—9,525)
E8	644 (380–1,275)	1,708 (950–3,275)	< 6,411 (2,525—12,275)
E10	644 (625–650)	1,478 (1,275–1,525)	< 4,636 (4,275—4,775)
E11	1,287 (725–3,025)	3,913 (2,025–7,275)	< 16,795 (5,025—27,275)

[1]Bin (net explosive weight, lb.): E0 (< 0.1), E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

[2] Estimated ranges for E0 are consistent with measurements from a series of monitoring events during training activities that used explosives in the Inland Waters (Hart 2012; U. S. Department of the Navy 2015; U.S. Department of the Navy 2009a; U.S. Department of the Navy 2013a; U.S. Department of the Navy 2014a; U.S. Department of the Navy 2015a; U.S. Department of the Navy 2015b; U.S. Department of the Navy 2015c).

Notes: NEW = net explosive weight, SPL_{peak} = Peak sound pressure level. Range to effects represent modeled predictions in different areas and seasons within the action area. Bin E0 and E3 are the only bins used in the Inland Waters. Bin E3 is used in both the Inland Waters and offshore area, therefore ranges to effect are shown for both the Inland Waters and the offshore area for this bin. All other range to effects were calculated in the offshore area where these bins would be used. Each cell contains the estimated average, minimum and maximum range to the specified effect.

Table 148. Range to TTS for fishes with a swim bladder from explosives.

Bin[1]	***Cluster Size***	***Range to Effects (meters)***
		TTS
		SEL_{cum}
E0[2] Inland Waters (Hood Canal)	1	< 59 (50-85)
E0[2] Inland Waters (Crescent Harbor)	1	< 48 (45–50)
E1	1	< 53 (45–55)
E2	1	< 58 (55–60)
E3 Inland Waters (Hood Canal)	1	<259 (160–440)
E3 Inland Waters (Crescent Harbor)	1	<250 (200–600)
E3 offshore area	1	< 150 (140–160)
E4	2	< 340 (270–750)
E5	1	< 158 (150–200)
	8	< 394 (380–430)
E7	1	< 974 (675–1,775)

E8	1	< 1,110 (725–1,775)
E10	1	< 570 (550–650)
E11	1	< 2,693 (1,525–5,025)

[1]Bin (net explosive weight, lb.): E0 (< 0.1), E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

[2] Estimated ranges for E0 are consistent with measurements from a series of monitoring events during training activities that used explosives in the Inland Waters (Hart 2012; U. S. Department of the Navy 2015; U.S. Department of the Navy 2009a; U.S. Department of the Navy 2013a; U.S. Department of the Navy 2014a; U.S. Department of the Navy 2015a; U.S. Department of the Navy 2015b; U.S. Department of the Navy 2015c).

Notes: NEW = net explosive weight, SEL_{cum} = Cumulative sound exposure level, TTS = Temporary Threshold Shift, "<" indicates that the given effect would occur at distances less than the reported range(s). Range to effects represent modeled predictions in different areas and seasons within the action area. Bin E0 and E3 are the only bins used in the Inland Waters. Bin E3 is used in both the Inland Waters and offshore area, therefore ranges to effect are shown for both the Inland Waters and the offshore area for this bin. All other range to effects were calculated in the Offshore Area where these bins would be used. Each cell contains the estimated average, minimum and maximum range to the specified effect.

The effects on species from exposure to the Navy's proposed use of explosives may result in mortality, non-lethal injury, temporary loss of hearing, physiological stress, masking, and behavioral responses. In the sections below we analyze the effects of these explosions on ESA-listed fish species in the offshore and inland portions of the action area. As stated above, effects on species is determined by the specific threshold criteria the Navy used based upon a fish's hearing sensitivity (e.g. hearing specializations and sound detections of the specific source) and physical characteristics of the species (e.g. presence and type of swim bladder). Along with these, several other factors influence the potential degree of impact, such as level and duration of sound, where in the sound field the fish is in proximity to the source, as well as the current condition and attentional focus of the fish.

Below provides a summary of our exposure analysis for offshore and inland waters within the NWTT action area. This assessment provides estimates on the number of injuries and mortalities from explosive events during NWTT training and testing activities. When assessing the effects of explosions on salmonids and other listed fishes, we chose to use available data (which is often quite limited) on fish numbers to describe the likelihood of any listed fish occupying an ocean area where an explosion would occur in. Other more qualitative approaches to describing this likelihood could be used and would arrive at very similar results, which are described in the sections below.

We used the available data on fish numbers to illustrate the relative sparseness of listed salmonids in the ocean. For example, the number reported for Central California Coast coho salmon adults should not be read as the precise number of adults NMFS believe occupied the Pacific Ocean in 2019. This number is the geometric mean of spawner survey data ranged from 2014 to 2018. This example is similar in scope for other ESA-listed fishes impacted by NWTT

activities. Abundance such as this is as precise and accurate as we can be based on the available data.

8.2.3.1.1 Offshore Area - Exposure Analysis

It is difficult to accurately estimate the number of individuals from each ESA-listed species (ESU/DPS) that would experience adverse effects from elevated underwater noise and sound pressures in the offshore environment because fish distribution is influenced by a number of environmental factors. Further, salmon, steelhead, and green sturgeon from different ESUs/DPSs (some ESA-listed and some not) are mixed in the open ocean environment (Bellinger et al. 2015), making it even more difficult to identify how many individuals of a certain ESU/DPS may be affected by an activity at a specified location and time. Ocean salmon fisheries off the west coast of the United States are managed using a tool that predicts cohort-based stock abundance and time and area stock compositions (Bellinger et al. 2015). This allows managers to predict harvest levels from specific stocks of fish in order to maximize harvest, while maintaining conservation goals. However, a similar analysis was not possible for this opinion because the Navy is not able to identify when and where specific activities will occur beyond the general terms described in Section 3 (e.g., > 50 nautical miles offshore of Washington, Oregon, or California).

We estimated the number of salmon, steelhead, eulachon, and green sturgeon (ESA-listed rockfish do not occur in the offshore portion of the action area) that may be killed or injured from detonations in the offshore environment using the acoustic threshold criteria established for this opinion and available information on the offshore densities and distributions of the ESA-listed fish species considered in this opinion. Below we summarize the information available regarding ESA-listed fish habitat use in offshore marine waters. This information will allow us to determine which ESUs and DPSs may be affected by a specific activity, which life stage may be affected, and to assess the effect of underwater explosions on the ESA-listed fish species considered in this opinion. For more details on the life history and distribution of these fish species in the action area refer to their respective Status of the Species sections above (Section 6.2).

Chinook Salmon Habitat Use

Chinook salmon are widely distributed in offshore waters of the North Pacific Ocean, however, little is known about their oceanic ecology (Courtney et al. 2019). Chinook salmon distribution in marine waters can be identified in general terms only because it varies seasonally and inter-annually due to a variety of environmental factors (PFMC 2014). Two general life history strategies have been described for Chinook salmon outmigrating from their natal rivers: subyearling life history types which enter marine waters during their first year of life and tend to remain in shallow coastal waters, and yearling types, which spend more time in freshwater before migrating to the ocean, and migrate further offshore and north faster than subyearlings (Burke et al. 2013). After the juvenile rearing phase, anadromous individuals migrate to the

ocean where they remain for one to six years, before reaching maturity and returning to their natal river to spawn. The spawning migration of Chinook salmon is variable with most northern populations (e.g., Alaska) returning in the spring (i.e., spring run), whereas southern populations may return in the spring, summer (i.e., summer run), or fall (i.e., fall run) months (Courtney et al. 2019). Additionally, some Chinook salmon originating from Puget Sound remain within the Salish Sea throughout their lives, before returning to their natal systems as adults to spawn (Chamberlin et al. 2011). In general, once Chinook salmon leave their natal rivers, they use the cool, upwelled waters of the continental shelf for migration and feeding (Bellinger et al. 2015). Juvenile Chinook salmon are generally found within 55 km (about 30 nm) of the Washington, Oregon, and California coast, with the vast majority found less than 28 km (about 15 nm) offshore (Fisher and Pearcy 1995; Pearcy and Fisher 1990; PFMC 2014). As a result, the majority of juvenile salmon would not overlap with the majority of the explosive events which occur outside 50 nm[21]. This supports Hassrick et al. (2016) which noted that in the weeks to months after ocean entry, California's juvenile Chinook salmon population appear to be primarily constrained to coastal waters near natal river outlets. Bi et al. (2011) similarly found that while juvenile Chinook salmon can occur in more offshore, deeper waters, they are most abundant on the inner continental shelf at a median depth of 55 meters. Bi et al. (2007) found that subyearling, yearling, and subadult Chinook salmon abundances were negatively correlated with depth. However, juvenile salmon may be located in deeper waters along the shelf due to seasonal changes. In the coastal waters of Southeast Alaska, immature Chinook salmon were reported to occur deeper in winter than in fall or spring, and smaller, younger fish were distributed shallowest. In the Georgia Strait, in coastal British Columbia, juvenile Chinook salmon were distributed deeper in September than from May to July (Beamish et al. 2011). Ultrasonic telemetry in Puget Sound, Washington, found immature Chinook salmon in increasing deeper depths seasonally: fish were closest to the surface in spring, deeper in summer, deeper yet in fall, and deepest in winter (Riddell et al. 2018). Peterson et al. (2010) found that during sampling of continental shelf and oceanic waters of the Pacific Northwest, juvenile salmonids were found almost exclusively in continental shelf waters. With this said, coded wire tag recoveries from high seas fisheries and tagging programs suggest specific populations of Chinook salmon do utilize areas outside the continental shelf. This is supported by data presented in Sharma and Quinn (2012) which showed that Snake River fall/spring/summer run Chinook, Upper Columbia River spring Chinook, and Upper Willamette River Chinook salmon were located at distances farther offshore than other Chinook ESUs. However, commercial fisheries catch data suggest that most maturing Chinook salmon off the West Coast of the continental United States are found within 60 km (about 32.3 nm) of the coast (PFMC 2014).

[21] This excludes juveniles from the Snake River fall/spring/summer run Chinook, Upper Columbia River spring Chinook, and Upper Williamette River Chinook salmon due to data presented in Sharma, R., and T. P. Quinn. 2012. Linkages between life history type and migration pathways in freshwater and marine environments for Chinook salmon, Oncorhynchus tshawytscha. Acta Oecologica 41:1-13. which show these populations occurring further offshore.

Chinook salmon are thought to be less surface oriented than other Pacific salmon, most abundant at depths of 30 to 70 m, and most often caught as adults in commercial troll fisheries at depths of 30 meters or greater (PFMC 2014). However, juvenile Chinook salmon are known to be more abundant than adults near the surface, most frequently found at depths of less than 37 meters (Fisher and Pearcy 1995). Walker et al. (2007) observed Chinook salmon at an average depth of 42 meters. Through observations of 883 purse seine net sets off the Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

Migratory patterns of Chinook salmon can vary greatly within and among populations (PFMC 2014), but some general patterns have been described. For example, Chinook salmon originating from north of Cape Blanco in Oregon tend to migrate towards the Gulf of Alaska, whereas those originating south of Cape Blanco tend to migrate west and south to forage in waters off Oregon and California (PFMC 2014). Weitkamp (2010) examined coded wire-tag recovery data and found that Chinook salmon originating from a particular freshwater region share a common marine distribution. Chinook salmon originating from Washington and Oregon were recovered within an area from their respective state coasts to southeast Alaska, and fish originating from southern Oregon and California were generally only recovered off the coast of Oregon and California. While these general patterns have been observed, Weitkamp and Neely (2002) suggested that Pacific salmon, including Chinook salmon, exhibit high diversity in ocean migration patterns, rivaling the variability that has been well demonstrated in freshwater life history. Celewycz et al. (2014) presented data analyzing Chinook salmon distribution in the eastern North Pacific Ocean from coded wire tag recoveries. Significant variability in ocean distribution was observed. For example, Chinook salmon from Washington and Oregon were recovered as far north as Bristol Bay, but as far south as northern California. Chinook salmon from Idaho were recovered as far north as the Gulf of Alaska and as far south as northern California. However, most recoveries occurred off the Washington coast and west of Vancouver Island.

Chinook salmon from the Puget Sound ESU were previously thought to be generally found in ocean environments from the Washington coast to the west coast of Vancouver Island, though a small percentage are recovered as far north as Alaska (NMFS 2008c). Bellinger et al. (2015) used genetic stock identification techniques in the ocean salmon troll fishery and found Puget Sound salmon as far south as Fort Bragg, California. Chinook salmon from the Lower Columbia River ESU are found as far north as Alaska (NMFS 2008c), but as far south as offshore of San Francisco (Bellinger et al. 2015). Upper Willamette River Chinook salmon are a far north migrating stock (NMFS 2008c), though Bellinger et al. (2015) observed individuals from this ESU caught as far south as the Klamath River region. Van Doornik et al. (2019) found a significant relationship between juvenile catch per unit effort of interior Columbia River spring-run stocks off the Southeast Alaska coast. The center of Snake River fall-run ESUs ocean distribution is thought to be located off the west coast of Vancouver Island (NMFS 2008c), though they have been found as far south as the Bay Area, California (Bellinger et al. 2015).

Bellinger et al. (2015) also observed Central Valley spring-run Chinook salmon caught in Central Oregon Coast fisheries. The California coastal ESU of Chinook salmon appears to be widely distributed along the mainland west coast of the U.S., with Bellinger et al. (2015) observed individuals from this ESU from Northern Oregon to south of Monterey Bay. This encompassed the author's entire study area, indicating the species' distribution could range even further.

A number of factors can drive variation in migratory pathways for Chinook salmon. Bi et al. (2011) found that copepod community structure helps determine salmon distribution in oceanic waters because it provides useful information on ocean conditions (i.e., strength of upwelling). Burke et al. (2013) found that Columbia River yearling Chinook salmon have stock-specific spatial distributions in the marine environment that shift through time. The authors found that geospatial variation (e.g., latitude and distance from shore) drove habitat selection in the marine environment more than environmental variation (e.g., chlorophyll *a* and temperature), potentially leading individuals to select habitat areas with suboptimal environmental conditions. Bi et al. (2007) indicated that coho salmon abundance was strongly correlated with variations in chlorophyll *a* concentrations (which vary annually), and observed large temporal variations in overall habitat usage in waters off the Washington and Oregon coast. Sampling in 2003 and 2000 indicated large areas of habitat usage off the coast of Washington and Oregon, whereas the total area of habitat usage was lower in 2001, 2002, and 2004. This study also highlighted the variability in habitat selection by life stage with subyearling and subadult Chinook salmon found closer to shore than yearling Chinook salmon.

Coho Salmon Habitat Use

In general, once coho salmon leave their natal rivers, they use the cool, upwelled waters of the continental shelf for migration and feeding (Bellinger et al. 2015). Two general patterns have been described for coho salmon once they emigrate from freshwater with some spending several weeks in coastal waters before migrating north and offshore, and others remaining in coastal waters for at least the first summer before migrating north (PFMC 2014). The degree to which juveniles migrate offshore appears to depend on the strength of upwelling, with strong upwelling years leading to wider dispersal, farther from shore (Pearcy 1992). Juvenile coho salmon appear to be distributed further offshore than juvenile Chinook salmon. Bi et al. (2011) observed juvenile coho salmon to be most abundant at a median depth of 88 meters and Chinook salmon to be most abundant at a median depth of 58 m. While coho salmon can be found further offshore, juvenile and maturing coho salmon are most abundant within 60 km off the coasts of Washington, Oregon, and California (PFMC 2014). The majority of juveniles are found within 37 km of the coast (Pearcy 1992; Pearcy and Fisher 1990). Similarly, Bi et al. (2007) found that abundance of yearling and subadult coho salmon was negatively correlated with depth.

In marine waters, coho salmon are generally found within the upper portion of the water column (PFMC 2014). Walker et al. (2007) found that the average depth of coho salmon in the North

Pacific Ocean was 11 meters. Through observations of 883 purse seine net sets off the Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

Similar to Chinook salmon, coho salmon distribution in the marine environment varies considerably among seasons, years, life stages, and populations. Weitkamp and Neely (2002) provided evidence that coho salmon exhibit high diversity in ocean migration patterns, rivaling the variability that has been well demonstrated in freshwater life history. The authors also showed that coho salmon from different freshwater regions inhabit different areas of the coastal ocean, identifying 12 distinct ocean distribution patterns from California to Alaska. However, despite these general patterns, fish from a given population were widely distributed in the coastal ocean (Weitkamp and Neely 2002). Bi et al. (2007) indicated that coho salmon abundance was strongly correlated with variations in chlorophyll *a* concentrations (which vary annually), and observed large temporal variations in overall habitat usage in waters off the Washington and Oregon coast. Sampling in 2003 and 2000 indicated large areas of habitat usage off the coast of Washington and Oregon, whereas the area of habitat usage was lower in 2001, 2002, and 2004. This study also highlighted the variability in habitat selection by life stage with subadult coho salmon found closer to shore than yearling coho salmon.

Chum Salmon Habitat Use

The ocean distribution of chum salmon is thought to be broad compared to other Pacific salmon (Neave et al. 1976), with the species found throughout the North Pacific Ocean north of the Oregon/Washington border. In general, chum salmon move north and west along the coast upon entering saltwater, and have moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). Pearcy and Fisher (1990) observed the highest catch per unit effort of juvenile chum salmon inshore of 37 km (approximately 20 nm), though some were caught over 55 km (approximately 30 nm) offshore. Hartt and Dell (1986) observed that the vast majority of juvenile chum salmon from Washington state migrate northward within a narrow coastal belt less than 20 nm. Though juvenile chum salmon are generally believed to migrate far to the north by the late summer after they have entered saltwater in the spring (Hartt and Dell 1986), Pearcy and Fisher (1990) suggest that at least some individuals reside in coastal Washington waters for several months after they enter the marine environment. Pearcy and Fisher (1990) noted that juvenile chum salmon were less abundant than either coho salmon or Chinook salmon off the Oregon and Washington coast. Chum salmon are known to be surface oriented, using the upper 20 meters of the water column 78 percent of the time during the day and 95 percent of the time at night. The remaining time, they can be found down to depths of 60 meters (Ishida et al. 1997). Similarly, Walker et al. (2007) found the average depth of chum salmon to be 16 meters in the North Pacific Ocean. Through observations of 883 purse seine net sets off the Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids. Data from Neave et al. (1976) indicated that catches of chum salmon in the NWTT action area off the coast of the continental U.S. were lower than areas further to the north and further offshore (e.g., the Gulf of Alaska, the Bering Sea, and areas far offshore in the North

Pacific). Due to this and due to the northwest migration pattern of chum, we do not expect the Hood Canal summer-run ESU of chum to overlap with offshore NWTT explosive activities. This is because individuals from the Hood Canal summer-run ESU are only expected to migrate through portions of offshore NWTT action area that will not conduct offshore testing and training explosive activities (i.e., the Juan de Fuca Eddy Marine Species Mitigation Area and the Olympic Coast National Marine Sanctuary).

Sockeye Salmon Habitat Use

In general, it is thought that sockeye salmon follow a similar migration pattern as chum salmon once they enter the ocean, moving north and west along the coast, and have moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). Previously, French et al. (1976b) summarized the general migration pattern of sockeye salmon originating in the various tributaries of the northeastern Pacific Ocean from the Alaska Peninsula to the Columbia River. Tag recovery data indicated a general mixing of these stocks during their residence in the northeastern Pacific Ocean. These fish primarily occur east of 160° W and north of 48° N (north of the NWTT action area). It is thought that most fish originating from these areas have departed the high seas by early August of their second year at sea, to return to their natal rivers to spawn (French et al. 1976b). Pearcy and Fisher (1990) observed the highest catch per unit effort of juvenile sockeye salmon inshore of 37 km (approximately 20 nm), though some were caught over 55 km (approximately 30 nm) offshore. They noted that, similar to juvenile chum salmon, juvenile sockeye salmon were less abundant than either coho salmon or salmon off the Oregon and Washington coast. Through observations of 883 purse seine net sets off the Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

ESA-listed sockeye salmon from Redfish Lake, Idaho and Ozette Lake, Washington may occur in the offshore action area prior to and during their migration north of their natal watersheds. Tucker et al. (2009) caught juvenile sockeye salmon originating from the Columbia River (inclusive of Redfish Lake sockeye salmon) and the Washington coast (inclusive of Lake Ozette sockeye salmon) from May through August. By October, all juvenile sockeye salmon from these areas had moved north to waters off Vancouver Island, southeast Alaska, and the Gulf of Alaska. Northward migration is thought to be rapid for juvenile sockeye salmon. For example, Tucker et al. (2009) recovered three coded wire tagged sockeye salmon and determined that each had traveled between 1,800 and 2,500 km in under two months at an estimated travel rate of 40-48 km/day.

Walker et al. (2007) recorded the vertical distribution of salmonids in North Pacific Ocean using data storage tags. The authors found that the average depth for sockeye salmon was three meters, though the species was found down to 83 meters.

Steelhead Habitat Use

Steelhead are thought to rely heavily on offshore marine waters for feeding, with high seas tagging programs indicating steelhead make more extensive migrations offshore in their first year than any other Pacific salmonids (Quinn and Myers 2004). Commercial fisheries catch data indicate similar trends as the species is generally encountered in trawl surveys much farther offshore, and in fewer numbers, than Chinook salmon or coho salmon (Moyle et al. 2017). The species spends approximately 1-3 years in freshwater, then migrates rapidly through estuaries, bypassing coastal migration routes of other salmonids, moving into oceanic offshore feeding grounds (Daly et al. 2014; Quinn and Myers 2004). Daly et al. (2014) analyzed NOAA Fisheries pelagic trawl survey data from off the coast of Oregon and Washington that targeted early marine phase juvenile salmonids to learn more about the distribution of steelhead in marine waters. Juvenile steelhead were consistently caught at the westernmost stations (> 55 km from shore) indicating a more offshore distribution for the species. Further, some of the steelhead that were caught in these far offshore waters had only been in saltwater for 1-3 days, indicating a rapid offshore migration (Daly et al. 2014). The results of Daly et al. (2014) were consistent with those presented by Pearcy and Fisher (1990), who found that catches of juvenile steelhead were generally highest at stations located more than 28 km from shore. Light et al. (1989) mapped the ocean distribution of steelhead in the North Pacific using catch per unit effort data from U.S., Canadian, USSR, and Japanese research vessels fishing with purse seines, gill nets, and longlines. As shown in Light et al. (1989), both juvenile and adult steelhead were distributed across the North Pacific throughout the year at distances greater than 50nm, but were in higher abundance closer to the U.S. and Canadian coasts in spring and winter, and more evenly distributed in summer and fall.

Tagging and diet studies indicate that adult and juvenile steelhead are surface oriented, spending most of their time in the upper portions of the water column (Daly et al. 2014). Walker et al. (2007) summarized information from a series of studies off British Columbia looking at the vertical distribution of steelhead and found the species spends 72 percent of its time in the top one meter of the water column, with few movements below seven meters.

Eulachon Habitat Use

Eulachon appear to live near the ocean bottom, or on the continental shelf at depths most commonly of 20 to 200 meters with 96.4 percent of occurrences at depths less than 300 meters. Although eulachon have been documented to occur in deeper water depths (maximum of 625 meters), these instances are rare and have only been observed from Alaskan trawl data which may greatly overestimate eulachon's true maximum depth as fish may become entrained into the nets, either on deployment or recovery (Hay and McCarter 2000). In research trawl surveys, most juvenile eulachon are taken at around 100 meters depth in British Columbia and between 137 and 147 meters off the U.S. West Coast (defined as Washington, Oregon and California)

(Gustafson et al. 2012). This species typically spends three to five years in saltwater before returning to freshwater to spawn.

Green Sturgeon Habitat Use

Green sturgeon spend the majority of their lives in the open ocean where they exhibit migratory behavior not associated with spawning. Lindley et al. (2008) found that green sturgeon migrate annually along the continental shelf from U.S. to Canadian waters in the fall and an apparent return migration in the spring. Although they have been detected on the outer continental shelf at depths up to 200 meters, green sturgeon in marine habitats are primarily found near the ocean bottom at depths of 20–60 meters and from 9.5–16.0 degrees Celsius (Huff et al. 2011). Huff et al. (2011) found that green sturgeon, on average, spent a longer duration in areas with high seafloor complexity, especially where a greater proportion of the substrate consists of boulders.

Estimation of Take Offshore – Salmonids and Eulachon

As described in the preceding sections, juvenile and adult salmon are generally expected to occur on the continental shelf in the action area, but are less likely to occur in portions of the action area past the shelf break (generally defined as the 200 meter depth contour along the west coast of Washington, Oregon, and California). Eulachon are also expected to occur in portions of the action area on the continental shelf, with less abundance at more offshore areas of the action area. Steelhead are expected to occur throughout the action area, as they will occur in the continental shelf portion of the action area during migration and in high seas portions of the action area while foraging. This information was used to determine the likelihood of co-occurrence between Navy explosive activities and each of these species (Table 149). Unless otherwise noted, species are expected to co-occur with all ordnances from the specified explosive bin. Explosions that may occur on the continental shelf are likely to adversely affect salmon (Chinook, chum, coho, and sockeye), steelhead, and eulachon, whereas explosions that occur off the continental shelf (> 50 nautical miles from shore) are only likely to adversely affect steelhead and the following Chinook salmon ESUs: Upper Columbia River spring-run, Snake River spring/summer run, and Upper Willamette River.

Table 149. Expected co-occurrence of ESA-listed fish species with Navy explosive ordnance activity in the offshore action area.

Species	Ordnance co-occurrence
Chinook salmon – all ESUs (except adults from the Upper Columbia River spring-run, Snake River spring/summer-run, and Upper Willamette River ESUs)	Testing: E7, E4
Chinook salmon – Adults from the Upper Columbia River spring-run, Snake River spring/summer run, and Upper Willamette River ESUs	Testing: E1, E3, E4, E7, E8, E11 Training: E1, E2, E5, E10

Species	Ordnance co-occurrence
Coho salmon – all ESUs	Testing: E7, E4
Chum salmon ESU- Columbia River ESU	Testing: E7, E4
Sockeye salmon ESU-Snake River ESU	Testing: E7, E4
Steelhead – Puget Sound, Middle Columbia River, Upper Willamette River, Lower Columbia River, Snake River Basin, Upper Columbia River, Northern California, California Central Valley, Central California Coast, South-Central California Coast	Testing: E1, E3, E4, E7, E8, E11 Training: E1, E2, E5, E10
Southern DPS Green Sturgeon	Testing: E7, E4
Southern DPS eulachon	Testing: E7, E4

After calculating offshore densities using abundance and distribution data (discussed in Section 2.4.2.1), we then used the mean range to effects values for injury and mortality (Table 147) developed for this consultation to calculate an area around each detonation that would result in mortality and injury. Using the range to effects to estimate a cylinder of the affected area is a conservative approach since the actual volume of water affected may be less depending on the location of the explosive in the water column and other environmental factors. We multiplied this area of injury or mortality by the density of each species to determine the number of individual fish from each ESU or DPS that would be expected to be killed or be injured from each detonation (in order to estimate the number of fish injured, the area of mortality was subtracted from the area of injury estimate; this ensured we did not double count). We then multiplied this result by the number of detonations expected for each explosive bin to get a total number of fish (juvenile or adult) that would be expected to die or be injured annually from each explosive bin. Once we calculated the estimated number of injured/killed fish for a single ESU or DPS, we then multiplied that number by the total proportion of hatchery fish with clipped fins, hatchery fish with unclipped fins, and natural (wild) born fish. An example of the calculation is shown below for juvenile fish in the Lower Columbia River ESU of Chinook salmon which are exposed to both Bin E7 and Bin E4:

-Hatchery Intact Adipose (Bin E7): 0.564 km^2 X 377.44 fish per km^2 X 5 explosions X 0.0218 = 23.204 mortalities

-Hatchery Adipose Clipped (Bin E7)Bin E7: 0.5645km^2 X 377.44 fish per km^2 X 5 explosions X 0.712 = 758.51 mortalities

-Natural (Bin E7): 0.5645km^2 X 377.44 fish per km^2 X 5 explosions X 0.267= 284.44 mortalities

-Hatchery Intact Adipose (Bin E4): 0.07065km^2 X 377.44 fish per km^2 X 36 explosions X 0.0218 = 20.93 mortalities

-Hatchery Adipose Clipped (Bin E4): 0.07065km^2 X 377.44 fish per km^2 X 36 explosions X 0.712 = 683.51 mortalities

-Natural (Bin E4): 0.07065km^2 X 377.44 fish per km^2 X 36 explosions X 0.267 = 256.315 mortalities

Results from these calculations for the offshore environment are presented in

Table 150, Table 151, and Table 152 below. Table 150 gives estimates for hatchery fish with an adipose fin intact, Table 151 gives estimates for hatchery fish with an adipose fin clipped, and Table 152 gives estimates for the number of natural (wild) fish. All estimates are the maximum annual number of injured or killed fish, based on a maximum year of NWTT proposed explosive activity levels.

Table 150. Estimated maximum annual number of ESA-listed salmonids (hatchery fish w/adipose fin intact) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook salmon	Adult	Sac River winter run - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Central valley spring run - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	California coastal - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Snake River fall - T	3	388	1	88	4	476
	Juvenile		116	14,197	-	-	116	14,197
	Adult	Snake River spring/summer - T	-	11	-	3	-	14
	Juvenile		31	3,845	-	-	31	3,845

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Adult[1]	Lower Columbia River - T	2	273	-	-	2	273
	Juvenile		44	5,421	-	-	44	5,421
	Adult[1]	Upper Willamette River - T	6	842	3	191	9	1,033
	Juvenile		-	1	-	-	-	1
	Adult	Upper Columbia River spring - E	1	89	-	20	1	109
	Juvenile		15	1,828	-	-	15	1,828
	Adult[1]	Puget Sound - T	1	68	-	-	1	68
	Juvenile		221	27,075	-	-	221	27,075
Coho salmon	Adult	Central California coast - E	0	4	-	-	0	4
	Juvenile		8	943	-	-	8	943
	Adult[1]	S. Oregon/N. California coast - T	2	194	-	-	2	194
	Juvenile		42	5,095	-	-	42	5,095
	Adult	Oregon coast - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Lower Columbia River - T	2	187	-	-	2	187
	Juvenile		22	2,657	-	-	22	2,657
Chum salmon	Adult	Columbia River - T	-	-		-	-	-
	Juvenile		1	109		-	1	109
Sockeye salmon	Adult	Snake River - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
Steelhead --	Adult	South-Central California - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Central California - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	California Central Valley - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult		-	-	-	-	-	-

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Juvenile	Northern California - T	-	-	-	-	-	-
	Adult	Upper Columbia River - E	-	3	-	2	-	5
	Juvenile		1	118	1	39	2	157
	Adult	Snake River basin - T	-	41	-	10	-	51
	Juvenile		4	601	2	140	6	741
	Adult[1]	Lower Columbia River - T	-	57	-	13	-	70
	Juvenile		-	8	-	2	-	10
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Middle Columbia River - T	-	-	-	-	-	-
	Juvenile		1	94	-	26	1	120
	Adult	Puget Sound - T	-	-	-	-	-	-
	Juvenile		1	96	-	22	1	118

[1] Includes mortality and injury estimates for fin-clipped and intact adipose fin hatchery fish for the life stage of this DPS/ESU.

Table 151. Estimated maximum annual number of ESA-listed salmonids (hatchery fish w/adipose clip) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook salmon	Adult	Sac River winter run - E	-	40	-	-	-	40
	Juvenile		10	1,199	-	-	10	1,199
	Adult	Central valley spring run - T	0	41	-	-	0	41
	Juvenile		106	13,005	-	-	106	13,005
	Adult	California coastal - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Snake River fall - T	2	359	1	81	3	440
	Juvenile		101	12,319	-	-	101	12,319

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Adult	Snake River spring/summer - T	-	63	-	14	-	77
	Juvenile		180	22,089	-	-	180	22,089
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		1,441	176,612	-	-	1,441	176,612
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		210	25,759	-	-	210	25,759
	Adult	Upper Columbia River spring - E	1	165	1	37	2	202
	Juvenile		25	3,084	-	-	25	3,084
	Adult[1]	Puget Sound - T	-	-	-	-	-	-
	Juvenile		1,103	135,159	-	-	1,103	135,159
Coho salmon	Adult	Central California coast - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	S. Oregon/N. California coast - T	-	-	-	-	-	-
	Juvenile		14	1,772	-	-	14	1,772
	Adult	Oregon coast - T	-	10	-	-	-	10
	Juvenile		4	532	-	-	4	532
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		632	77,508	-	-	632	77,508
Chum salmon	Adult	Columbia River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
Sockeye salmon	Adult	Snake River - E	-	1	-	-	-	1
	Juvenile		-	29	-	1	-	30
Steelhead	Adult	South-Central California - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Central California - T	-	10	-	2	-	12
	Juvenile		4	553	2	155	6	708
	Adult	California Central Valley - T	-	10	-	3	-	13
	Juvenile		9	1,365	5	318	14	1,683
	Adult		-	-	-	-	-	-

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Juvenile	Northern California - T	-	-	-	-	-	-
	Adult	Upper Columbia River - E	-	14	-	7	-	21
	Juvenile		4	586	3	193	7	779
	Adult	Snake River basin - T	1	203	1	47	2	250
	Juvenile		19	2,813	10	654	29	3,467
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		7	1,021	4	232	11	1,253
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Middle Columbia River - T	-	1	-	-	-	1
	Juvenile		2	379	1	107	3	486
	Adult	Puget Sound - T	-	-	-	-	-	-
	Juvenile		1	94	-	21	1	115

Table 152. Estimated maximum annual number of ESA-listed salmonids (natural fish) that would die or be injured by explosive activities in the offshore environment. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook salmon	Adult	Sac River winter run - E	-	4	-	-	-	4
	Juvenile		10	1,171	-	-	10	1,171
	Adult	Central valley spring run - T	1	67	-	-	1	67
	Juvenile		38	4,649	-	-	38	4,649
	Adult	California coastal - T	1	127	-	-	1	127
	Juvenile		63	7,662	-	-	63	7,662
	Adult	Snake River fall - T	2	296	1	67	3	364
	Juvenile		28	3,436	-	-	28	3,436
	Adult		2	339	1	77	3	416

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Juvenile	Snake River spring/summer - T	41	4,997	-	-	41	4,997
	Adult	Lower Columbia River - T	2	208	-	-	2	208
	Juvenile		540	66,159	-	-	540	66,159
	Adult[1]	Upper Willamette River - T	2	273	1	62	3	335
	Juvenile		54	6,629	-	-	54	6,629
	Adult	Upper Columbia River spring - E	1	76	0	17	1	93
	Juvenile		19	2,325	-	-	19	2,325
	Adult	Puget Sound - T	1	98	-	-	1	98
	Juvenile		92	11,302	-	-	92	11,302
Coho salmon	Adult	Central California coast - E	0	22	-	-	0	22
	Juvenile		7	899	-	-	7	899
	Adult	S. Oregon/N. California coast - T	1	161	-	-	1	161
	Juvenile		146	17,844	-	-	146	17,844
	Adult	Oregon coast - T	14	1,672	-	-	14	1,672
	Juvenile		480	58,855	-	-	480	58,855
	Adult	Lower Columbia River - T	5	635	-	-	5	635
	Juvenile		57	7,035	-	-	57	7,035
Chum salmon	Adult	Columbia River - T	-	5	-	-	-	5
	Juvenile		8	977	-	-	8	977
Sockeye salmon	Adult	Snake River - E	-	-	-	-	-	-
	Juvenile		-	2	-	-	-	2
Steelhead	Adult	South-Central California - T	-	2	-	1	-	3
	Juvenile		-	67	-	32	-	99
	Adult	Central California - T	-	6	-	1	-	7
	Juvenile		1	212	1	59	2	271
	Adult	California Central Valley - T	-	4	-	1	-	5
	Juvenile		4	537	2	125	6	662

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Adult	Northern California - T	-	18	-	5	-	23
	Juvenile		5	700	3	177	8	877
	Adult	Upper Columbia River - E	-	5	-	3	-	8
	Juvenile		1	170	1	56	2	226
	Adult	Snake River basin - T	-	27	-	6	-	33
	Juvenile		4	681	2	158	6	839
	Adult	Lower Columbia River - T	-	33	-	7	-	40
	Juvenile		2	300	1	68	3	368
	Adult	Upper Willamette River - T	-	7	-	2	-	9
	Juvenile		1	120	1	45	2	165
	Adult	Middle Columbia River - T	-	13	-	4	-	17
	Juvenile		2	348	1	98	3	446
	Adult[1]	Puget Sound - T	-	49	-	11	-	60
	Juvenile		12	1,873	6	422	18	2,295

[1] Includes mortality and injury estimates for natural and hatchery fish (intact and clipped numbers).

Eulachon appear to live near the ocean bottom, or on the continental shelf at depths most commonly of 20 to 200 meters (NMFS 2006b). Because most eulachon are expected to occur on the continental shelf, we assessed impacts to eulachon for the detonations that may occur over the continental shelf and in water deep enough where they may be affected by explosions as determined by the range to injury and mortality values (explosions in bins E4 and E7). We do not expect the eulachon to be affected by explosions off the shelf break due to the depth at which the animals will occur in these areas (near the ocean bottom) and sound propagation loss.

To estimate the number of eulachon that may be killed or injured from Navy explosive activities in the offshore environment, we used a similar methodology as described for estimating salmonid take in the offshore environment. First, we determined the average density of eulachon in the offshore environment. Next, we used distribution data from NMFS (2015a) which estimated the southern DPS of eulachon has a marine distribution area of 1,183,304 km^2. As stated in Section 2.4.2, when determining density, we assumed equal distribution throughout their area of habitat since information does not suggest we should expect higher densities of these species in some areas versus others. We then multiplied this density by the area of mortality and injury for each explosive bin that would be expected to co-occur with the species (in order to estimate the number of fish injured, the area of mortality was subtracted from the

area of injury estimate; this ensured we did not double count). The latest estimate of the population abundance of the southern DPS of eulachon was 18,796,090 spawners estimated in the Columbia River and Fraser River from 2014 to 2018. Because we do not have estimates of eulachon abundance in marine waters, the number of spawners in the Columbia River and Fraser River were used as a proxy for abundance in the oceanic environment. We multiplied the number of returning adults by the average number of years the species spends at sea before returning to spawn, in order to account for all age classes of fish that would be expected in the oceanic environment (i.e., four years for eulachon). This resulted in a density of 63.54 fish per square kilometer. We recognize that since this methodology is based on the number of returning adults, it does not account for individuals that die before returning to spawn. However, this does not inhibit our ability to accurately assess jeopardy and determine whether or not to expect any population level effects from this action because we are assessing jeopardy and the potential for any population level effects by comparing mortality from this action to the number of returning adults (generally how eulachon abundance and trends are tracked). Results of these calculations are presented in Table 153.

Table 153. Estimated maximum annual number of ESA-listed eulachon from the Southern DPS that would die or be injured by explosive activities in the offshore environment.

Species	Testing		Training		Total	
	Mortality	Injury	Mortality	Injury	Mortality	Injury
Eulachon - Southern DPS	341	2,520	0	0	341	2,520

Estimation of Take Offshore – Green Sturgeon

The first step in estimating take of Southern DPS green sturgeon from NWTT explosives was to estimate the density of this population in areas where detonations could occur. We estimated Southern DPS green sturgeon density in the offshore portion of the action area based on unpublished acoustic tag detection data from the NMFS Northwest Fisheries Science Center and population abundance estimates from (Mora et al. 2018). For details on the approach used to estimate green sturgeon offshore density refer to Section 2.4.2.1.

Mine countermeasure and neutralization testing could occur greater than three nautical miles from shore within the Quinault Range Site (but not in the Olympic Coast National Marine Sanctuary), or greater than 12 nautical miles from shore elsewhere in the offshore area. Mine countermeasure and neutralization testing using explosives within bin E7 will be no closer to shore than six nautical miles in the Quinault Range Site. For this analysis, we conservatively assume that all detonations would occur within the Quinault Range at a distance from shore where green sturgeon are most likely to be found (i.e., depths of less than 60 meters).

As part of the proposed action, a maximum of five E7 charges and 36 E4 charges could be detonated in the offshore shore portion of the action area per year as part of mine countermeasure and neutralization testing events. The Navy's proposed action also includes a cap on the seven-year total number of offshore explosives for this activity as follows: 108 E4 and 15 E7 explosives. The Navy has also proposed seasonal mitigation to limit the number of explosives that could occur within 20 nautical miles from shore from October through June: 20 E4 and 3 E7 annually; 60 E4 and 9 E7 over seven years. For our exposure analysis, we conservatively assume that the maximum allowed level of explosives would occur during the one annual event conducted during the October through June seasonal time frame, when Southern DPS green sturgeon densities are expected to be higher (as compared to the July through September time frame).

To quantitatively estimate the effects of explosives on Southern DPS green sturgeon, we use the same general approach as described above for salmonids. The mean range to onset mortality is estimated at 424 meters for bin E7 explosives and 150 meters for bin E4 explosives. Based on these ranges, the area of mortality is 0.5645 km^2 for bin E7 and 0.07065 km^2 for bin E4. To estimate the maximum annual number of Southern DPS green sturgeon mortalities, we multiplied the area of mortality by the estimated density (from Section 2.4.2.1 above) for each seasonal time frame by the number of annual explosions for each seasonal time frame (assuming maximum proposed for October through June when estimated density is higher), as follows:

October through June

Bin E4: 0.07065 km^2 X 3.85 fish per km^2 X 20 explosions = 5.44 mortalities

Bin E7: 0.5645 km^2 X 3.85 fish per km^2 X 3 explosions = 6.52 mortalities

July through September

Bin E4: 0.07065 km^2 X 0.53 fish per km^2 X 16 explosions = 0.60 mortalities

Bin E7: 0.5645 km^2 X 0.53 fish per km^2 X 2 explosions = 0.60 mortalities

Combining the two bins and two seasonal time frames, we get a total estimate of 13.2 Southern DPS green sturgeon mortalities per year as a result of NWTT offshore explosives during a maximum year of mine countermeasure and neutralization testing. Since the proposed seven year mine countermeasure and neutralization testing explosive activity level is three times the maximum year level, we multiply the maximum annual estimate by three to get the seven year total Southern DPS green sturgeon mortality estimate as follows: 13.2 X 3 = 39.6. We divide the seven year total estimate by seven to arrive at an average annual mortality estimate of 5.7 Southern DPS green sturgeon.

For onset to green sturgeon injury, as explained above for salmonids, we use the ranges to effects based on the acoustic injury criteria of >207 dB peak re 1 µPa for fishes with a swim bladder.

From Table 147 above, these are which are 4,259 meters for bin E7 explosives and 1,829 meters for bin E4 explosives.

Based on these ranges, the area of injury (excluding the mortality zone) is 56.392 km^2 for bin E7 and 10.433 km^2 for bin E4. To estimate the maximum annual number of Southern DPS green sturgeon injuries, we multiplied the area of injury by the estimated density (from Section 2.4.2.1 above) for each seasonal time frame by the number of annual explosions for each seasonal time frame (assuming maximum for October through June when estimated density is higher), as follows:

October through June

Bin E4: 10.433 km^2 X 3.85 fish per km^2 X 20 explosions = 803.34 injuries

Bin E7: 56.392 km^2 X 3.85 fish per km^2 X 3 explosions = 651.33 injuries

July through September

Bin E4: 10.433 km^2 X 0.53 fish per km^2 X 16 explosions = 88.47 injuries

Bin E7: 56.392 km^2 X 0.53 fish per km^2 X 2 explosions = 59.78 injuries

Combining the two bins and two seasonal time frames, we get a total estimate of 1,602.9 Southern DPS green sturgeon injuries per year as a result of NWTT offshore explosives during a maximum year of mine countermeasure and neutralization testing. Since the proposed seven year mine countermeasure and neutralization testing explosive activity level is three times the maximum year level, we multiply the maximum annual estimate by three to get the seven year total Southern DPS green sturgeon injury estimate as follows: 1,602.9 X 3 = 4,808.7. We divide the seven year total estimate by seven to arrive at an average annual injury estimate of 687.0 Southern DPS green sturgeon.

Some proportion of exposures resulting in injury would likely also produce TTS, since the estimated ranges to TTS from explosives are smaller than ranges to injury for fish (see Table 147 and Table 148 for comparison). Although we cannot quantify the effects, we also anticipate additional Southern DPS green sturgeon would be exposed to explosives at levels resulting in a behavioral response or physiological stress response.

The mortality and injury estimates above include both subadult (3-20 years and 60-165 cm length) and adult (greater than 165 cm in length and older than 20 years) Southern DPS green sturgeon. We assume the proportion of mortalities and injuries that would be adults/subadults is roughly equivalent to the proportional distribution for the population as a whole; i.e., from Mora et al. (2018), about 16 percent adults and 84 percent subadults.

Underwater explosions also may reduce available prey items by either directly killing or injuring green sturgeon benthic prey or by impacting the structure and complexity of the benthic environment where such prey occur. The secondary effects on green sturgeon prey and benthic habitat are discussed in more detail in the critical habitat effects analysis section below (Section

8.3.3.3). With benthic associated species such as green sturgeon, there is also the possibility that large explosions could kick-up and disturb bottom sediments, thus exposing more benthic prey, which, in turn, could attract green sturgeon to the area following a detonation. While this could temporarily increase the availability of prey, it could also make foraging green sturgeon more vulnerable to future explosions in the same area as the initial explosion.

As discussed previously, the estimates of Southern DPS green sturgeon mortality and injury are based on the following conservative assumptions: 1) the proposed offshore mine countermeasure and neutralization events (two per year) would overlap in both time and space with the maximum densities of Southern DPS green sturgeon in the action area, and 2) the maximum number of proposed detonations for the October through June seasonal time frame (i.e. annual maximum and seven year total) would be used. Thus, we anticipate that these estimates would be somewhat lower if some portion of detonations occurred farther offshore (i.e., in water depths greater than 80 meters) or during a time frame when estimated green sturgeon densities in the Quinault Range would be lower than those estimated above. The Navy has also agreed to the following conservation measure: *The Navy will, to the maximum extent practicable, conduct mine countermeasure and neutralization testing from July 1 through September 30 when operating within 20 nautical miles shore in the Marine Species Coastal Mitigation Area*. As discussed above, densities of Southern DPS green sturgeon are anticipated to be significantly lower from July through September as compared to other months when this activity would likely occur (e.g., five times lower based on our density estimation approach). Although the Navy has indicated several reasons ((i.e., platform availability, range availability, and weather) why conducting this activity from July through September may not always be practicable, this conservation measure would still likely result in lower levels of Southern DPS green sturgeon injury and mortality than those estimated by our quantitative analysis above.

As noted previously, our range to effects for fish injury are based on the more conservative acoustic injury criterion of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors, and others described above, likely result in conservatively high estimates of Southern DPS green sturgeon fish injury from Navy explosives.

In addition, the ranges to effects (i.e., mortality and injury) for fish from explosives were developed based on the conservative assumption that the zone of impact would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. Given their large size, subadult and adult green sturgeon would likely be less sensitive to the effects of explosives than the species and life stages these criteria were based on. In Section 8.2.3.2 below, we discuss how Southern DPS green sturgeon would likely respond to the effects of explosives.

8.2.3.1.2 Inland Waters - Exposure Analysis

Detonations in inland waters (Puget Sound) will occur in Crescent Harbor and Hood Canal. For NWTT Phase III training, the Navy has proposed to conduct three events per year involving Mine Neutralization activities in both the Hood Canal and Crescent Harbor EOD range sites. The total ordnance proposed per year from these 12 events would be six 2.5-lb. charges (E3) and 36 charges under one pound (E0). One E3 charge and up to six E0 charges would be used per event.

In Hood Canal, charges are generally placed a few feet (1-3) above the seafloor, in approximately 27 feet of water (Andrea Balla-Holden, Navy, personal communication to J. Molineaux, NMFS, February 20, 2020). However, surface detonations are possible. In Crescent Harbor, E3 charges are generally placed in water depths greater than 40 feet, in areas with no bottom vegetation (Andrea Balla-Holden, Navy, personal communication to J. Molineaux, NMFS, February 26, 2020). At the Crescent Harbor EOD Range, the Navy would conduct explosive activities at least 1,000 meters from the closest point of land to avoid or reduce impacts on fish (e.g., juvenile Chinook salmon) in nearshore habitat areas.

Below we present information on the ESA-listed fish species considered in this Opinion that are likely to occur in Puget Sound, their likelihood of co-occurring with detonations in Crescent Harbor and Hood Canal, and the methods we used to estimate the number of fish that may be killed or injured by Navy activities. Fish species considered in this Opinion that are not expected to occur in Puget Sound (e.g., Lower Columbia River Chinook salmon ESU) are not discussed in this section (i.e., we anticipate inland use of explosives would have no effect on these species).

Puget Sound Chinook salmon – Inland Habitat Use

Juvenile and adult Puget Sound Chinook salmon would likely be affected by Navy EOD activities at both Crescent Harbor and Hood Canal. Crescent Harbor is located at the north end of Saratoga Passage, just west of Skagit Bay. There is no known Chinook salmon spawning habitat in streams flowing into Crescent Harbor (http://apps.wdfw.wa.gov/salmonscape/map.html), but Puget Sound Chinook salmon from the Skagit, Stillaguamish, and Snohomish Rivers (and to a lesser degree, rivers from southern Puget Sound) would be expected to occur in the Harbor (NMFS 2012a). Additionally, an unknown number of Puget Sound Chinook salmon which mature solely in the Salish Sea and do not migrate into the North Pacific Ocean (i.e., blackmouths), may spend at least a portion of their time foraging in Crescent Harbor. Survey and research sampling within Crescent Harbor have indicated juvenile Puget Sound Chinook salmon use both the intertidal and offshore areas of the Harbor with smaller individuals generally found in intertidal habitat and larger individuals found in the offshore (NMFS 2015c). This is consistent with Puget Sound Chinook salmon life history information which suggests that most juveniles enter estuaries and occupy nearshore habitats primarily during the spring and early summer, but by mid-summer, most Chinook salmon have migrated to more offshore habitats (Duffy et al. 2010). Beamish et al. (1998) studied intra-annual changes in the abundance of Chinook salmon in Puget Sound and collected the majority of juveniles in rope trawl tows at

depths ranging from 0 to 15 meters. Beamer et al. (2003) studied wild Chinook salmon pocket estuary, nearshore, and offshore habitat use in Skagit Bay. The authors found juvenile Chinook salmon are over 100 times more abundant in pocket estuary habitat (0.5 meter max depth) and 10 times more abundant in nearshore habitat (2.8 meter max depth) than in more offshore areas adjacent to nearshore habitat. Based on the range to injury and mortality for the charges proposed for use in Crescent Harbor and the distance of the site from these nearshore habitats (EOD range is in waters greater than 40 feet deep and 1,000 meters offshore), we do not expect injurious sound levels to be able to reach nearshore and pocket estuary habitats (as described in (Beamer et al. 2003) in Crescent Harbor that would be expected to hold the highest concentrations of fish.

As stated in Section 2.4.2.2, to estimate the density of juvenile Puget Sound ESU Chinook salmon in Crescent Harbor we used juvenile density information from unpublished data sampled in southwest Skagit Bay, directly adjacent to Crescent Harbor, by the Northwest Fisheries Science Center. Because we do not know when the Navy will be conducting EOD activities, we used the average annual maximum densities from 2003 to 2019 equating to 446 unmarked and 406 marked fish/square kilometer (C. Greene, NMFS, personal communication to J. Molineaux, NMFS, August 7, 2020).

Also, as stated in Section 2.4.2.2, information suggests that adult Chinook salmon density in Crescent Harbor is likely much lower than would be expected in some other areas and habitat types in the Sound (e.g., river mouths, deltas, and estuaries of the Skagit and Snohomish Rivers) because there are no documented spawning rivers in the immediate vicinity of the EOD range. Therefore, adult salmon are unlikely to stage in large numbers in the Harbor, but available information suggests that very low numbers of migrating adults could occur within the range to effects for underwater detonations for very short periods of time (though Crescent Harbor is not a direct migratory route from the ocean to any Chinook salmon spawning rivers). Given the very low densities of adults that may occur in the area, the short amount of time any individuals would be likely to spend in Crescent Harbor, and the low number of explosion events that will occur throughout the year, we do not expect more than one adult Puget Sound Chinook salmon may be injured or killed each year from Navy explosive activities at Crescent Harbor.

Hood Canal has extant populations of Puget Sound Chinook salmon in the Skokomish River watershed and Mid Hood Canal region (including spawning populations in the Hamma, Duckabush, and Dosewallips watersheds) (Ford 2011a). Hood Canal Chinook salmon typically spawn mid-September to late October (Commission 2003). All juvenile Chinook salmon emigrating from or adults immigrating to these watersheds must pass by the Hood Canal EOD site. Additionally, an unknown number of Puget Sound Chinook salmon which mature solely in the Salish Sea and do not migrate into the North Pacific Ocean, are likely to spend at least a portion of their time foraging in proximity to the Hood Canal EOD range (e.g., (Tribe and Wildlife 2007)). We do not have information on the marine residence times of individual Hood Canal chinook salmon, but based on life history information, we assume juvenile Chinook salmon spend significantly more time rearing in Hood Canal nearshore areas than steelhead or

chum salmon. Generally, Puget Sound Chinook salmon juveniles emigrate from freshwater natal areas to estuarine and nearshore habitats from January through April as fry, and from April through early July as larger subyearlings (NMFS 2011c).

As indicated in the *Status of Listed Resources* section of this opinion, the number of natural origin spawners returning to Hood Canal Rivers averaged 389 individuals from 2012-2016. Over the same time period, hatchery origin spawners averaged 1,275 individuals per year. NMFS (2020c) calculated that this number of returning spawners would result in a total of 133,094 outmigrants (i.e., juveniles) per year with 101,980 originating from hatcheries and 31,114 being natural origin.

In order to minimize impacts to adult Puget Sound Chinook salmon from Hood Canal, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (>0.5-2.5 pounds), and only conduct training with one ounce or less charges during August, September, and October. Because the Navy could not rule out conducting explosions using these larger size charges during the adult migration and spawning period and in order to be conservative (i.e., assess the maximum effect to the species which could be considered reasonable), we assumed E3 detonations could occur during the adult migration period.

To estimate the number of Chinook salmon that we would expect to be killed or injured from EOD activity at Hood Canal, we first estimated the average density of Chinook salmon of each life stage in Hood Canal. We do not have EOD range-specific density information for Chinook salmon at Hood Canal. Since there are no documented Chinook salmon spawning rivers in the immediate vicinity of the Hood Canal EOD range, adult and juvenile Chinook salmon are unlikely to stage in large concentrations near the range. This information suggests Chinook salmon density at the EOD ranges is likely lower than would be expected in some other areas and habitat types in Hood Canal (e.g., Hood Canal salmon spawning river mouths/estuaries). Most individuals that will occur in close proximity to the range would be expected to be migrating through the range, though some juveniles could reside in higher numbers in nearshore areas within the injury/mortality range of the explosions. In order to be conservative, we assumed equal distribution of each life stage throughout the Canal since information does not suggest we should expect higher densities of these species at the EOD ranges than in other areas of the Canal. In order to derive our density estimates, we divided the number of adult or juvenile fish expected in the Hood Canal[22] by the area of the Canal (Hood Canal has a surface area of

[22] Without better information on residence time of outmigrating juvenile or returning adult Chinook salmon in Hood Canal and since Chinook are thought to spend significantly more time rearing in estuarine and nearshore waters before migrating into open water habitats (months instead of days), we assumed all Chinook salmon originating from and returning to Hood Canal rivers could reside in the Canal concurrently. This likely overestimates Chinook salmon abundance in the Canal since some proportion of juveniles will spend less time in the Canal than others and some will die from other causes (e.g., predation) while migrating through the Canal. Additionally, adult migration is

358.4 km^2). We then multiplied this density by the area of mortality and injury as estimated by the Navy (in order to estimate the number of fish injured, the area of mortality was subtracted from the area of injury estimate; this ensured we did not double count).

We recognize that this methodology likely overestimates the number of adult Puget Sound Chinook salmon from Hood Canal that will die or be injured from Navy EOD activities. Most notably, our analysis and resulting take estimates assume all detonations (E3 and E0 charges) will occur when adult Puget Sound Chinook salmon are in the Canal. While we did this to be conservative and to analyze the highest potential effect of the activity on ESA-listed species that could be considered reasonable, it is also possible that EOD activity may be spread more evenly throughout the year and occur during times when the species is not in Hood Canal or near the EOD range, particularly since the Navy will only conduct detonations with the larger charges during their migration period if necessitated by readiness requirement. If the Navy were to conduct some or all EOD activities during times of the year when adult Puget Sound Chinook salmon are not present, the number of individuals that would die or be injured as a result of Navy EOD activities would be expected to be much lower. Also, this methodology does not consider habitat selection within the Hood Canal by the species. Adult fish in close proximity to the EOD range that may be affected would only be migrating through the area since there are no spawning rivers in immediate proximity to the range and would not be staging to enter freshwater spawning rivers at this location. Further, juveniles are expected at lower densities in the EOD range than in other areas of the Canal, where individuals of this life stage would be more likely to congregate (e.g., tidal channels, mudflats, marshes, and eelgrass meadows). These habitat types are not present in the Hood Canal EOD range. Additionally, it is worth noting that Navy monitoring of previous similar explosive activity at Hood Canal have never documented fish kills of the magnitude estimated to occur. However, Navy monitoring may not detect all fish that would die from an explosive event as some may not float to the surface of the water in order to be detected by visual observation and some may be captured by foraging birds prior to detection. Additionally, injured fish that may later die would not be expected to be detected as they also may not immediately float to the surface of the water to be detected by visual observation. Over multiple years (2002-2018) of monitoring EOD underwater detonations at both Hood Canal and Crescent Harbor, most observed mortalities were Pacific herring. There have been no observed mortalities of salmonids (S. Kotecki, Navy, personal communication to NMFS, July 24, 2020).

Puget Sound Steelhead – Inland Habitat Use

Juvenile and adult Puget Sound steelhead would likely be affected by Navy training activities using explosives at both Crescent Harbor and Hood Canal. Puget Sound steelhead abundance in Crescent Harbor is likely very low. Steelhead are seldom observed in Puget Sound at any life stage (NMFS 2012a). Steelhead do not spawn in the vicinity of Whidbey Island, including

staggered such that some returning fish will have likely already entered freshwater before others have entered Hood Canal. This methodology results in a conservative estimate of take.

streams flowing into Crescent Harbor (http://apps.wdfw.wa.gov/salmonscape/map.html). As mentioned previously, steelhead do not rear in estuarine or nearshore areas for long periods of time, but migrate quickly through these areas into oceanic waters of the North Pacific Ocean. Puget Sound steelhead generally migrate to the ocean from mid-April through late May when they are typically two to three years old. At this age and size, they are not dependent on shallow nearshore areas, and they are well dispersed throughout Puget Sound. Therefore, any outmigrating steelhead occurring in Crescent Harbor, would likely only be present for a short period of time as they migrate towards open water. Similarly, Crescent Harbor is not in direct route to steelhead spawning rivers on the eastern side of Skagit Bay from oceanic waters, so any adults that may migrate through Crescent Harbor would likely only occur in the harbor for a brief period of time. This information leads us to conclude that steelhead density in Crescent Harbor is expected to be much lower than in other locations in Puget Sound (e.g., in marine waters directly offshore from their natal rivers). However, a few individuals could occur in the Harbor while migrating.

We do not have information available to estimate the density of juvenile or adult steelhead that may migrate through the EOD site at Crescent Harbor. However, the information presented above suggests that very low numbers of migrating juveniles and adults could occur within the range to effects for underwater detonations for very short periods of time. Given the very low densities of juveniles and adults that may occur in the area, the short amount of time any individuals would be likely to spend in Crescent Harbor, and the low number of explosion events that will occur throughout the year, we do not expect more than one juvenile and one adult Puget Sound may be injured or killed each year from Navy explosive activities at Crescent Harbor.

Steelhead spawn in rivers flowing into Hood Canal, though not in rivers adjacent to the Hood Canal EOD range. Immigrating and emigrating steelhead must pass by the Hood Canal EOD range on their way to their natal rivers to spawn or on their way out to oceanic waters in the North Pacific to mature. Typically, Puget Sound steelhead juveniles emigrate from their natal rivers as two-year olds from March through June (NMFS 2011c). Moore et al. (2010) found that acoustically tagged steelhead smolts from Hood Canal rivers migrated from their natal river mouths to areas outside of the Hood Canal over an average of 15 to 17 days. Therefore, based on emigration timing and residence time in Hood Canal, most steelhead smolts will have left Hood Canal by mid-July.

As indicated in the *Status of Listed Resources* section of this opinion, the number of natural-origin and hatchery-production spawners averaged 1,146 individuals from 2012 to 2016 (NMFS 2020c). The number of spawners were then used to calculate a total of 130,348 outmigrants per year (NMFS 2020c).

We estimated the number of juvenile Puget Sound steelhead that may be injured or killed from detonations at the Hood Canal EOD site using a similar methodology described for estimating offshore take. As stated in Section 2.4.2.2, to estimate steelhead juvenile density at the EOD site

in Hood Canal, we first determined the average number of fish leaving freshwater from Hood Canal rivers per day by dividing the annual number of outmigrating juveniles by the number of days in the migration period (March through June; 122 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (17 days, as described above). This results in the average number of juvenile Puget Sound steelhead we would expect in the Canal on a given day during the middle portion of the out-migration period (during first and last 17 days of the out-migration period, fewer individual steelhead would be expected in the Canal). We then divide this number by the area of Hood Canal (358.4 km^2) to determine juvenile steelhead density during the outmigration period. To be conservative, we assumed all detonations will occur during the outmigration period as the Navy is unable to predict when detonations will occur. For adults, we used unpublished redd count data shared by the Northwest Fisheries Science Center as a surrogate to determine the maximum total percentage of adults that may be in Hood Canal at any time (B. Berejikian, NMFS, personal communication to J. Molineaux, NMFS, July 28, 2020). To determine the maximum proportion of adult steelhead in Hood Canal, we used the maximum monthly percentage of redd counts in Hood Canal for years 2007 to 2016 averaged across all major Hood Canal rivers and tributaries. This resulted in a maximum proportion of 51.5 percent of steelhead adults in Hood Canal at any period of time throughout the year. We then multiplied this percentage by the total annual abundance of adult steelhead that could be present in Hood Canal[23] to get a total maximum abundance of adults that could be present in Hood Canal during the year. This number was then divided by the area of Hood Canal to estimate adult density in the Canal on a given day during the run. This resulted in an adult density of 1.6467 per km^2 steelhead.

Hood Canal Summer-run Chum salmon – Inland Habitat Use

The only ESU of chum salmon that is likely to be adversely affected by NWTT inland water explosive stressors are Hood Canal summer-run chum salmon. Hood Canal summer-run chum salmon juveniles migrate from freshwater into estuary habitat generally from the first week in February through the second week in April. Estimated peak emergence occurs around March 22 (Wildlife and Tribes 2000). Upon initially arriving in the estuary, chum salmon inhabit nearshore waters, at depths ranging between 1.5 and 5 meters (Wildlife and Tribes 2000). At this initial stage, they are thought to concentrate in the top few meters of the water column, forming loose aggregations during daylight hours. Once fry reach a size of approximately 45 to 50 millimeters, they tend to move rapidly into deeper offshore areas, migrating approximately 7 to 14 km/day (Wildlife and Tribes 2000). At this speed and growth rate, Hood Canal chum salmon exit the Canal approximately 14 days after entering seawater, with approximately 90 percent of the annual population exiting by late April, on average (Wildlife and Tribes 2000).

[23] These are derived from abundance numbers from all populations of Puget Sound steelhead found in Hood Canal. See Section 8.2.3.1 for more details.

Returning adults begin to arrive in Hood Canal in early August, and are thought to stage in front of their stream of origin for approximately ten to twelve days. During this time, they are generally found in subestuary habitats (e.g., tidal channels, mudflats, marshes, and eelgrass meadows) of the Hood Canal. Migration into freshwater spawning grounds generally occurs from late August to late October (Wildlife and Tribes 2000). The number of natural origin spawners in Hood Canal averaged 20,809 individuals from 2013 to 2017, and the number of hatchery origin spawners averaged 1,452 individuals over the same period (NMFS 2020c). NMFS (2020c) estimated this number of spawners would produce 3,255,599 outmigrants annually.

Based on the timing of returning adults and out-migrating juveniles, we do not expect Hood Canal summer chum salmon of any life stage to be present in the Hood Canal portion of the action area from May through July, or November through January. From February through April we would expect juveniles to be present in the action area as they rear in Hood Canal, and then migrate towards the main basin of Puget Sound and into the North Pacific Ocean. Additionally, we expect adults to be present in the Hood Canal portion of the action area from August through October. If Navy EOD activities were to occur during either of these time periods, Hood Canal summer-chum salmon could be affected by underwater explosions.

In order to minimize impacts to juvenile Hood Canal summer-run chum salmon, the Navy will not conduct any of the planned larger size (E3; >0.5 to 2.5 pounds) detonations at the Hood Canal site during February, March, and April (the juvenile outmigration period of juvenile chum salmon Wildlife and Tribes 2000). To be conservative, we assumed all (LIMPET) charges will occur during the outmigration and spawning period as the Navy is unable to predict when these detonations will occur (as noted in the Navy's BA Navy 2020e). However, E3 charges will not be used during the outmigration period[24]. Therefore, when estimating the number of juvenile and Hood Canal summer-run chum salmon that may be killed or injured, we assumed there would be no co-occurrence with the use of E3 detonations.

In order to minimize impacts to adult Hood Canal summer-run chum salmon, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (>0.5 to 2.5 pounds), and only conduct training with one ounce or less charges during August, September, and October, the adult migration period for Hood Canal summer-run Chum salmon. Because the Navy could not rule out conducting explosions using these larger size charges during the adult migration period and in order to be conservative (i.e.,

[24] As stated in Navy. 2020e. Northwest Testing and Training Biological Assessment to Support ESA Section 7 Consultation with the National Marine Fisheries Service. Navy Systems Commands (Naval Sea Systems Command and Naval Air Systems Command), as Represented by
Commander, Naval Sea Systems Command., the Navy will not detonate E3 charges in Hood Canal from February to April and August to October. This period covers the entire spawning and outmigration periods for the

assess the maximum effect to the species which could be considered reasonable), we assumed E3 detonations could occur during the adult migration period.

We estimated the number of Hood Canal summer-run Chum salmon (juvenile and adult) that may be injured or killed from detonations in inland waters using a similar methodology as described for estimating offshore take. To estimate juvenile chum salmon density at the EOD site in Hood Canal, we first parsed out the number of outmigrants from natural origin vs. hatchery origin using the average percentage given in Table 79. Next, we determined the average number of fish leaving freshwater from Hood Canal rivers per day by dividing the annual number of outmigrating juveniles by the number of days in the migration period (74 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (14 days, as described above). This results in the average number of juvenile Hood Canal summer-run chum salmon we would expect in the Canal on a given day during the middle portion of the out-migration period (during first and last 14 days of the out-migration period, fewer individual salmon would be expected in the Canal). We then divide this number by the area of Hood Canal (358.4 km^2) to determine summer-run chum salmon density during the outmigration period. To be conservative, we assumed all bin E0 detonations will occur during the outmigration period as the Navy is unable to predict when these detonations will occur (as noted above, E3 charges will not be used during the outmigration period). For adults, we did not have information to estimate the amount of time each fish would be expected to reside in Hood Canal before entering freshwater to spawn. Therefore, we assumed all adults from populations of Hood Canal summer-run chum salmon found within Hood Canal (excluding Strait of Juan de Fuca Populations) could occur in the Canal on a given day. We then divided this number by the area of Hood Canal to estimate adult density in the Canal on a given day during the run. Once we had juvenile and adult densities, we multiplied these numbers by the area of injury and mortality for each detonation, then multiplied this number by the number of detonations (in order to estimate the number of fish injured, the area of mortality was subtracted from the area of injury estimate; this ensured we did not double count).

We recognize that this methodology likely overestimates the number of Hood Canal summer-run chum salmon that will die or be injured from Navy EOD activities. Most notably, our analysis and resulting take estimates assume all LIMPET charge detonations occur when juvenile Hood Canal summer-run chum salmon are in the Canal. Additionally, our analysis and resulting take estimates assume all detonations (bin E3 and E0 charges) will occur when adult Hood Canal summer-run chum salmon are in the Canal. While we did this to be conservative and to analyze the highest potential effect of the activity on ESA-listed species that could be considered reasonable, it is also possible that EOD activity may be spread more evenly throughout the year and occur during times when the species is not in Hood Canal or near the EOD range. If the Navy were to conduct some or all EOD activities during times of the year when Hood Canal summer-run chum salmon are not present, the number of individuals that would die or be injured as a result of Navy EOD activities would be expected to be much lower or zero. Also, this

methodology does not consider habitat selection within the Hood Canal by summer-run chum salmon adults or juveniles. Adult fish in close proximity to the EOD range that may be affected would only be migrating through the area since there are no spawning rivers in immediate proximity to the range and would not be staging to enter freshwater spawning rivers at this location. Further, juveniles are expected at lower densities in the EOD range than in other areas of the Canal, where individuals of this life stage would be more likely to congregate (e.g., tidal channels, mudflats, marshes, and eelgrass meadows). These habitat types are not present in the Hood Canal EOD range. Additionally, it is worth noting that Navy monitoring of previous similar underwater explosive activities at Hood Canal have never documented fish kills of the magnitude estimated to occur NMFS (2015a). However, Navy monitoring may not detect all fish that would die from an explosive event as some may not float to the surface of the water in order to be detected by visual observation and some may be captured by foraging birds prior to detection. Additionally, injured fish that may later die would not be expected to be detected as they also may not immediately float to the surface of the water to be detected by visual observation.

Table 154. Estimated maximum annual number of ESA-listed salmonids that would die or be injured by explosive activities in Puget Sound. -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Location	Life Stage	Natural		Hatchery: adipose clip		Hatchery: adipose intact	
			Mortality	Injury	Mortality	Injury	Mortality	Injury
Chinook salmon - T	Crescent Harbor	Juveniles	101	22,686	-	-	92	20,652
		Adults*	1	-	-	-	-	-
	Hood Canal	Juveniles	33	2,059	109	6,748	-	-
		Adults	0	26	1	84	-	-
Steelhead - T	Crescent Harbor	Juveniles	1	-	-	-	-	-
		Adults	1	-	-	-	-	-
	Hood Canal	Juveniles	72	4,442	-	-	-	-
		Adults	1	39	-	-	-	-
Chum salmon - T	Hood Canal	Juveniles	224	16,791	16	1,171	-	-
		Adults	22	1,377	2	96	-	-

*Estimated mortality or injury of one adult Chinook salmon at Crescent Harbor may be of natural or hatchery origin.

Eulachon – Inland Habitat Use

Eulachon occur within Puget Sound, but are at very low abundance relative to coastal waters, and typically occupy very deep waters (Donnelly and Burr 1995) (74 FR 10857). Because of their relative low abundance and occurrence in very deep waters where EOD activities will not occur, we do not expect underwater explosions to adversely affect the Southern DPS of eulachon.

Green Sturgeon – Southern DPS

Southern DPS green sturgeon could be exposed to sound and energy from explosives associated with training and testing activities conducted within the Navy's two EOD ranges in Puget Sound (i.e., Hood Canal and Crescent Harbor). As discussed above in the green sturgeon status of the species section (Section 6.2.25), recent tag detection results indicate that relatively large numbers of Southern DPS green sturgeon are migrating into the Strait of Juan de Fuca. However, only a few have been detected within Puget Sound (Moore 2020), suggesting that green sturgeon use the Puget Sound estuary at a lower rate compared to the Strait of Juan de Fuca or some other estuaries along the Pacific coast (Lindley et al. 2011). There have been no reported detections of tagged green sturgeon at the Hood Canal receiver array, although these receivers were only monitored for green sturgeon for 13 months over two calendar years (2017-2018) (Moore 2020). While there are no tag detection receivers near the Navy's Crescent Harbor EOD range site, based on the lack of detections at other arrays within Puget Sound (i.e., Central Puget Sound and Tacoma Narrows), we do anticipate this is a frequently used area by Southern DPS green sturgeon. Considering the anticipated rareness of Southern DPS green sturgeon in either Hood Canal or Crescent Harbor, the small number of explosive events (i.e., six per EOD range site) proposed annually in inland waters, and the short duration of each event (i.e., up to four hours), we find it extremely unlikely (i.e., discountable) that Southern DPS green sturgeon would be exposed to the stressors associated with Navy's explosives in inland waters.

Rockfish

Bocaccio and yelloweye rockfish may be exposed to sound and energy from explosives associated with training and testing activities conducted within the Navy's detonation sites in Puget Sound (i.e., Hood Canal and Crescent Harbor). As described below, our exposure analysis for rockfish is a qualitative assessment of the likelihood of exposure to explosives based on available information including species' life histories and distribution, density information (where available), the proposed Navy activities that involve explosives (i.e., location, frequency, NEW), and the Navy's predicted range to effects as shown above (Table 147 and Table 148).

Both bocaccio and yelloweye rockfish have pelagic larval and juvenile stages, followed by a sub-adult/adult stage occurring in deeper waters. Larvae can make small movements on their own, but predominantly their movement is determined by ocean currents (Drake et al. 2010). Larvae

and pelagic juveniles are expected to occur throughout the water column in Puget Sound (Love et al. 2002b; Weis 2004). Once juvenile bocaccio reach 1 to 3.5 inches, they move into shallow nearshore waters, with rocky or cobble substrates, preferably with kelp (Love et al. 1991; 2002b). Alternatively, juvenile yellow rockfish do not occupy intertidal habitat, but are observed in deeper, offshore waters greater than 30 meters (Love et al. 1991; Studebaker et al. 2009; Yamanaka et al. 2006). Adults of both rockfish species are most commonly found between 40 and 250 meters (Love et al. 2002b; Orr et al. 2000).

Based on rockfish life histories, as described above, and the locations and water depths of the Navy's detonation sites in Puget Sound (i.e., Hood Canal and Crescent Harbor), pelagic juvenile bocaccio, larval bocaccio, and larval yelloweye rockfish could be exposed to the effects of NWTT underwater explosions. Adult bocaccio and yelloweye rockfish have been documented in Hood Canal and Whidbey Basin (82 FR 7711). Palsson et al. (2009) noted that of all areas within Puget Sound, Hood Canal had the greatest frequency of adult yelloweye rockfish observed in both trawl and scuba surveys. Adult bocaccio and adult and juvenile yelloweye rockfish, which prefer deep water areas, would not likely be affected by Navy underwater explosions which are typically conducted in shallower portions of Hood Canal and Crescent Harbor where these life stages would not be expected. Thus, take from inland explosives on adult bocaccio and adult and juvenile yelloweye rockfish as a result of the proposed action is not likely to occur.

We anticipate the onset of mortality to exposed fish to occur at 229 dB peak re 1 μPa. The onset of the lowest level of injury along the injury continuum would be either greater than 207 dB peak re 1 μPa, or greater than 186 dB SEL_{cum} dB re 1 μPa^2-s for TTS as indicated in Table 9 above. The effects of explosions on fish eggs and larvae are not well studied (Popper et al. 2014a). Although there is limited available information on the effects of explosives on early life stages of fish, we assume that rockfish larvae and juveniles within these ranges could be susceptible to the effects (including mortality, injury, and TTS) from 2.5-lb. NEW explosives (bin E3) and charges of less than 1-oz. (bin E0) proposed for use in Hood Canal and Crescent Harbor.

We cannot estimate the total number of individual fish (i.e., pelagic juvenile bocaccio, larval bocaccio, and larval yelloweye rockfish) that may be affected by activities using explosives due to the lack of density and abundance information on these particular species and life stages in the action area. Juvenile bocaccio have only been rarely documented in Puget Sound. This may be due to a relative lack of studies in Puget Sound that assessed nearshore rockfish assemblages prior to the onset of fisheries removals of adult rockfish (NMFS 2014a). Greene and Godersky (2012) sampled larval fish at representative sites across the major biogeographic basins in the Puget Sound from April-October 2011. Larvae of any rockfish species were found in Hood Canal samples only in April and May, and in Whidbey Basin (the region where Crescent Harbor is located) samples only in April. No rockfish larvae were observed at either site from June through October. The authors conducted a meta-analysis to examine patterns of larval timing expected from spawn timings of various rockfish species. Bocaccio spawning in Puget Sound

was reported only during March, while yelloweye rockfish spawning was reported from March through September (Greene and Godersky 2012).

The highest density of rockfish larvae (all species) sampled in Hood Canal was 26.5 rockfish larvae/1000 m^3, observed near the Skokomish River in April (Greene and Godersky 2012). The highest density of rockfish larvae (all species) sampled in Whidbey Basin was 16.3 rockfish larvae/1000 m^3, observed at Hoypus Point in April. Hood Canal and Whidbey Basin ranked 5th and 6th, respectively, in terms of rockfish larval density among the six major geographic basins (other basins sampled were Central Basin, Admiralty Inlet, Rosario Basin, and South Sound). This study did not provide density estimates by species and, as noted by the authors, some rockfish species are difficult to distinguish from one another during the larval phase. One listed species that can be readily identified visually at early larval stages are bocaccio, due to the pronounced size and distinct pigmentation of larval pectoral fins (Matarese et al. 2011). Greene and Godersky (2012) reported that of the 495 total rockfish larvae in their samples, none were identified as bocaccio. Although the study by Greene and Godersky (2012) was only conducted for a seven month period during one year with limited sampling stations, it represents the best available information on the density of rockfish larvae in Puget Sound. From this study, rockfish larvae (of any species) are only expected to occur in Hood Canal or Whidbey Basin during a few months out of the year. Rockfish larvae occurred in lower densities and fewer months within these two basins compared to the other four basins sampled in this study.

The Navy calculated ranges to effects for fish from explosives based on sound level exposure criteria discussed in Section 2.3.2 (Table 146, Table 147, and Table 148). Given the proposed net explosive weight (bin E3, 2.5 pounds) of the largest explosives proposed for these inland areas, only rockfish in the ranges predicted would be affected (i.e., Hood Canal - 162 meters for mortality, less than 1,210 meters for injury; Crescent Harbor - 129 meters for mortality, less than 1,674 meters for injury). Ranges to TTS for a 2.5-lb. detonation in inland waters are given as less than 159 meters for Hood Canal and less than 150 meters for Crescent Harbor. Only three E3 charges are proposed annually at each inland EOD range site, and inland explosions would involve a single charge per event. The range to TTS for a single charge are close to the range for mortality and well within the range for injury. Therefore, rockfish within range to experience TTS would likely also experience more serious effects including injuries with fitness consequences and mortality.

ESA-listed rockfish could also be exposed to sound levels from explosives resulting in physiological stress and behavioral responses. Range to effects for these responses are unknown for fishes but are anticipated to extend beyond the ranges to mortality or injury. ESA-listed rockfish (pelagic juvenile bocaccio, larval bocaccio, and larval yelloweye rockfish) could also be exposed to LIMPET charges (formally referred to as SWAG charges) in bin E0 (< 0.1 lbs.). Estimated range to onset mortality from an E0 charge is 49 meters for Hood Canal and 35 meters for Crescent Harbor. Estimated range to onset injury from an E0 charge is 427 meters for Hood Canal and 661 meters for Crescent Harbor.

ESA-listed species of rockfish are less likely to overlap with Navy explosives in time and space given their generally low abundance in Puget Sound (particularly bocaccio). Considering the small areas within Hood Canal and Crescent Harbor that would be affected by Navy explosives relative to the large area within Puget Sound representing the range of these species, and the small number of explosive events proposed annually, only a very small proportion of ESA-listed rockfish populations would likely be exposed. In the analysis conducted for NWTT Phase II, NMFS (2015c) estimated that the proportion of Puget Sound larval rockfish (bocaccio and yelloweye) and juvenile rockfish (bocaccio) that would be injured and killed as a result of Navy explosives was 0.003 and 0.002, respectively.

In summary, a relatively small number of juvenile bocaccio and larval bocaccio and yelloweye rockfish would likely be exposed to NWTT explosives in inland waters at levels resulting in a range of adverse effects including mortality, serious injuries leading to mortality or fitness consequences, and recoverable injuries. Although we could not estimate the number of individual fish affected, based on the available information and our qualitative effects analysis, we anticipate the overall impact on the Puget Sound/Georgia Basin DPSs of bocaccio and yelloweye rockfish to be extremely minor.

8.2.3.1.3 Exposure Over Seven Years of Navy Activity - Exposure Analysis

The maximum number of explosive activities that will occur over the course of the Navy's proposed seven years of activity are shown in Table 155 below. Expressing take over a seven year period allows us to consider the effects of low levels of take that may not be expected to occur in a single year, but would be likely over a longer period of time.

Table 155. Maximum number of explosive activities during the Navy's proposed NWTT Training and Testing Activities from November 2020 to November 2027.

Net Explosive Weight (lb)	Location in the NWTT Offshore[1]	In-water or at surface?	Activity	Training Max no. explosives during 7 years of activity	Testing Max no. explosives during 7 years of activity
Offshore Waters					
0.1–0.25	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	840	--
0.1–0.25	> 50 NM	Water depth of >100 ft	Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft	--	56
> 0.25–0.5	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	910	--

> 0.5-2.5	> 50 NM	Water depth of >100 ft	Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (SUS))	--	504
>2.5–5	< 1,000 feet depth	In-water	Mine Countermeasure and Neutralization	--	108
> 5–10	> 50 NM	Surface	Gunnery Exercise (Surface-to-Surface) – Ship	784	--
> 20– 60	< 1,000 feet depth	In-water	Mine Countermeasure and Neutralization	--	15
> 60– 100	> 50 NM	Surface	Torpedo (Explosive) Testing (MK-46/54)	--	28
> 250– 500	> 50 NM	Surface	Missile Exercise Air-to-Surface	14	--
		Surface	Bombing Exercise Air-to-Surface	14	--
> 500–650	> 50 NM	Surface	Torpedo (Explosive) Testing (MK-48)	--	28
Inland Waters					
< 0.1	Crescent Harbor	In-water	Mine Neutralization – Explosive Ordnance Disposal (EOD)	126	--
	Hood Canal			126	
> 0.5–2.5	Crescent Harbor	In-water	Mine Neutralization – Explosive Ordnance Disposal (EOD)	21	--
	Hood Canal			21	

We assume for the purposes of this analysis that the current annual level of activities and impacts will continue into the reasonably foreseeable future, and we have made our jeopardy determination on that basis. If the status of the species and Navy activity levels remain the same, we would expect the estimated number of fish killed or injured by Navy explosive activities in a seven year period to continue into the reasonably foreseeable future. However, we believe that because our estimates of the number of ESA-listed fish that will be injured or killed is dependent on fish abundance, attempting to predict the number of fish affected by Navy activities beyond seven years into the future is too speculative to allow for a scientifically meaningful assessment of the impacts on ESA-listed species beyond this timeframe. This is because fish abundance may

change significantly over even a limited period of time due to a variety of factors including changes in fisheries harvest strategies, habitat conditions, and other changes in the environmental baseline that may impact reproductive success (See Section 6.2). That being said, there is no evidence that expressing take over a seven year period risks overlooking effects that might appear only over a longer period of time because our analysis was structured such that the magnitude of the percentage of the population affected would not be expected to change over time unless Navy activity levels changed. Additionally, because MMPA take authorization is limited to seven years, we will need to reassess the Navy's proposed action under both the MMPA and ESA prior to the expiration of the proposed take authorization, and not less than every seven years thereafter, assuming the Navy makes similar requests for MMPA incidental take authorizations for continued testing and training activities. These periodic reassessments will allow us to update the environmental baseline, assess any changes in ESA-listed fish species status, and to adjust our estimates of injury and mortality accordingly. In this manner, we will be able to ensure that our assessment of the effect of the Navy's ongoing activities and the resultant impacts on all listed species, including fish species, are up to date, and that no impacts are omitted from consideration.

Results for the seven year exposure estimates of Navy explosive activities from the offshore portion of the action area are shown in Tables 156-159. Results from inland waters are presented in Table 160. Further discussion on the impacts of these seven-year exposure estimates on a population-wide level are discussed in detail in Section 10 *Integration and Synthesis*, while the section below (Section 8.2.3.2) details responses to the effects from explosive activities at the individual level.

Table 156. Estimated number of ESA-listed salmonids (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities.

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
Chinook	Adult	Sac River winter run - E	1	120	-	-	1	120
	Juvenile		29	3,597	-	-	29	3,597
	Adult	Central valley spring run - T	1	123	-	-	1	123
	Juvenile		318	39,014	-	-	318	39,014
	Adult	California coastal - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Adult	Snake River fall - T	15	2,226	8	570	23	2,796
	Juvenile		302	36,956	-	-	302	36,956
	Adult	Snake River spring/summer - T	3	392	1	101	4	493
	Juvenile		541	66,267	-	-	541	66,267
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		4,324	529,836	-	-	4,324	529,836
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		631	77,277	-	-	631	77,277
	Adult	Upper Columbia River spring - E	7	1,024	4	262	11	1,286
	Juvenile		75	9,251	-	-	75	9,251
	Adult	Puget Sound - T	-	-	-	-	-	-
	Juvenile		3,309	405,478	-	-	3,309	405,478
Coho	Adult	Central Calif coast - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	S. Oregon/N. Calif coast - T	-	-	-	-	-	-
	Juvenile		43	5,317	-	-	43	5,317
	Adult	Oregon coast - T	-	30	-	-	-	30
	Juvenile		13	1,595	-	-	13	1,595
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		1,897	232,524	-	-	1,897	232,524
Chum	Adult	Columbia River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
Sockeye	Adult	Snake River - E	-	4	-	-	-	4
	Juvenile		1	88	-	10	1	98
Steelhead	Adult	South-Central California - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Adult	Central Calif - T	-	64	-	17	-	81
	Juvenile		23	3,592	15	1,086	38	4,678
	Adult	California Central Valley - T	-	64	-	18	-	82
	Juvenile		57	8,860	34	2,228	91	11,088
	Adult	Northern Calif - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Upper Columbia River - E	1	88	1	48	2	136
	Juvenile		25	3,806	18	1,352	43	5,158
	Adult	Snake River basin - T	9	1,320	5	331	14	1,651
	Juvenile		118	18,267	69	4,575	187	22,842
	Adult	Lower Columbia River - T	-	-	-	-	-	-
	Juvenile		43	6,626	25	1,622	68	8,248
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Middle Columbia River - T	-	7	-	2	-	9
	Juvenile		16	2,463	10	746	26	3,209
	Adult	Puget Sound - T	-	-	-	-	-	-
	Juvenile		4	609	2	148	6	757

Table 157. Estimated number of ESA-listed salmonids (hatchery produced with intact adipose) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities.

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
Chinook	Adult	Sac River winter run - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Central valley spring run - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	California coastal - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Snake River fall - T	16	2,411	9	618	25	3,029
	Juvenile		348	42,591	-	-	348	42,591
	Adult	Snake River spring/summer - T	0	69	0	18	1	87
	Juvenile		94	11,536	-	-	94	11,536
	Adult	Lower Columbia River - T	7	818	-	-	7	818
	Juvenile		133	16,264	-	-	133	16,264
	Adult	Upper Willamette River - T	34	5,228	20	1,339	54	6,568
	Juvenile		0	3	-	-	0	3
	Adult	Upper Columbia River spring - E	6	903	4	241	9	1,144
	Juvenile		45	5,485	-	-	45	5,485
	Adult	Puget Sound - T	2	204	-	-	2	204
	Juvenile		663	81,226	-	-	663	81,226
Coho	Adult	Central Calif coast - E	-	34	-	-	-	34
	Juvenile		70	8,574	-	-	70	8,574
	Adult	S. Oregon/N. Calif coast - T	5	581	-	-	5	581
	Juvenile		125	15,286	-	-	125	15,286

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Adult	Oregon coast - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Lower Columbia River - T	5	561	-	-	5	561
	Juvenile		65	7,970	-	-	65	7,970
Chum	Adult	Columbia River - T	0	1	-	-	0	1
	Juvenile		3	326	-	-	3	326
Sockeye	Adult	Snake River - E	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
Steelhead	Adult	South-Central California - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Central Calif - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	California Central Valley - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Northern Calif - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Upper Columbia River - E	-	19	-	11	-	30
	Juvenile		5	767	4	272	9	1,039
	Adult	Snake River basin - T	2	268	1	67	3	335
	Juvenile		25	3,905	15	978	40	4,883
	Adult	Lower Columbia River - T	2	370	1	91	3	461
	Juvenile		-	51	-	12	-	63
	Adult	Upper Willamette River - T	-	-	-	-	-	-
	Juvenile		-	-	-	-	-	-
	Adult	Middle Columbia River - T	-	2	-	1	-	3
	Juvenile		4	611	3	185	7	796

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Adult	Puget Sound - T	-	-	-	-	-	-
	Juvenile		4	623	2	151	6	774

Table 158. Estimated number of ESA-listed salmonids (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities.

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
Chinook	Adult	Sac River winter run - E	0	11	-	-	0	11
	Juvenile		29	3,513	-	-	29	3,513
	Adult	Central valley spring run - T	2	201	-	-	2	201
	Juvenile		114	13,946	-	-	114	13,946
	Adult	California coastal - T	3	380	-	-	3	380
	Juvenile		188	22,985	-	-	188	22,985
	Adult	Snake River fall - T	12	1,840	7	471	19	2,311
	Juvenile		84	10,309	-	-	84	10,309
	Adult	Snake River spring/summer - T	14	2,104	8	539	22	2,643
	Juvenile		122	14,991	-	-	122	14,991
	Adult	Lower Columbia River - T	5	625	-	-	5	625
	Juvenile		1,620	198,477	-	-	1,620	198,477
	Adult	Upper Willamette River - T	11	1,695	6	434	17	2,129
	Juvenile		162	19,887	-	-	162	19,887
	Adult	Upper Columbia River spring - E	3	472	2	121	5	593
	Juvenile		57	6,976	-	-	57	6,976
	Adult	Puget Sound - T	2	294	-	-	2	294

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Juvenile		277	33,907	-	-	277	33,907
Coho	Adult	Central Calif coast - E	2	200	-	-	2	200
	Juvenile		67	8,174	-	-	67	8,174
	Adult	S. Oregon/N. Calif coast - T	4	482	-	-	4	482
	Juvenile		437	53,531	-	-	437	53,531
	Adult	Oregon coast - T	41	5,015	-	-	41	5,015
	Juvenile		1,441	176,566	-	-	1,441	176,566
	Adult	Lower Columbia River - T	16	1,906	-	-	16	1,906
	Juvenile		172	21,105	-	-	172	21,105
Chum	Adult	Columbia River - T	-	14	-	-	0	14
	Juvenile		24	2,932	-	-	24	2,932
Sockeye	Adult	Snake River - E	-	1	-	-	-	1
	Juvenile		-	7	-	1	-	8
Steelhead	Adult	South-Central California - T	-	12	-	5	-	17
	Juvenile		3	438	3	226	6	664
	Adult	Central Calif - T	-	36	-	10	-	46
	Juvenile		9	1,377	6	416	15	1,793
	Adult	California Central Valley - T	-	28	-	8	-	36
	Juvenile		23	3,489	13	878	36	4,367
	Adult	Northern Calif - T	1	120	-	32	1	152
	Juvenile		29	4,547	18	1,240	47	5,787
	Adult	Upper Columbia River - E	-	32	-	18	-	50
	Juvenile		7	1,104	5	392	12	1,496
	Adult	Snake River basin - T	1	175	1	44	2	219
	Juvenile		29	4,419	17	1,107	46	5,526
	Adult	Lower Columbia River - T	1	215	1	52	2	267
	Juvenile		13	1,949	7	477	20	2,426

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
	Adult	Upper Willamette River - T	-	48	-	14	-	62
	Juvenile		5	777	4	317	9	1,094
	Adult	Middle Columbia River - T	1	84	-	27	1	111
	Juvenile		15	2,257	10	684	25	2,941
	Adult	Puget Sound - T	2	321	1	78	3	399
	Juvenile		79	12,160	45	2,951	124	15,111

Table 159. Estimated number of ESA-listed Southern DPS Eulachon that would be killed or injured in the offshore portion of the NWTT action area over a seven year period. - Indicates no estimates of ESA-listed Southern DPS Eulachon affected by Navy testing activities.

Species	Life stage	ESU/DPS	Testing		Training		Total	
			Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
Eulachon	Adult	Southern DPS	1,023	37,104	-	-	1,023	7,559

Table 160. Estimated number of ESA-listed salmonids that would be killed or injured in the inland portion of the NWTT action area over a seven year period. - Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury from training or testing activities.

Species	Location	Life Stage	Natural		Hatchery: adipose intact		Hatchery: adipose clip		Total	
			Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)	Mortality (7 years)	Injury (7 years)
Chinook salmon - T	Crescent Harbor	Juveniles	286	88,584	-	-	260	80,639	286	88,584
		Adults	1	-	-	-	-	-	1	-
	Hood Canal	Juveniles	211	13,235	692	43,380	-	-	211	13,235
		Adults	3	165	9	542	-	-	3	165
Steelhead - T	Crescent Harbor	Juveniles	1	-	-	-	-	-	1	-
		Adults	1	-	-	-	-	-	1	-
	Hood Canal	Juveniles	456	28,555	-	-	-	-	456	28,555
		Adults	4	251	-	-	-	-	4	251
Chum - T	Hood Canal	Juveniles	1,568	117,537	109	8,198	-	-	1,568	117,537
		Adults	141	8,852	10	618	-	-	141	8,852

8.2.3.2 Response Analysis

8.2.3.2.1 Injury and Mortality

As described previously, NMFS considers the potential effects from explosive exposure to pose the highest risk of injury and mortality compared to all other sound sources the Navy proposes to use. Based upon the range to effect calculations for onset of injury to fishes from the sound produced from explosions, fish located within hundreds (most of the charges) to a few thousand meters (largest charges) could be injured or killed. In general, the explosives that belong to larger bins (with large NEWs) produce longer ranges within each effect category. For the largest charges, there are usually only one or very few of this type of explosives proposed for use during the seven-year duration of the activities. Some ranges will also vary depending upon the number of explosions in a single activity, depth and weight of the charge.

It is important to note injury was defined as the distance from the detonation where no injury would occur; we consider this the point of onset injury. Injury refers to the direct effects on the tissues or organs of a fish. We expect the majority of fish injuries to be minor and recoverable although some injuries may lead to internal bleeding, barotrauma, and death. The blast wave from an in-water explosion is lethal to fishes at close range, causing massive organ and tissue damage (Keevin and Hempen 1997). At greater distance from the detonation point, the extent of mortality or injury depends on a number of factors including fish size, body shape, depth, physical condition of the fish, and perhaps most importantly, the presence of a swim bladder (Keevin and Hempen 1997; Wright 1982; Yelverton and Richmond 1981; Yelverton et al. 1975). At the same distance from the source, larger fishes are generally less susceptible to death or injury, elongated forms that are round in cross-section are less at risk than deep-bodied forms, and fishes oriented sideways to the blast suffer the greatest impact (Edds-Walton and Finneran 2006; O'Keeffe 1984; O'Keeffe and Young 1984; Wiley et al. 1981; Yelverton et al. 1975). Species with a swim bladder are much more susceptible to blast injury from explosives than fishes without them (Gaspin 1975b; Gaspin et al. 1976; Goertner et al. 1994).

Fish that are near the range to mortality may be more likely to incur a more severe injury that could lead to mortality with time (e.g., internal bleeding, barotrauma, higher susceptibility to predation). For example, if a fish is close to an explosive detonation, the exposure to rapidly changing high pressure levels can cause barotrauma. Barotrauma is injury due to a sudden difference in pressure between an air space inside the body and the surrounding water and tissues. Rapid compression followed by rapid expansion of airspaces, such as the swim bladder, can damage surrounding tissues and result in the rupture of the airspace itself. The swim bladder is the primary site of damage from explosives (Wright 1982; Yelverton et al. 1975). Gas-filled swim bladders resonate at different frequencies than surrounding tissue and can be torn by rapid oscillation between high- and low-pressure waves (Goertner 1978). Swim bladders are a characteristic of most bony fishes with the notable exception of flatfishes (e.g., halibut). Sharks and rays are examples of fishes without a swim bladder. Small airspaces, such as micro-bubbles

that may be present in gill structures, could also be susceptible to oscillation when exposed to the rapid pressure increases caused by an explosion. This may have caused the bleeding observed on gill structures of some fish exposed to explosions (Goertner et al. 1994). Sudden very high pressures can also cause damage at tissue interfaces due to the way pressure waves travel differently through tissues with different densities. Rapidly oscillating pressure waves might rupture the kidney, liver, spleen, and sinus and cause venous hemorrhaging (Keevin and Hempen 1997).

Several studies have exposed fish to explosives and examined various metrics in relation to injury susceptibility. Sverdrup (1994) exposed Atlantic salmon (1 to 1.5 kilograms [2 to 3 pounds]) in a laboratory setting to repeated shock pressures of around 2 MPa (300 psi) without any immediate or delayed mortality after a week. Hubbs and Rechnitzer (1952a) showed that fish with swim bladders exposed to explosive shock fronts (the near-instantaneous rise to peak pressure) were more susceptible to injury when several feet below the water surface than near the bottom. When near the surface, the fish began to exhibit injuries around peak pressure exposures of 40 to 70 psi. However, near the bottom (all water depths were less than 100 feet [ft.]) fish exposed to pressures over twice as high exhibited no sign of injury. Yelverton et al. (1975) similarly found that peak pressure was not correlated to injury susceptibility; instead, injury susceptibility of swim bladder fish at shallow depths (10 feet or less) was correlated to the metric of positive impulse (Pa-s), which takes into account both the positive peak pressure, the duration of the positive pressure exposure, and the fish mass, with smaller fish being more susceptible.

Dahl et al. (2020) reported the effects of underwater explosions on one species of Clupeiform fish, Pacific sardines (*Sardinops sagax*), with a physostomous swim bladder (an open swim bladder with direct connection to the gut via pneumatic duct). Fish were stationed at various distances prior to each explosion, in addition to a control group that was not exposed. Necropsies following explosions observed significant injuries, including fat hematoma, kidney rupture, swim bladder rupture, and reproductive blood vessel rupture. While most significant injuries were consistently present at close range (<50 meters), there were inconsistent findings at the 50 to 154 meter range. For example, higher rates of injury were shown at a distance of 154 meters compared to 68 meters. The acoustic analysis led to the suggestion that, at a range of 154 meters, the greater rate of pressure change and deeper low-pressure point during the decompression phase were linked to swim bladder expansion damage and rupture. This includes damage to immediately neighboring organs as manifested by kidney data that are highly correlated with swim bladder data. This is the proposed cause of the observed higher injury rate at 154 meters despite the higher peak pressures observed at 68 meters, and has been suggested by earlier models of swim bladder related injury (Goertner, 1978 and Wiley et al., 1981 as cited in Dahl et al. 2020).

Gaspin et al. (1976) exposed multiple species of fish with a swim bladder, placed at varying depths, to explosive blasts of varying size and depth. Goertner (1978) and Wiley (1981)

developed a swim bladder oscillation model, which showed that the severity of injury observed in those tests could be correlated to the extent of swim bladder expansion and contraction predicted to have been induced by exposure to the explosive blasts. Per this model, the degree of swim bladder oscillation is affected by ambient pressure (i.e., depth of fish), peak pressure of the explosive, duration of the pressure exposure, and exposure to surface rarefaction (negative pressure) waves. The maximum potential for injury is predicted to occur where the surface reflected rarefaction (negative) pressure wave arrives coincident with the moment of maximum compression of the swim bladder caused by exposure to the direct positive blast pressure wave, resulting in a subsequent maximum expansion of the swim bladder. Goertner (1978) and Wiley et al. (1981) found that their swim bladder oscillation model explained the injury data in the Yelverton et al. (1975) exposure study and their impulse parameter was applicable only to fishes at shallow enough depths to experience less than one swim bladder oscillation before being exposed to the following surface rarefaction wave.

O'Keeffe (1984) provides calculations and contour plots that allow estimation of the range to potential effects of in-water explosions on fish possessing swim bladders using the damage prediction model developed by Goertner (1978). O'Keeffe's (1984) parameters include the charge weight, depth of burst, and the size and depth of the fish, but the estimated ranges do not take into account unique propagation environments that could reduce or increase the range to effect. The 10 percent mortality range represents the maximum horizontal range predicted by O'Keeffe (1984) for 10 percent of fish suffering injuries that are expected to not be survivable (e.g., damaged swim bladder or severe hemorrhaging). Fish at greater depths and near the surface are predicted to be less likely to be injured because geometries of the exposures would limit the amplitude of swim bladder oscillations. In contrast, detonations at or near the surface (i.e., similar to most Navy activities that utilize bombs and missiles) would result in energy loss at the water air interface resulting in lower overall ranges to effect than those predicted here.

In contrast to fish with swim bladders, fishes without swim bladders have been shown to be more resilient to explosives (Gaspin 1975b; Gaspin et al. 1976; Goertner et al. 1994). For example, some small (average 116 mm length; approximately 1 oz.) hogchokers (*Trinectes maculatus*) exposed less than 5 feet from a 10 pounds pentolite charge immediately survived the exposure with slight to moderate injuries, and only a small number of fish were immediately killed; however, most of the fish at this close range did suffer moderate to severe injuries, typically of the gills or around the otolithic structures (Goertner et al. 1994).

Studies that have documented caged fishes killed during planned underwater explosions indicate that most fish that die do so within one to four hours, and almost all die within a day (Yelverton et al. 1975). Mortality in free-swimming (uncaged) fishes may be higher due to increased susceptibility to predation. Fitch and Young (1948) found that the type of free-swimming fish killed changed when blasting was repeated at the same location within 24 hours of previous blasting. They observed that most fish killed on the second day were scavengers, presumably attracted by the victims of the previous day's blasts.

8.2.3.2.2 Hearing Impairment (TTS)

ESA-listed fishes may experience TTS as a result of explosives in the action area. We are unaware of any research demonstrating TTS in these species (or other fish species with a swim bladder not involved in hearing) from explosives. Although TTS has not been demonstrated in these species' groups, this does not mean it does not occur. Because we know it can occur from other acoustic stressors, we assume it is possible from exposure to an explosive sound stressor. The criteria used for TTS was based upon a conservative value for more sensitive fish species and life stages with swim bladders. If TTS does occur, it would likely co-occur with barotraumas (i.e., non-auditory injury), and therefore would be within the range of other injuries these fishes are likely to experience from blast exposures. None of the ESA-listed fish considered in this opinion (i.e., salmonids, eulachon, sturgeon or rockfish) have a hearing specialization or a swim bladder involved in hearing, thus, minimizing the likelihood of each instance of TTS affecting an individual's fitness. Most fish species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in salmonid migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Depending on the severity of the TTS and underlying degree of hair cell damage, a fish would be expected to recover from the impairment over a period of weeks (for the worst degree of TTS).

In summary, because the ESA-listed fish species considered in this opinion are not known to rely on hearing for essential life functions, and any effects from TTS would be short-term and temporary, instances of TTS would not likely result in measurable effects on any individual's fitness.

8.2.3.2.3 Physiological Stress and Behavioral Responses

Physiological and behavioral responses of fishes to acoustic stressors have been described in greater detail for other acoustics stressors on fishes. Exposure to explosions could cause spikes in stress hormone levels, or alter a fish's natural behavioral patterns. There are currently no behavioral thresholds for explosives established for fishes. Behavioral responses could be expected to occur within the range to effects for other injurious or physiological responses, and perhaps be extended beyond these ranges if a fish could detect the sound at those greater distances. Given that none of the species considered here have any specialized hearing adaptations, and the threshold for TTS is considered conservative for these hearing groups, most behavioral responses would be expected to occur within the range to effects for injury, mortality and TTS. However, because sound generated from a detonation is brief, long-term effects on fish behavior are unlikely. Similarly, long periods of masking are unlikely from blast exposure for fishes, although some brief masking periods could also occur if multiple detonations occurred (within a few seconds apart). If multiple exposures occurred within a short period of times, such as over the course of a day or consecutive days, fishes may also choose to avoid the area of disturbance. The Navy's training and testing activities involving explosions are generally dispersed in space and time throughout the large action area, and repeated exposure of individual

fishes to sound and energy from underwater explosions over the course of a day or multiple days is not likely. Thus, most physiological stress and behavioral effects are expected to be temporary, of a short duration, and would return to normal quickly after cessation of the blast wave.

In summary, for all ESA-listed fish species, behavioral effects resulting from reactions to sound created by Navy explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion.

8.3 Designated and Proposed Critical Habitat Effects Analysis

In this section, we analyze the effects of the proposed action on designated (or proposed) critical habitat that may be affected by the proposed action (see Section 6.2 for description and status of these critical habitats).

8.3.1 Marine Mammals

8.3.1.1 Humpback Whale Central America DPS and Mexico DPS Proposed Critical Habitat

Proposed critical habitat for the humpback whale Central America and Mexico DPSs is described in Section 6.2.3. Here we evaluate the effects of the proposed action on the one identified physical and biological feature (PBF) of this critical habitat: euphausiids and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth (84 FR 54354). These schooling prey fishes include species such as capelin, herring, and mackerel.

The offshore portion of the action area overlaps with most of the proposed critical habitat for the humpback whale Central America DPS and a portion of the proposed critical habitat for the humpback whale Mexico DPS (Figure 101). Training activities that would be carried out in this portion of the action area include air warfare, anti-submarine warfare, electronic warfare, surface warfare, and other activities (see Table 21 above). Testing activities that would be carried out in this portion of the action area include anti-submarine warfare, mine warfare, surface warfare, unmanned systems, vessel evaluation, electronic warfare, and other activities (see Table 22 and Table 23 above).

The portion of the action area around Western Behm Canal overlaps the Alaska portion of the proposed critical habitat for the humpback whale Mexico DPS (Figure 102). Testing activities that would be carried out in this portion of the action area would include anti-submarine warfare, vessel evaluation, and other activities (see Table 24 and Table 25 above). There are no NWTT activities involving active sonar or explosives proposed in the portion of the action area around Western Behm Canal.

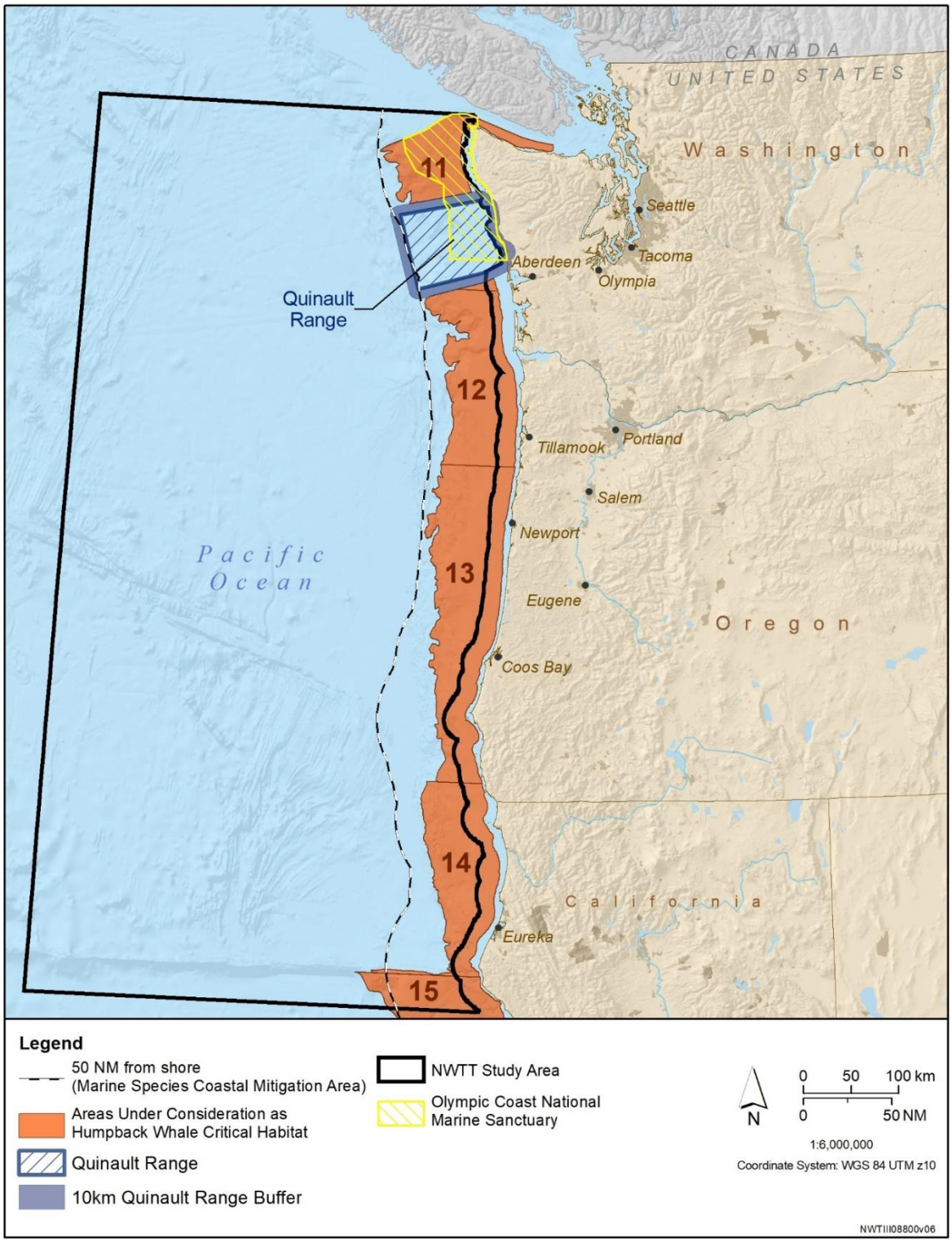


Figure 101. Overlap of proposed humpback whale critical habitat (Units 11 through 15) with the NWTT offshore action area.

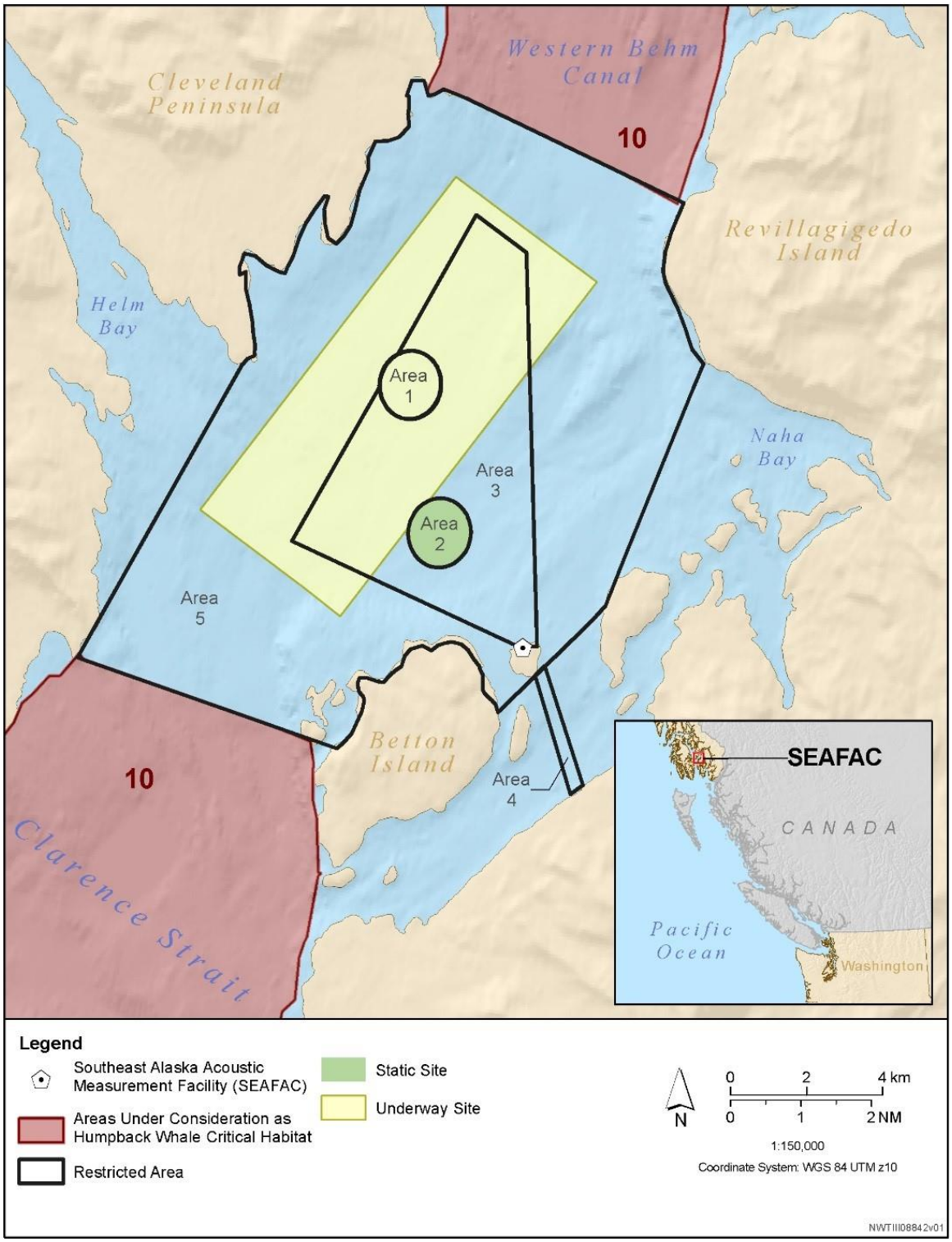


Figure 102. Overlap of proposed humpback whale critical habitat (Unit 10) and Western Behm Canal portion of the NWTT action area.

As discussed in Section 8.1.3, energy, physical disturbance and strike, entanglement, ingestion, and non-impulsive acoustic stressors are not likely to adversely affect ESA-listed fishes. For these reasons, it is extremely unlikely that humpback whale prey items (euphausiids and schooling fishes) would be adversely affected by these stressors. In addition, the abundance of prey items in the proposed critical habitat for the Central America and Mexico DPSs is likely several orders of magnitude higher than those of ESA-listed fishes in the region.

Regarding impulsive acoustic stressors from explosives in the offshore portion of the action area, humpback whale prey items may be adversely affected if they happen to be in the vicinity of detonations, as discussed in Section 8.2.3 for ESA-listed fishes. Adverse effects may include injury, TTS, physiological stress, behavioral reactions, and mortality. Those fish that are killed within the proposed critical habitat would no longer be available to humpback whales as prey items. Adverse effects other than mortality would not be anticipated to remove individuals (prey) from their respective populations, nor would any non-mortal temporary or isolated impacts to prey items be expected to reduce the quality of prey in terms of nutritional content.

Crustaceans, including euphausiids, lack a swim bladder but instead have a statocyst, a sac-like structure with sensory hairs that may be used for orientation. Anatomical damage in invertebrates from low-frequency sounds is limited but statocyst damage has been observed in cephalopods exposed to sounds from seismic surveys. No evidence of effects from mortality at the population level has been observed in shrimp following seismic airgun exposure and shrimp have not been observed responding to low-frequency sounds, but more research is needed to confirm this (reviewed in Carroll et al. 2017). Despite the lack of evidence regarding adverse effects to euphausiids from low-frequency noise, these organisms may experience adverse effects if they are close enough to a detonation.

No explosive training activities would occur within the proposed humpback critical habitat off Washington or Oregon (Units 11, 12, and 13; see Figure 101). Beyond 50 nautical miles from the northern California shoreline there is a small portion of Unit 14 and a small portion of Unit 15 (see Figure 101) that could overlap with one explosive Navy training activity, explosive Gunnery Exercise Surface-to-Surface. These exercises are proposed to occur anywhere in the NWTT action area outside 50 nautical miles from the coast.

One testing event involving the use of explosives, Torpedo (Explosive) Testing, may potentially overlap with the small portion of Unit 14 or the small portion of Unit 15 (see Figure 101) located beyond 50 nautical miles from the coast. The event duration is one to two days. One testing event involving the use of explosives, Explosive Mine Countermeasure and Neutralization Testing, may potentially overlap with Units 11, 12, or 13. Explosive Mine Countermeasure and Neutralization Testing activities are multi-day events involving explosives (Bin E4 and Bin E7) that would occur up to twice per year. Operational parameters require explosive Mine Countermeasure and Neutralization Testing to occur in water depths up to 1,000 feet and typical depths of 300 feet. The activity would occur for E4 at three nautical miles or greater, and E7 at

six nautical miles or greater from shore at the Quinault Range Site. Outside of the Quinault Range Site, the activity would occur 12 nautical miles or greater from shore off Washington and Oregon. This activity is not conducted south of the Oregon/California border due to operational requirements.

Impulsive acoustic stressors from explosives would not occur in the Western Behm Canal portion of the action area, as explosives are not used by the Navy in this area. Therefore, critical habitat for the Central America and Mexico DPSs of humpback whales around the Western Behm Canal (Unit 10 in Figure 102) would not be affected by impulsive acoustic stressors from explosives.

If prey items are killed within humpback whale critical habitat, it is likely that only a low number of individuals and therefore a small portion of prey species populations may be killed. Although some prey items could be killed within the described mortality ranges during an explosive activity, other prey items would likely be available to humpback whales in the immediate area surrounding the activity or would return to the area after the activity is complete. Exposure to explosions would be highly dependent on the limited number of explosive activities that overlap proposed critical habitat and the actual presence of prey species at the time explosive activities occur. The portion of Navy activities using explosives within the proposed critical habitat would be small relative to the portion of activities that could occur in the overall Offshore Area, further reducing the potential for effects on prey items. This would result in a minimal change in the overall quantity or availability of prey items within the habitat as a whole. Although some individual prey items may be killed, long-term consequences for fish and invertebrate populations and the effect on overall quantity, quality and availability of prey items would be insignificant.

Given the large area of the proposed critical habitat compared to the relatively small size of the offshore area where detonations would occur, the frequency of how often these events would occur, the short duration of these events, the various mitigation measures (including halting of activities until marine mammals are out of the area and are not observed feeding, see Section 3.5.1) and the relatively large number of prey items available throughout the proposed critical habitat, we conclude that any impacts of explosives resulting from NWTT activities on prey availability for the humpback whale Central America and Mexico DPSs would be insignificant at the population level. In summary, although explosives would likely result in injury and mortality to humpback whale prey species within proposed critical habitat areas, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of the Central America and Mexico DPSs. The effects of all stressors analyzed on the essential PBF were found to be either insignificant or discountable and not likely to reduce the conservation value of proposed critical habitat. Therefore, for the reasons provided above, we determine that the proposed action

may affect but is not likely to adversely affect proposed critical habitat for the Central America and Mexico DPSs of humpback whales.

8.3.1.2 Southern Resident Killer Whale DPS Critical Habitat

In this section we analyze the effects of the proposed action on the essential PBFs of inland designated critical habitat and coastal proposed critical habitat for Southern Resident killer whales. The identified physical and biological features (PBFs) that are essential to the conservation of the Southern Resident killer whales DPS are the same for both inland and coastal areas. These are: (1) water quality to support growth and development; (2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and (3) inter-area passage conditions to allow for migration, resting, and foraging.

NMFS previously considered identifying "sound levels that do not exceed thresholds that inhibit communication or foraging activities or result in temporary or permanent hearing loss" as a potential essential feature of the whales' inland critical habitat (69 FR 76673; December 22, 2004), but ultimately concluded that sufficient information was not available to do so (NMFS 2019e). An acoustic environment, or soundscape, in which Southern Resident killer whales can detect and interpret sounds is critical for carrying out basic life functions including communication, navigation, and foraging. Although sound has not been identified as a separate essential feature (of either designated inland or proposed coastal critical habitat), we can assess adverse habitat-related effects of anthropogenic sound by evaluating impacts to the prey and passage essential features of critical habitat for Southern Resident killer whales. That is, we evaluate whether acoustic stressors resulting from the proposed action might alter the conservation value of habitat by reducing the quantity, quality, or availability of the whales' prey in a particular foraging area, by reducing the effective echolocation space for the whales to forage, or by creating a barrier that restricts movements through or within an area necessary for migration, resting, or foraging.

8.3.1.2.1 Designated Critical Habitat – Inland Waters

Several of the activities associated with the proposed action that could affect critical habitat would occur entirely in the offshore portion of the action area and, therefore, would not overlap spatially with Southern Resident killer whale inland critical habitat. Many other activities that are proposed for the inland waters would occur in inland areas that are not part of the critical habitat designation (e.g., Hood Canal) or were excluded for national security reasons (e.g., Crescent Harbor Explosive Ordnance Units Training Area and Strait of Juan de Fuca and Whidbey Island naval restricted areas – see Section 6.2.4 full list of excluded areas). Examples of activities or stressors that would not overlap spatially with inland critical habitat include: weapons firing and impact noise (except small caliber gunnery using blanks); laser testing; medium and large decelerators and parachutes; chemical and biological simulants; and expended chaff and flares.

For those activities that may overlap with Southern Resident killer whale inland critical habitat, many would have no pathway through which the resulting stressors could affect any of the essential features of critical habitat (i.e., water quality, prey, or passage conditions). These stressors include electromagnetic devices, expended wires and cables, small decelerators and parachutes, and ingestion stressors including fragments from non-explosive practice munitions, explosives, and targets. As discussed in Section 8.1.1.6 (*Stressors Resulting in Effects to Marine Mammal Habitat or Prey*), any secondary effects of stressors resulting from the proposed action on Southern Resident killer whales via impacts to water quality were found to be either insignificant or discountable.

The use of biodegradable polymers is proposed for two locations in the inland portion of the action area: Keyport Range Site and Dabob Bay Range Complex. While designated Southern Resident killer whale inland critical habitat does not include either location, there is some limited overlap between the periphery of the Keyport Range Site and the designated critical habitat. Based on the limited number of annual activities involving biodegradable polymers in inland waters (i.e., up to four annual events), the anticipated concentration of biodegradable polymers at any given time is likely very low within the range sites, and presumably lower along the periphery with critical habitat. As discussed above (Sections 8.1.1.4 and 8.1.3.4), it is extremely unlikely that a killer whale or their salmonid prey would become entangled in biodegradable polymers. None of the essential features of Southern Resident killer whale inland critical habitat (i.e., water quality, prey, or inter-area passage) would likely be adversely affected by the use of biodegradable polymers in inland waters.

Most sounds sources proposed for use during training and testing activities overlapping or adjacent to critical habitat in the action area would not fall within the frequency range of salmonid hearing, thereby presenting no plausible route of effect on salmonids (Navy 2020e). The few sound sources used within salmonid hearing range would be limited and typically transient and temporary. As noted above (Section 8.1.3.1), no long-term consequences for any exposed ESA-listed salmonids are expected from sonar and other transducers. Thus, any effects on ESA-listed salmonids that may occur would be minor and temporary and are not anticipated to lead to a significant disruption of normal behavior patterns. Results from fecal DNA analysis indicate that greater than 98 percent of Southern Resident killer whale diet in inland waters is made up of salmonids, with Chinook salmon comprising 79.5 percent of the overall diet (Ford et al. 2016). While Southern Resident killer whale prey includes both ESA-listed and non ESA-listed salmonid populations, we expect the effect pathways and resulting responses to Navy sonar and other transducers would be the same for all salmonid species, DPSs or ESUs. Thus, any effects of Navy sonar and other transducers on the quantity, quality, and availability of Southern Resident killer whale prey would not likely reduce the conservation value of Southern Resident killer whale inland critical habitat.

As discussed in Section 8.1.3.3, potential effects on ESA-listed fish from vessels and in-water devices were determined to be discountable (in the case of vessel strike) or insignificant (in the

case of behavioral or stress response from vessel noise or presence). Similar to the effects from sonar, we expect the effect pathways and resulting responses to Navy vessels and in-water devices would be the same for all salmonid populations, regardless of their ESA status. The minor, infrequent contribution of naval vessel noise to background noise would not chronically reduce effective foraging echolocation space in critical habitat (Navy 2020e). Thus, any effects of Navy vessels and in-water devices on the quantity, quality, and availability of Southern Resident killer whale prey would not likely reduce the conservation value of Southern Resident killer whale inland critical habitat.

Southern Resident killer whales are heavily reliant on sound for important life functions. Acoustic stressors resulting from the proposed action that would occur in the inland portion of the action area and may overlap with designated critical habitat include vessel noise, weapons noise from small-caliber gunnery (using blanks), and sonar. We evaluate the effects of these stressors on the inter-area passage essential feature of critical habitat (i.e., inter-area passage conditions to allow for migration, resting, and foraging).

As discussed in Section 8.1.1.1, while Southern Resident killer whales could be exposed to sound from Navy vessels and aircraft, these sounds sources are expected to cause minimal disturbance. If a killer whale detects a vessel and avoids it, or has a temporary stress response from the noise disturbance, these responses are expected to be temporary and only endured while the vessel transits through the area where the animal encountered it. Activities that may produce weapons noise in the inland portion of the action area would only involve the use of small-caliber gunnery (using blanks). The inter-area passage essential feature of Southern Resident killer whale inland critical habitat would not likely be adversely affected by the insignificant noise produced by small arms fire in the air. Similarly, since the proposed use of sonar and other transducers during NWTT activities would be transient and temporary, any effects on Southern Resident killer whales would likely be temporary. The Navy's quantitative analysis estimates up to three exposures resulting in behavioral harassment per year as a result of NWTT activities in inland waters (see Section 8.2.1.1 for details). No exposures resulting in PTS or TTS of Southern Resident killer whales in inland waters were predicted by the model.

Adverse habitat-related effects from acoustic stressors are more likely to result from chronic or long-lasting noise sources that degrade the value of habitat by interfering with the sound-reliant animal's ability to gain benefits from that habitat (i.e., altering the conservation value of the habitat) (NMFS 2019e). Availability of Chinook salmon prey to killer whales may also be impacted by sound from vessels or other sounds sources if they raise average background noise within the animal's critical bandwidth to a level that is expected to chronically or regularly reduce echolocation space (Joy et al. 2019b; Veirs et al. 2016). However, none of the sound sources resulting from the proposed action that may overlap with Southern Resident killer whale inland critical habitat are considered chronic or long-lasting. Instead, exposure to vessel noise, weapons noise, and sonar as a result of the proposed action would be transient, brief and temporary, with no lasting or appreciably impact on killer whale's ability to migrate, feed or rest.

In addition, many of the activities within the Inland Waters portion of the action area that would result in acoustic stressors are proposed in areas that are either not part of the critical habitat designation (e.g., Hood Canal) or in areas that were excluded for national security reasons (refer to Table 21, Table 24, and Table 25). Proposed mitigation measures, as described in Section 3.5, would further reduce any potential effects of acoustic stressors on the Southern Resident killer whale's soundscape. As such, any effects of Navy acoustic stressors on inter-area passage conditions would not likely reduce the conservation value of Southern Resident killer whale inland critical habitat.

Seafloor devices used during NWTT activities that are deployed onto the seafloor and recovered include moored mine shapes, recoverable anchors, bottom-placed instruments, bottom cable arrays, energy harvesting devices, and robotic vehicles. While seafloor devices could obstruct killer whale movements, considering the size of these devices, the mobility of killer whales to avoid or swim around them, and the temporary nature of deployment, any effects associated with obstruction from seafloor devices would likely be insignificant. Thus, the effects of seafloor devices on the inter-area passage essential feature are not expected to reduce the conservation value of Southern Resident killer whale inland critical habitat.

NWTT activities involving the use of explosives in inland waters would only occur within two Navy EOD Training Range locations (Hood Canal and Crescent Harbor), neither of which overlaps spatially with designated inland critical habitat for Southern Resident killer whales. Hood Canal was not included in the inland critical habitat designation due to the lack of confirmed sightings of Southern Resident killer whales within this area. The EOD training range within Hood Canal is a sufficient distance from the entrance to Hood Canal such that any effects from the explosives proposed for use at this range (i.e., bin E0 and E3) would have no impact on any of the essential PBFs in areas that are included in the critical habitat designation (Navy 2020e).

Southern Resident killer whales have been sighted in waters near the Crescent Harbor EOD Range. The Crescent Harbor EOD Training Range was excluded from critical habitat designation because the benefits of exclusion for national security were found to outweigh the benefits of inclusion. However, sound from explosives during Navy mine neutralization training events at Crescent Harbor would likely overlap Southern Resident killer whales DPS designated critical habitat in the inland waters, as this training range is located outside of but adjacent to the Puget Sound portion of critical habitat (Navy 2020e).

Since sound and energy from explosions at the Crescent Harbor EOD Range would be short in duration and would occur a very limited number of times (up to six) within any year, there would be no plausible route to obstruct waterways or impact the conservation function of passage conditions to allow for migration, resting, and foraging in designated critical habitat. As noted above, any secondary effects of stressors (including explosives) resulting from the proposed

action on Southern Resident killer whales via impacts to water quality were found to be either insignificant or discountable.

Adverse effects on fish species that Southern Resident killer whales prey upon resulting from underwater detonations in Crescent Harbor could affect the prey species essential PBF of inland critical habitat (i.e., prey species of sufficient quantity, quality, and availability). As discussed above, although the Crescent Harbor EOD Training Range was excluded from critical habitat designation, the range to effects (i.e., mortality and injury) on fish extends into areas that are included in the designation. Previously (Section 8.2.1.4), as part of our jeopardy analysis we addressed the secondary effects on Southern Resident killer whale resulting from injury and mortality of prey fish species from explosions in all portions of the action area, including those areas that are not included in the Southern Resident killer whale designated (or proposed) critical habitat (e.g., Hood Canal, Quinault Range). In this section, for our critical habitat analysis, we focus on the effects of Navy explosives in the Crescent Harbor EOD Training Range on the prey essential PBF of inland critical habitat.

The use of Navy explosives at the Crescent Harbor EOD Range would likely result in the mortality and injury of ESA-listed fish preyed upon by Southern Resident killer whales. For purposes of our critical habitat effects analysis, we analyze only those impacts expected to occur within designated critical, and not within the excluded EOD range site. The Navy has indicated that the shortest distance between designated inland critical habitat and the nearest detonation point within the EOD range site is 610 meters (Navy 2020b). Since this distance is well beyond the estimated maximum range to mortality for the largest explosives proposed in the Crescent Harbor EOD site (i.e., bin E3 maximum range to onset mortality is 180 meters), we do not anticipate any mortality of salmon prey to occur within Southern Resident killer whale designated inland critical habitat. However, injury of salmon could occur within Southern Resident killer whale designated inland critical habitat since the average range to effects for bin E3 explosives is 1,674 meters (from Table 147 above).

As discussed above (Section 8.2.1.4), our analysis focuses on effects to Chinook salmon, coho salmon and chum salmon, since other Southern Resident killer whales prey species are not expected to account for a significant portion of their inland diet. Chinook salmon are the dominant component of the Southern Resident killer whales' diet in inland waters (Ford et al. 1998; Ford et al. 2016; Hanson et al. 2010). As such, effects on the Puget Sound Chinook salmon ESU likely represent the greatest potential risk to the prey PBF of inland critical habitat. Based on our analysis above on the effects of inland explosives on ESA-listed fish (Section 8.2.3.1), we estimate zero adult injuries and one adult mortality from the Puget Sound Chinook salmon ESU per year as a result of Navy explosives within the Crescent Harbor EOD Range. From Section 8.2.1.4, we do not expect more than one adult Puget Sound/Georgia Strait ESU coho salmon and one adult Puget Sound/Georgia Strait ESU chum salmon may be injured or killed each year from Navy explosive activities at Crescent Harbor. Thus, any effects from explosives within the Crescent Harbor EOD Range on adult salmon within Southern Resident killer whale inland

critical habitat are anticipated to be extremely minor. While this activity would likely impact a greater number of juvenile salmon inside critical habitat, Southern Resident killer whales typically seek out and feed on larger, adult salmon (Hilborn et al. 2012), which provide a much high energy content compared to juveniles (O'Neill et al. 2014). Since Southern Resident killer whales are likely targeting adult fish, juvenile injuries are anticipated to have much less of an effect on the prey PBF in terms of immediate impacts on Southern Resident killer whales prey availability.

Aside from immediate effects, injuries to juvenile salmon could lead to fitness consequences, which could affect the future availability of prey for Southern Resident killer whales. For juvenile Puget Sound Chinook salmon, from Section 8.2.3.1 we conservatively estimate that about 0.75 percent of the natural origin juvenile population and 0.05 percent of the hatchery origin juvenile population would be injured annually as a result of the proposed action. These estimates include fish injured both within critical habitat and outside of critical habitat (i.e., within the excluded area). Considering the distance between where explosives would likely occur and where critical habitat occurs, it is likely that a large number of these estimated injuries would occur outside of critical habitat. While the shortest distance between designated inland critical habitat and the nearest detonation point (i.e., 610 meters) is within the average range to injury (i.e., 1,674), the center of the EOD is approximately 1,450 meters, and many locations where detonations could occur are outside of this range to effects. Further, in general, the severity of injuries occurring within critical habitat would likely be reduced due to the distance from the detonation point to critical habitat. More of the higher severity injuries that are likely to result in fitness consequences would be expected to occur closer to the detonation point (i.e., within the EOD range site which is excluded from critical habitat).

As noted previously (Section 2.3.2), our range to effects for fish injury are based on the more conservative acoustic injury criterion of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors, and others described in Section 8.2.3.1 above, likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

In addition, the Navy has proposed to continue procedural mitigations that may further reduce the estimated impacts of inland explosives on the prey PBF of Southern Resident killer whale critical habitat. At the Crescent Harbor EOD Range, the Navy will conduct explosive activities at least 1,000 meters from the closest point of land to avoid or reduce impacts on fish (e.g., juvenile Chinook salmon) in nearshore habitat areas (see Section 3.5.1.9 for details). Therefore, based on the information available, we do not expect injuries to juvenile Puget Sound Chinook salmon within critical habitat to have a measurable effect on the future availability of prey for Southern Resident killer whales.

Although we could not quantify, we expect higher densities of juvenile coho and chum salmon, as compared to adults, would be found near the Crescent Harbor EOD Range site. However, we anticipate the number of juvenile coho and chum salmon exposed to make up a very small percent of the total juvenile population of the large and expansive Puget Sound/Georgia Strait ESUs for these species. Similar to Chinook salmon, many of the impacts from this activity, including a large portion of the more severe injuries that would likely lead to fitness consequences, are expected to occur within the excluded EOD range site. As discussed above, Navy mitigation measures may further reduce impacts to juvenile salmon within inland critical habitat. Therefore, based on the information available, we do not expect injuries to juvenile Puget Sound/Georgia Strait coho or chum within critical habitat to have a measurable effect on the future availability of prey for Southern Resident killer whales.

Following an explosion, the abundances of prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Any impacts to prey availability within an area would be temporary, only occurring during activities involving explosives, and no lasting effect on prey availability or the pelagic food web would be expected. Further, any prey species injured, but not killed, by an explosion could result in the prey item being more easily captured by a Southern Resident killer whale.

In addition to physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that might include swimming to the surface or scattering away from the source. Behavioral effects on Southern Resident prey resulting from reactions to sound created by the explosions would be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion.

While explosives may result in injury to Southern Resident killer prey species within designated inland critical habitat, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of this endangered DPS. As such, we find the effects of inland explosives on the prey PBF of Southern Resident killer whale designated inland critical habitat to be insignificant.

In summary, many of the stressors associated with activities in the Inland Waters portion of the action area would either not overlap with Southern Resident killer whale inland critical habitat or would have no pathway to affect the PBFs of this critical habitat. The effects of all other stressors analyzed, including sonar and explosives, on the critical habitat PBFs were found to be either insignificant or discountable, and not likely to reduce the conservation value of inland critical habitat. Thus, we conclude that the proposed action may affect but is not likely to adversely affect Southern Resident killer whale designated inland critical habitat.

8.3.1.2.2 Proposed Critical Habitat – Coastal Areas

The PBFs of inland critical habitat and proposed coastal critical habitat for Southern Resident killer whales are the same (i.e., water quality, prey, or passage conditions). For many of the activities and resulting stressors associated with the proposed action, we would expect the pathway to effects and potential impacts on these PBFs to be the same for areas designated as inland critical habitat and areas proposed as coastal critical habitat. Thus, for these activities/stressors, we can apply the same logic and reach the same conclusion regarding the effects on the PBFs of inland critical habitat (as discussed above) as we do for proposed coastal critical habitat. As we concluded for inland critical habitat above:

- Several of the activities that may overlap with Southern Resident killer whale proposed coastal critical habitat would have no pathway through which the resulting stressors could affect any of the essential features of critical habitat (i.e., water quality, prey, or passage conditions). These stressors include electromagnetic devices, expended wires and cables, small decelerators and parachutes, and ingestion stressors including fragments from non-explosive practice munitions, explosives, and targets.
- The inter-area passage essential feature of Southern Resident killer whale proposed coastal critical habitat would not likely be adversely affected by the minor, temporary and short-term effects of sound from Navy vessels and aircraft.
- Any secondary effects of potential stressors (including explosives) resulting from the proposed action on Southern Resident killer whales via impacts to water quality were found to be either insignificant or discountable.
- The effects of Navy sonar and other transducers on the quantity, quality, and availability of Southern Resident killer whale prey would not likely reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.
- Any effects of Navy vessels and in-water devices on the quantity, quality, and availability of Southern Resident killer whale prey would not likely reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.
- The effects of seafloor devices on the inter-area passage essential feature are not expected to reduce the conservation value of Southern Resident killer whale inland critical habitat.

The following activities (or potential stressors) would only occur in the offshore portion of the action area: weapons firing and impact noise (other than small caliber gunnery using blanks); laser testing; medium and large decelerators and parachutes; chemical and biological simulants; and expended chaff and flares. Since we did not previously evaluate these for inland critical habitat above, we evaluate the effects of these activities/stressors on the PBFs (i.e., water quality, prey, or passage conditions) of Southern Resident killer whale proposed coastal critical habitat here.

From our analysis above (Section 8.1.3.1), we determined that the effects of weapons noise on ESA-listed fishes, including salmonid species that Southern Resident killer whales feed on, are

anticipated to be minor, temporary, and are not expected to lead to a significant disruption of normal behavior patterns (i.e., insignificant effects). By the same logic, we would expect the effects of weapons noise on non-listed salmonids that Southern Resident killer whales feed on to also be insignificant. Thus, the effects of weapons firing and impact noise on the prey PBF are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat. Also from our effects analysis above (see Section 8.1.1.1), ESA-listed cetaceans (including Southern Resident killer whales) are either not likely to respond to Navy weapons noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns that include, but are not limited to, breeding, feeding or sheltering. If they do occur, behavioral reactions would likely be short-term (seconds to minutes) and multiple exposures of the same animal over a short duration are not anticipated. For these reasons, the effects of weapons firing and impact noise on the passage conditions PBF are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat. Proposed mitigations in the offshore portion of the action area would likely further minimize exposures of Southern Resident killer whales and their prey to weapons noise, thus reducing potential impacts of this stressor on the passage conditions and prey PBFs. Based on recent tagging data, in the offshore environment Southern Resident killer whales spend about 95 percent of their time within 21.1 mi of shore and about 83 percent within 12.4 mi of shore (Hanson et al. 2017b). Within the Marine Species Coastal Mitigation Area, no non-explosive small- and medium-caliber gunnery activities would occur within 12 nautical miles of the coast and no non-explosive large-caliber gunnery activities (with associated blast noise) would occur within 20 nautical miles of the coast (see Section 3.5.2.2 for details). In addition, procedural mitigation for weapons firing noise described in Section 3.5.1.3 would reduce the potential for exposure to large-caliber gunnery blast noise for any animals present farther offshore. All missile and kinetic energy weapon use would occur greater than 50 nautical miles from shore, where the Southern Resident killer whales DPS is not present. There is no pathway to effects from weapons firing and impact noise on the water quality PBF of proposed coastal critical habitat.

Based on our effects analysis above (Section 8.1.3.2), the effects from high-energy laser weapons on the ESA-listed fish species considered in this opinion, including salmonid species that Southern Resident killer whales feed on, are discountable. By the same logic, we would expect the effects of weapons noise on non-listed salmonids that Southern Resident killer whales feed on to also be discountable. Also from above (Section 8.1.1.2), we find it extremely unlikely that ESA-listed cetaceans would be exposed to high energy lasers (i.e., discountable). There is no pathway to effects from lasers on the water quality PBF of proposed coastal critical habitat. Thus, the effects of Navy laser use as part of the proposed action are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

While decelerators and parachutes represent an entanglement risk for marine mammals and fish, from our analysis above we concluded that the chances of an ESA-listed marine mammal (Section 8.1.1.4) or ESA-listed fish (Section 8.1.3.4) encountering a Navy decelerator or

parachute in the action area and becoming entangled is extremely low. By the same logic, we would expect the chances of any non-listed fish that Southern Resident killer whales feed on encountering a Navy decelerator or parachute in the action area and becoming entangled is also extremely low. There is no pathway to effects from the use of decelerators and parachutes on the water quality PBF of proposed coastal critical habitat. Thus, the effects of the Navy's proposed use of decelerators and parachutes as part of the proposed action are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

As discussed above (Section 8.1.3.6), the effects of chemical and biological simulants used during Navy training and testing on ESA-listed fishes via water quality and prey are discountable (i.e., extremely unlikely to occur). Applying the same logic, we would expect the effects of chemical and biological simulants on non-listed fish that are part of the Southern Resident killer whales prey PBF via water quality and prey are also discountable. While the water quality PBF of proposed coastal critical habitat could be exposed to the effects of chemical and biological simulants, such effects, if they occur at all, would be extremely short-term and localized. Simulants released in the offshore portion of the action area would be expected to disperse rapidly into the ocean environment. There is no pathway to effects from chemical and biological simulants on the passage conditions PBF of proposed coastal critical habitat. Thus, the effects of the Navy's proposed use of chemical and biological simulants as part of the proposed action are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

Based on our effects analysis above, we found that Southern Resident killer whales prey (i.e., ESA-listed salmonids) were not likely to be adversely affected by expended chaff and flares resulting from the proposed action. By the same logic, non-listed fish species prey upon by Southern Resident killer whales are also not likely to be adversely affected by expended chaff and flares. Thus, the effects of expended chaff and flares on the prey essential feature are not expected to reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat. There is no pathway to effects from expended chaff and flares on the passage conditions and water quality PBFs of proposed coastal critical habitat.

The Navy's quantitative analysis estimates 46 exposures resulting in behavioral harassment and 2 exposures resulting in TTS per year as a result of NWTT activities in the offshore area (see Section 8.2.1.1 for details). No exposures resulting in PTS of Southern Resident killer whales were predicted by the model. With an estimated population size of 73, each individual killer whale would be exposed to Navy sonar at levels resulting in behavioral harassment or TTS less than once per year, on average. While some may be exposed more frequently (and others less) than average, based on the quantitative analysis we would not expect more than one instance of TTS and a few instances of behavioral harassment for any individual whale in a given year. In addition, some proportion of these exposures would likely occur within the Quinault Range Site, which was excluded from proposed coastal critical habitat. Since the proposed use of sonar and other transducers during NWTT activities would be transient and temporary, any effects on

Southern Resident killer whales would likely be temporary. Adverse habitat-related effects from acoustic stressors are more likely to result from chronic or long-lasting noise sources that degrade the value of habitat by interfering with the sound-reliant animal's ability to gain benefits from that habitat (i.e., altering the conservation value of the habitat) (NMFS 2019e). Instead, exposure to sonar as a result of the proposed action would be transient, brief and temporary, with no lasting or appreciably impact on killer whale's ability to migrate, feed or rest. In addition, the passage conditions PBF may be less likely to be impacted in coastal ocean waters compared to the more geographically constricted inland waters because killer whales would likely be able to more easily navigate around potential acoustic obstructions in the open ocean (84 FR 49214).

While whales may learn to avoid particular areas where a reoccurring and harmful sound source is located, this is not likely for Navy offshore activities which are generally transient in nature and widely dispersed over a very large action area. Also, as noted above, many Navy activities involving the use of sonar and other transducers would occur within the excluded Quinault Range Site. In addition, proposed mitigation measures, as described in Section 3.5, would further reduce any potential effects of Navy sonar on the passage conditions PBF. As such, any effects of Navy sonar and other transducers on the inter-area passage conditions essential PBF would not likely reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore, and, therefore, outside of the areas proposed as Southern Resident killer whale coastal critical habitat. Sound and energy from explosive mine countermeasure and neutralization testing activities within (or adjacent to) proposed coastal critical habitat would be short in duration, infrequent and would occur over a very limited number of days (i.e., approximately two events per year, from 1 to 10 days per event, a maximum of five E7 and 36 E4 explosions per year) within any year. Some proportion of the explosions associated with this activity would likely occur within the excluded Quinault Range Site, within the excluded 10 km buffer around the range site, or beyond the 200 meters depth contour that defines the western edge of proposed critical habitat. As noted above, acoustic stressors that obstruct or interfere with passage condition or movement are more likely associated with chronic or long-lasting exposure to acoustic sources that appreciably degrade the value of the animal's soundscape. By contrast, the effects of the Navy's proposed use of explosives in offshore areas that could overlap with proposed coastal critical habitat would be episodic and temporary. In addition, the Navy's quantitative analysis, using the maximum number of explosions per year under the proposed action, estimates no impacts to Southern Resident killer whales from explosives use as part of NWTT Phase III activities. Procedural mitigation involving lookouts for explosive mine countermeasure and neutralization activities will likely further reduce any potential impacts of explosives on Southern Resident killer whales killer movement in offshore areas. Therefore, any effects of Navy explosives on the inter-area

passage conditions essential PBF would not likely reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

Next, we consider the effects of the Navy's proposed offshore explosive mine countermeasure and neutralization testing events on the prey essential PBF of Southern Resident killer whale proposed coastal critical habitat. Mine countermeasure and neutralization testing involving the use of explosives would occur in waters three nautical miles or greater from shore within the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nautical miles or greater from shore elsewhere in the NWTT offshore area. The use of Navy explosives within (or adjacent to) areas proposed as coastal critical habitat would likely result in the mortality and injury of salmon populations preyed upon by Southern Resident killer whales. Following an explosion, the abundances of prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Any impacts to prey availability within an area would likely be temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web would be expected. Salmon that are injured (but not killed) by an explosion could be more easily captured as prey by Southern Resident killer whales actively feeding in the area. In addition to physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that might include swimming to the surface or scattering away from the source. Behavioral effects on Southern Resident prey resulting from reactions to sound created by the explosions would be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion.

Our analysis of the effects of the proposed action on the prey essential PBF of Southern Resident killer whale proposed coastal critical habitat is focused on the effects of explosives on Chinook salmon populations. While Southern Resident killer whales may feed opportunistically on other salmonid species (i.e., coho salmon, chum salmon, and steelhead) and non-salmonids (e.g., halibut, herring, and lingcod), we have no information from the limited studies conducted on Southern Resident killer whale diet to indicate that species other than Chinook salmon are a major component of their offshore diet. Studies on the Southern Resident killer whale diet indicate that Chinook salmon is their primary prey both in inland waters and offshore and during all months of the year (Ford et al. 2010b; Ford et al. 2016; Hanson In prep; Hanson et al. 2010; Hilborn et al. 2012; NMFS 2019e). As such, mortality and injury of Chinook salmon likely represents the greatest impact to the prey PBF of proposed coastal critical habitat resulting from Navy mine countermeasure and neutralization detonations. Southern Resident killer whales likely travel considerable distances to locate Chinook salmon aggregations and the seasonal timing of the return of this high-energy preferred prey to different river systems likely influences their movements along the Northwest Pacific coast (NMFS 2019e).

The estimated number of Chinook salmon mortalities and injuries as a result of Navy offshore explosions were summarized (by ESU and life stage) in Section 8.2.1.4 (Table 135 for ESA-listed ESUs and Table 136 for non-listed ESUs) above, where we discuss the effects on Southern Resident killer whales. The mortality of an adult Chinook salmon can have immediate impacts on the availability of prey for Southern Resident killer whales, as well as the reproductive potential of the population. From our analysis above we found that an extremely small percentage of each Chinook salmon ESU would likely be killed from NWTT explosive activities during a maximum year of explosive activity levels (i.e., less than 0.02 percent across all ESUs, life stages, and hatchery/natural origin). We concluded that the impacts of salmon mortalities as a result of the proposed action on the nutritional needs of the Southern Resident killer whales population would likely be extremely small.

From our quantitative analysis (Section 8.2.1.4), we expect a significantly larger number of salmon would be injured (as compared to killed) as a result of NWTT explosives. Several of the highest priority Chinook salmon ESUs for Southern Resident killer whale conservation, also had some of the highest estimated injury rates (as a proportion of the population size) from offshore explosives among the ESUs analyzed. These include the Snake River fall run ESU, Snake River spring/summer ESU, Upper Columbia River fall run ESU, and Upper Columbia River spring run ESU (all with estimated adult injury rates of between 1.0 and 1.2 percent). As discussed in Section 8.2.1.4, we expect that some proportion of salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected.

Whereas our species effects analysis considered the impacts to Southern Resident killer whale prey in all portions of the action area, our proposed critical habitat effects analysis is more narrowly focused only on those impacts that are likely to occur within proposed critical habitat. As noted above, some proportion of the explosions associated with this activity would likely occur outside of proposed coastal critical habitat (i.e., either within the excluded Quinault Range Site, the excluded buffer area, or beyond the 200 meters depth contour that defines the western edge of proposed critical habitat). Thus, any effects on prey from detonations within the Quinault Range Site should not be considered in this analysis. Given the range to effects for the explosives proposed (i.e., bins E4 and E7), depending on the distance to proposed critical habitat, detonations within the buffer area or beyond the 200 meter depth contour could still impact Southern Resident killer whale prey within proposed critical habitat.

The Navy has proposed to conduct up to two mine countermeasure neutralization testing events per year. Based on information provided by the Navy, it is likely that at least one of these events (and possibly both events) within a given year would be conducted within the Quinault Range Site (Navy 2020d). Mine countermeasure and neutralization testing is limited by availability of certain depths, bottom types, and other bathymetric conditions (e.g., water clarity) which may only be found in certain portions of the Quinault Range Site (Navy 2020d). To meet mine warfare testing requirements in the NWTT action area, the Navy needs to have the ability to conduct mine countermeasure and neutralization testing in relatively shallow water environments

(typically, in water depths of less than 100 meters). This testing activity is typically conducted in water depths of 300 ft., and in some cases may require testing in water depths of less than 100 feet (Navy 2020d). As the Quinault Range Site is the only portion of the offshore area that goes as far landward as three nautical miles from shore (outside of the Olympic Coast National Marine Sanctuary), mine countermeasure and neutralization events requiring shallower water depths would likely be limited to certain portions of the range (Navy 2020d). In addition, the Navy has proposed to not conduct explosive mine countermeasure and neutralization testing activities in the Olympic Coast National Marine Sanctuary, the Juan de Fuca Eddy Marine Species Mitigation Area, the Stonewall and Heceta Bank Humpback Whale Mitigation Area (from May through November), and the Point St. George Humpback Whale Mitigation Area (from July through November). These mitigation area measures would further restrict the available areas where the Navy could conduct explosive mine countermeasure and neutralization testing activities.

It is difficult to assess how reductions in prey abundance may vary throughout proposed critical habitat, and we have less confidence in our understanding of how reductions could result in localized depletions in the areas of proposed critical habitat. Given the likelihood that a large proportion of the mine countermeasure and neutralization explosives would occur within the Quinault Range Site, we expect the impacts of the proposed action on Southern Resident killer prey within proposed coastal critical habitat to be considerably smaller than the overall impacts to prey throughout the action area (as described in our species effects analysis in Section 8.2.1.4). Mine countermeasure and neutralization explosives detonated within proposed coastal critical habitat would be at least 12 nautical miles from shore, as compared to those with the Quinault Range Site which could as close as three nautical miles for E4 explosives and six nautical miles for E7 explosives. Although densities vary by population, season, and location, in general we would expect higher densities of Chinook salmon in coastal areas closer to shore, particularly during certain times of year associated with annual migrations.

In summary, while explosives may result in injury and mortality to Southern Resident killer whale prey species within proposed coastal critical habitat, based on our analysis we find that this stressor would not have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of this endangered DPS. The anticipated annual number of salmon mortalities from NWTT offshore explosives represents an extremely small proportion of the number of salmon required to meet the nutritional needs of the Southern Resident killer whale DPS.

While a larger number of Chinook salmon would be injured as a result of offshore explosives, we expect a large majority of the estimated injuries would occur either within the excluded Quinault Range Site, within the excluded buffer, or outside of proposed critical habitat (i.e., beyond the 200 meter depth contour). Salmon that are injured (but not killed) by an explosion could be more easily captured as prey by Southern Resident killer whales actively feeding within the proposed critical habitat area. In addition, as discussed previously, our fish effects analysis is

based on a number of conservative assumptions that likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

Further, the Navy has proposed mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months of October through June (see Section 3.5.1.9 for details). Since this timeframe includes months when peak densities of many Chinook salmon ESUs would be expected in the action area, this mitigation measure would likely result in reduced impacts to several of the Chinook salmon ESUs preyed upon by Southern Resident killer whales. Southern Residents are also more likely to occur in the offshore portion of the action area from October through June as compared to July through September, when they are more often seen in the inland waters. These conservation measures would likely further reduce the impacts of offshore explosives on the prey PBF of Southern Resident killer whales. As such, we find the effects of explosives on the prey PBF of Southern Resident killer whale proposed coastal critical habitat to be insignificant.

In summary, many of the stressors associated with activities in the offshore portion of the action area would either not overlap with designated critical habitat or would have no pathway to affect the PBFs of Southern Resident killer whale proposed coastal critical habitat. The effects of all other stressors analyzed, including sonar and explosives, on the essential PBFs were found to be either insignificant or discountable and not likely to reduce the conservation value of proposed critical habitat. Thus, we conclude that the proposed action is not likely to adversely affect Southern Resident killer whale proposed coastal critical habitat.

8.3.2 Leatherback Sea Turtle Designated Critical Habitat

Leatherback sea turtle designated critical habitat is described in Section 6.2.8. Here we evaluate the effects of the proposed action on the one identified essential physical and biological feature of this critical habitat: the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (e.g., Chrysaora, Aurelia, Phacellophora, and Cyanea), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

In general, very little is known about sound detection and the use of sound by aquatic invertebrates (Budelmann 2010). Organisms may detect sound by sensing either the particle motion or pressure component of sound, or both. Aquatic invertebrates, including jellyfish and other leatherback prey, probably do not detect pressure since they are the same density as water and they lack air cavities that would function like the fish swim bladder in responding to pressure (Budelmann 2010). Because any acoustic sensory capabilities, if present at all, are limited to detecting water motion, and water particle motion near a sound source falls off rapidly with distance, aquatic invertebrates are probably limited to detecting nearby sound sources rather than sound caused by pressure waves from distant sources (Navy 2019a). Only jellyfish in very close proximity to Navy sonar sources within their hearing range could be exposed for a short duration and would likely experience minor and temporary effects. Similarly, exposure to other acoustic stressors (i.e., vessel noise and weapons noise) would likely either have no effect or very minor

effect on leatherback prey species. Impacts, if any, to leatherback prey species would not be expected to occur on a scale necessary to affect the overall prey availability for leatherback turtles. The use of sonar and other transducers and other acoustic sources during NWTT activities would not have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. In addition, acoustic stressors are not mentioned by the leatherback Critical Habitat Review Team as an activity that may impact the prey essential feature. In summary, we find that any impacts on leatherback critical habitat through effects to prey from sonar and other transducers would likely be insignificant and thus not result in a reduction of the value of critical habitat for conservation of the species.

Leatherback prey species could be affected by physical disturbance and strike stressors including vessel strike, military expended materials, and in-water devices (see Section 5.3 for details on these stressors). However, as with acoustic stressors, it is unlikely that such impacts would occur on a spatial or temporal scale that would result in a measureable impact on the condition, distribution, diversity, abundance or density of prey species necessary for growth and success of leatherback sea turtles. In summary, we find that any impacts on leatherback critical habitat through effects to prey from physical disturbance and strike stressors resulting from the proposed action would likely be insignificant and thus not result in reduction of value of critical habitat for conservation of the species.

Jellyfish and other invertebrate prey species could be vulnerable to effects from Navy explosives. As discussed above for sonar, only jellyfish in relatively close proximity to a blast would likely be exposed at levels that could result in injury or mortality. Proposed mitigation measures may further reduce the impacts of NWTT explosives on leatherback critical habitat. Procedural mitigation includes ceasing explosive detonations if a sea turtle is observed in the mitigation zone (see Section 3.5.1). The Navy will also implement mitigation measures for in-water explosives within mitigation areas, which may reduce impacts on leatherback prey species (see Section 3.5.2). Leatherback sea turtle critical habitat overlaps part of the Marine Species Coastal Mitigation Area. The Navy will not conduct explosive training and testing activities within 50 nautical miles from shore, with the exception of explosive mine countermeasure and neutralization testing activities. Leatherback sea turtle critical habitat also encompasses the Olympic Coast National Marine Sanctuary Mitigation Area. Within the Olympic Coast Mitigation Area and Juan de Fuca Eddy Marine Species Mitigation Area, the Navy will not conduct any explosive training and testing activities.

Explosives are not mentioned by the leatherback Critical Habitat Review Team as an activity that may impact the prey essential feature. Impacts, if any, to leatherback prey would not likely occur on a temporal or spatial scale necessary to affect the overall prey availability for leatherback turtles. In summary, although explosives may result in injury and mortality to leatherback prey species, we have no information to indicate that this stressor would have a measureable impact

on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. Any impacts of explosives resulting from NWTT activities on leatherback sea turtle prey would likely be insignificant. The effects of all stressors analyzed on the essential PBF were found to be either insignificant or discountable and not likely to reduce the conservation value of designated critical habitat. Therefore, for the reasons provided above, we determine that the proposed action may affect but is not likely to adversely affect leatherback sea turtle designated critical habitat.

8.3.3 Fishes

Below, we analyze the effects of the proposed action on ESA-listed fish designated critical habitat that may be affected by the proposed action.

8.3.3.1 Bocaccio and Yelloweye Rockfish Puget Sound/Georgia Basin DPS Designated Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPSs of bocaccio and yelloweye rockfish overlap with the inland portion of the NWTT action area. The essential PBFs for ESA-listed rockfish are: 1) Quantity, quality, and availability of prey species to support individual growth, survival, and feeding opportunities; 2) Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities; and 3) For deepwater habitat, the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance. For a more detailed description of rockfish designated critical habitat in the action area refer to Section 6.2.37.

In general, only a relatively small number of the proposed NWTT activities that occur in the inland portion of the action area would overlap spatially with rockfish critical habitat. Many of the stressors resulting from these activities would have no effect on rockfish prey or water quality within designated critical habitat. As discussed in Section 8.1.3.6 (*Stressors Resulting in Effects to Fish Habitat or Prey*), the secondary effects of stressors resulting from the proposed action on ESA-listed rockfish via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage).

NWTT activities involving the use of explosives in inland waters would only occur within two Navy EOD Training Range locations (Hood Canal and Crescent Harbor). Both of these locations were excluded from bocaccio and yelloweye rockfish critical habitat designation because the benefits of exclusion for national security were found to outweigh the benefits of inclusion. The EOD training range within Hood Canal is a sufficient distance from the entrance to Hood Canal such that any effects from the explosives proposed for use at this range (i.e., bin E0 and E3) would have no impact on any of the essential PBFs in areas that are included in the critical

habitat designation. Sound from explosives during Navy mine neutralization training events at Crescent Harbor could overlap ESA-listed rockfish designated critical habitat in the inland waters, as this training range is located outside of but adjacent to critical habitat (Navy 2020e).

The Navy's use of explosives at inland EOD ranges within Crescent Harbor could impact nearshore (< 30 meters) critical habitat designated for juvenile bocaccio. Underwater explosions may reduce available pelagic prey items within juvenile bocaccio critical habitat by either directly killing prey or by scaring them from the area. The abundance of fish prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Any of these scenarios would likely be short-term and temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web expected. Due to the infrequent use of underwater explosives (up to six events per year at each EOD range) and the limited locations at which underwater explosives are used, it is not expected their use will have a persistent effect on prey availability or the health of the aquatic food web.

As discussed above (Section 8.1.3.6), the effects of explosive byproducts, metals, and chemicals introduced into the marine environment from NWTT activities on ESA-listed fish via impacts on water quality are considered discountable.

Precision anchoring could affect the critical habitat PBF associated with structure and rugosity in deepwater habitats occupied by adult bocaccio and adult and juvenile yelloweye rockfish. The Navy has proposed between 30-40 precision anchoring events per year in the inland portion of the action area at the following four designated locations (shown as blue hexagons in Figure 31 above): Naval Magazine Indian Island, Naval Station Everett, Navy 3 OPAREA, and Eastern Banks Area. Precision anchoring involves a vessel navigating to a precise, pre-determined location and releasing the ship's anchor to the seafloor. The coordinates of these locations are provided during the exercise, and operators are to anchor within 100 feet of the location. When considering these buffer areas that surround the precision anchoring drill sites, all of the precision anchoring sites overlap some portion of rockfish critical habitat. This activity could damage rocky reefs and reduce the structural complexity of deepwater habitat areas used by rockfish. However, impacts from precision anchoring would likely by highly localized, mainly restricted to the same few designated anchorages that have been used by the Navy for many years. The total bottom area affected by precision anchoring would be extremely minimal compared to the 414.1 square miles of deepwater critical habitat designated for these species. In addition, the Navy has proposed to implement mitigation that includes not conducting precision anchoring (except in designated areas) within the anchor swing circle of live-hard bottom (rocky reef), artificial reefs, and shipwrecks to avoid potential impacts from seafloor devices (including anchors and anchor bases) on seafloor resources.

In summary, impacts, if any, to the prey PBF of ESA-listed rockfish critical habitat would not likely occur on a temporal or spatial scale necessary to affect the overall prey availability for

these species. Although explosives may result in injury and mortality to bocaccio prey species, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of bocaccio. Any impacts of explosives resulting from NWTT activities on bocaccio prey would likely be insignificant. While precision anchoring could affect the rockfish critical habitat PBF associated with structure and rugosity in deepwater habitats, the area potentially affected would be extremely small compared to the total area of deepwater critical habitat designated such that any effects would be insignificant. The effects of all stressors analyzed on the essential PBFs were found to be either insignificant or discountable and not likely to reduce the conservation value of designated critical habitat. Therefore, for the reasons provided above, we determine that the proposed action may affect but is not likely to adversely affect bocaccio and yelloweye rockfish designated critical habitat.

8.3.3.2 Chinook Salmon Puget Sound ESU and Chum Salmon Hood Canal Summer-Run ESU Designated Critical Habitat

Designated critical habitat for Puget Sound Chinook salmon ESU and Hood Canal summer-run chum salmon includes rivers, lakes (for Chinook salmon only) and nearshore marine habitat. The nearshore marine habitat component of critical habitat for these two species overlaps with the inland portion of the NWTT action area. The identified estuarine and nearshore marine PBFs within the inland portion of the action area include the following: areas free of obstruction; water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh-and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. As there is no mechanism for the stressors associated with the proposed action (Section 5) to affect the water quantity or salinity, these PBFs will not be discussed further. While the critical habitat rule (70 FR 52630) describes PBFs for offshore marine areas, NMFS has not designated any specific areas based on this PBF for these species. Therefore, offshore marine areas will not be evaluated as part of this analysis. For a more detailed description of Puget Sound Chinook salmon ESU and Hood Canal summer-run chum salmon designated critical habitat in the action area refer to the species' respective status sections above (Section 6.2).

Many of the stressors resulting from NWTT activities would have no effect on water quality, salmonid natural cover, or salmonid forage within designated critical habitat. As discussed in Section 8.1.3.6 (*Stressors Resulting in Effects to Fish Habitat or Prey*), the secondary effects of stressors resulting from the proposed action on ESA-listed salmonids via impacts to their habitat or prey were all found to be either insignificant or discountable.

Seafloor devices used during NWTT activities that are deployed onto the seafloor and recovered include moored mine shapes, recoverable anchors, bottom-placed instruments, bottom cable

arrays, energy harvesting devices, and robotic vehicles. While seafloor devices could obstruct fish movements, considering the size of these devices, the mobility of salmonids to avoid or swim around them, and the temporary nature of deployment, any effects associated with obstruction from seafloor devices would likely be insignificant, and thus not result in reduction of value of critical habitat for conservation of the species.

In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). None of the proposed NWTT activities involving explosives would occur within areas designated as Puget Sound Chinook salmon ESU or Hood Canal summer-run chum salmon critical habitat (i.e., explosives would occur either in the offshore portion or within inland EOD ranges in Puget Sound which have been excluded from critical habitat designation). Sound from explosives detonated at Crescent Harbor could overlap ESA-listed salmon designated critical habitat in the inland waters, as this training range is located outside of but adjacent to critical habitat (Navy 2020e). Underwater explosions may reduce available pelagic prey items within Chinook salmon and chum salmon critical habitat by either directly killing prey or by scaring them from the area. The abundance of fish prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Any of these scenarios would likely be short-term and temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web expected. Due to the infrequent use of underwater explosives (up to six events per year at each EOD range) and the limited locations at which underwater explosives are used, it is not expected their use will have a persistent effect on prey availability or the health of the aquatic food web.

In summary, while seafloor devices could temporarily obstruct fish movements, any effects associated with obstruction from seafloor devices on Puget Sound Chinook salmon ESU or Hood Canal summer-run chum salmon critical habitat would likely be insignificant. Impacts, if any, to the forage PBF of Puget Sound Chinook salmon ESU or Hood Canal summer-run chum salmon critical habitat would not likely occur on a temporal or spatial scale necessary to affect the overall prey availability for these species. Although explosives may result in injury and mortality to forage species, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support the growth and maturation of individual Puget Sound Chinook salmon ESU or Hood Canal summer-run chum salmon. Thus, any impacts of explosives resulting from NWTT activities on the forage PBF of these species would likely be insignificant. The effects of all stressors analyzed on the essential PBFs were found to be either insignificant or discountable and not likely to reduce the conservation value of designated critical habitat. In conclusion, for the reasons provided above, we determine that the proposed action may affect but is not likely to adversely affect Puget Sound Chinook salmon ESU or Hood Canal summer-run chum salmon designated critical habitat.

8.3.3.3 Green Sturgeon Southern DPS Designated Critical Habitat

Critical habitat for the Southern DPS of green sturgeon overlaps with the offshore coastal marine portion of the NWTT action area within the 60 fathoms depth contour and within the Strait of Juan de Fuca. The attributes of essential PBFs for ESA-listed green sturgeon in coastal marine areas include food resources, migratory corridor, and water quality. The attributes of essential PBFs in estuarine areas include the three above for marine areas as well as three additional attributes: water flow, water depth, and sediment quality. As there is no mechanism for the stressors associated with the proposed action (Section 5) to affect the water flow or water depth attributes of green sturgeon critical habitat, these PBFs will not be discussed further. For a more detailed description of green sturgeon designated critical habitat in the action area refer to Section 6.2.25.

Many of the stressors resulting from NWTT activities would have no effect on green sturgeon food resources, migratory corridor, water quality or sediment quality within designated critical habitat. As discussed in Section 8.1.3.6 (*Stressors Resulting in Effects to Fish Habitat or Prey*), the secondary effects of stressors resulting from the proposed action on ESA-listed green sturgeon via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Since the effects of explosive byproducts, metals, and chemicals introduced into the marine environment from NWTT activities on ESA-listed fish via impacts on water or sediment quality were found to be discountable, these stressors are not expected to reduce the value of the water quality PBF of green sturgeon coastal marine critical habitat.

Most of the proposed NWTT activities involving explosives would occur outside of areas designated as green sturgeon critical habitat (i.e., either farther offshore or within inland EOD range sites in Puget Sound). However, the Navy has proposed a new offshore mine countermeasure and neutralization testing activity that would occur closer to shore than other activities that involve the use of in-water explosives. This activity would typically occur greater than three nautical miles from shore in the Quinault Range Site, or greater than 12 nautical miles from shore elsewhere in the offshore area. The Navy has proposed two mine countermeasure and neutralization testing events per year in the offshore area. Up to 5 mine explosives (bin E7; up to 60 pounds NEW) and 36 mine neutralizers (bin E4; up to 5 pounds NEW) could be used annually across the two events. The Navy's proposed action also includes a cap on the seven year total number of offshore explosives for this activity of 108 E4 and 15 E7 explosives. The Navy has also proposed mitigation measures that would reduce the likelihood of overlap between explosive activity and Southern DPS green sturgeon. This includes seasonal mitigation that limits the number of explosives that could be used during the higher green sturgeon density months of October through June to 3 E7 and 20 E4 explosives annually, and no more than 9 E7 and 60 E4 explosives over a seven-year period.

This new activity could impact the migratory corridor and food resources PBFs of coastal marine critical habitat designated for Southern DPS green sturgeon. The migratory corridor PBF is defined as a migratory pathway necessary for the safe and timely passage of Southern DPS green sturgeon within marine and between estuarine and marine habitats (74 FR 52300). Safe and timely passage is further defined to mean that human-induced impediments, either physical, chemical, or biological, do not alter the migratory behavior of the fish such that its survival or the overall viability of the species is compromised. As discussed in Section 8.1.3.6, with the exception of explosives, all other stressors associated with the proposed action were found to not likely adversely affect Southern DPS green sturgeon. While some stressors may overlap spatially with the migratory pathway (e.g., vessels, seafloor devices), none of the stressors addressed in Section 8.1.3.6 would likely impede green sturgeon migration in a meaningful or measureable way. Any effects on migration from these stressors (other than explosives) would likely be short-term and minor, with sturgeon easily swimming around or avoiding the impeding object in its path.

The Navy's proposed use of explosives in coastal marine critical habitat could overlap with green sturgeon migratory routes. In Section 8.2.3.1, we analyzed the effects of offshore explosives on individual green sturgeon exposed, including injury and mortality. Here, we evaluate the effects of offshore explosives on the migratory corridor PBF of Southern DPS green sturgeon critical habitat. Large explosives, in particular, could result in behavioral responses which may temporarily disrupt green sturgeon migrations. Range to effects (i.e., mortality or injury) for the proposed use of explosives (i.e., bins E4 and E7) in green sturgeon critical habitat vary from a few hundred to a few thousand meters. Range to effects for behavioral responses, including startling, avoidance, disorientation and stress, are likely greater. Two such explosive events, ranging from one to ten days per event, could overlap with green sturgeon coastal marine critical habitat are proposed annually. Explosions would be brief and occur sporadically over the course of an event. Exposed green sturgeon would likely resume their normal migratory behavior shortly after the explosion occurs. Although explosives could result in behavioral effects within a radius extending over several thousand meters, as discussed above, the area impacted would still be relatively small compared to the total marine area designated as critical habitat within the green sturgeon's migratory corridor. While explosives may affect the safety and timeliness of green sturgeon coastal migrations, the magnitude of such effects on the migratory corridor PBF would not likely compromise individual survival or the overall viability of the species. As discussed above, seasonal mitigation proposed by the Navy, that limits the number of explosives that could be used in months when green sturgeon densities are expected to be highest, would likely further reduce the impact of explosives on the migratory corridor PBF. Therefore, the effects of explosives used in mine countermeasure and neutralization testing on the migratory corridor PBF of Southern DPS green sturgeon critical habitat are likely insignificant.

Underwater explosions may reduce available prey items by either directly killing or injuring green sturgeon benthic prey or by impacting the structure and complexity of the benthic

environment where such prey occur. If green sturgeon prey items are killed or injured within critical habitat, it is likely that only a low number of individuals and therefore a small portion of benthic prey species populations would be affected. Although some prey items could be exposed within the described mortality ranges during an explosive activity, other prey items would likely be available to green sturgeon in the immediate (unaffected) area surrounding the activity. Given the small number of events and short duration of the activity, the immediate impact of explosives on prey availability within critical habitat would likely be short-term. In addition, the Navy has proposed mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months of October through June (see Section 3.5.1.9 for details). Since this time frame includes months when peak densities of Southern DPS green sturgeon would be expected in the action area, this mitigation measure would likely result in reduced impacts to the food resources PBF during important migratory times of year.

A potentially longer term impact from this stressor could occur from the effects of explosives on the benthic environment that green sturgeon prey inhabit. Huff et al. (2011) found that green sturgeon, on average, spent a longer duration in areas with high seafloor complexity, especially where a greater proportion of the substrate consists of boulders. The authors suggest that green sturgeon may seek out complex seafloor habitat because it coincides with the distribution of benthic prey taxa or provides refuge from predators. Explosive detonations in green sturgeon critical habitat could result in physical disturbance that reduces seafloor complexity, which attracts green sturgeon and their prey to such habitats. Compared to pelagic prey species which would be more likely to repopulate an area shortly after an explosive activity is completed, benthic prey may be less likely to recolonize seafloor habitat areas that have been physically damaged by the blast of explosive detonations. Thus, physical disturbances that impact the substrate's ability to function as a habitat could have longer lasting effects on benthic prey availability. Disturbances can result from several sources including the physical impact of the expended material contacting the substrate, the covering of the substrate or biogenic habitats by the expended material, or the alteration of the substrate from one type to another (e.g., converting soft bottom substrate into hard bottom resulting from solid expended materials overlying soft substrates) (Navy 2020e). In areas where suspended sediment settles on the bottom, some smothering can occur which disrupts the benthic communities. The speed of recovery by benthic communities is affected by several factors, including the intensity of the disturbance, with greater disturbance increasing the time to recovery. Additionally, the ability of a disturbed site to recolonize is affected by whether or not adjacent benthic communities are nearby that can re-seed the affected area.

Explosive mines (bin E7) associated with this activity would be moored in the water column at least 75 feet from the seafloor, and no explosive mines would be detonated on the ocean bottom. The Navy has indicated that, given the distance of moored mines from the bottom, a crater footprint on the seafloor would not occur as a result of explosive mine detonations (Navy 2020a) Aside from cratering, the force of a large explosive mine detonated in the water column could

result in other impacts on features of benthic habitat that may be utilized by green sturgeon prey. These include effects on water quality from increased siltation and turbidity and physical disturbance of habitats utilized by benthic invertebrates. However, we have no specific information on the likelihood or duration of such impacts from an E7 explosive mine detonated 75 feet from the seafloor. The locations within the action area where mine countermeasure and neutralization testing could overlap with green sturgeon critical habitat consist of primarily soft bottom (primarily sand) habitat (see Figure 27 above), which is dynamic in nature and would be expected to recover relatively quickly from disturbance (Navy 2020c).

Mine neutralizers (bin E4 explosives) are shaped charges that would be used to penetrate live moored explosive mines, inert moored targets, or bottom-placed (inert only) targets. Unlike typical high explosives, mine neutralizers have a directional force aimed at the target. Shaped charges focus a majority of the energy produced by the high explosive into a narrow beam. This allows all of the energy from a shaped charge to be directed at a specific target rather than into the surrounding environment. The debris field produced by the detonation of a shaped charge will also be limited due to the focused nature of the charge and the close proximity of the target (Navy 2020c).

Given their smaller net explosive weight (5 lbs. versus 60 lbs. for E7 mines), and the fact that the inert target would absorb some of the impact, E4 mine neutralizers detonated on impact with moored targets (i.e., that are neither on nor near the seafloor) would not be expected to disturb the seafloor in any meaningful way. Mine neutralizers detonated on impact with inert targets on or near the seafloor could disturb the seafloor and impact benthic habitat. However, given the proposed net explosive weight and properties of explosions from shaped charges hitting inert targets discussed above, the area of Southern DPS green sturgeon critical habitat that could be impacted by such detonations would be relatively small. In addition, as noted above, the locations within the action area where this activity would likely overlap with critical habitat consist primarily of soft bottom sandy habitat. As described in Section 3.5.2.1 (*Mitigation Areas for Seafloor Resources*), the Navy will avoid potential impacts on live hard bottom habitat (i.e., rocky reef Habitat Area of Particular Concern) and artificial structures (i.e., artificial reefs, shipwrecks), which are areas with high seafloor complexity that green sturgeon inhabit, shelter in, and feed among (Huff et al. 2011). Within a 350 yard (320 meter) radius of live hard bottom, artificial reefs, and shipwrecks, the Navy will not conduct explosive mine countermeasure and neutralization activities, and the Navy will not place mine shapes, anchors, or mooring devices on the seafloor. This mitigation measure would likely further minimize potential impacts from this activity to benthic habitats utilized by Southern DPS green sturgeon prey within critical habitat.

In summary, we find that stressors associated with the proposed action would either have no effect or insignificant effects on the PBFs of Southern DPS green sturgeon designated critical habitat. While explosives may affect the safety and timeliness of green sturgeon coastal migrations, the magnitude of such effects on the migratory corridor PBF would not likely

compromise individual survival or the overall viability of the species. Therefore, the effects of explosives used in mine countermeasure and neutralization testing on the migratory corridor PBF of Southern DPS green sturgeon critical habitat are likely insignificant. Explosive detonations could result in physical disturbance of the seafloor that reduces quality of benthic habitats utilized by green sturgeon and their benthic prey species. The only detonations occurring on (or near) the seafloor would be from shaped charges (E4 neutralizers) hitting inert targets. Given the relatively small net explosive weight of the proposed neutralizers, the unique properties of explosions from shaped charges hitting inert targets, the primarily soft bottom areas where this activity would likely occur, and the proposed mitigation to reduce impacts on areas of high seafloor complexity, we find that the impacts of this activity on benthic habitats within Southern DPS green sturgeon designated critical habitat are likely insignificant. Therefore, the effects of all stressors analyzed on the essential PBFs were found to be either insignificant or discountable, and not likely to reduce the conservation value of Southern DPS green sturgeon designated critical habitat. In conclusion, for the reasons provided above, we determine that the proposed action may affect but is not likely to adversely affect Southern DPS green sturgeon designated critical habitat.

9 CUMULATIVE EFFECTS

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, we searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. We conducted electronic searches of business journals, trade journals, and newspapers using First Search, Google, and other electronic search engines. We did not find any information about non-Federal actions other than what has already been described in the *Environmental Baseline* (Section 7), most of which we expect would continue in the future. An increase in these activities could similarly increase their effects on ESA-listed species and for some, an increase in the future is considered reasonably certain to occur. In particular, threats associated with climate change, coastal development and pollution, marine debris, fisheries bycatch, vessel strike, and anthropogenic ocean noise are likely to continue in the future. For many of the activities and associated threats identified in the Environmental Baseline, and other unforeseen threats, the magnitude of increase and the significance of any anticipated effects remain unknown. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on populations of ESA-listed species. Thus, this opinion assumes effects in the future would be similar to those in the past and, therefore, are reflected in the anticipated trends described in the *Species and Designated Critical that May be Affected* (Section 6) and *Environmental Baseline* (Section 7).

Future tribal, state, and local government actions will likely be in the form of legislation, administrative rules, or policy initiatives and fishing permits. Activities occurring in action area are primarily those conducted under state, tribal or federal government management. These actions may include changes in ocean policy and increases and decreases in the types of activities currently seen in the action area, including changes in the types of fishing activities, resource extraction, and designation of marine protected areas, any of which could impact listed species or their habitat. Government actions are subject to political, legislative and fiscal uncertainties. As a result any analysis of cumulative effects is difficult, particularly when taking into account the geographic scope of the action area, the various authorities involved in the action, and the changing economies of the region. Although state, tribal and local governments have developed plans and initiatives to benefit listed fish, they must be applied and sustained in a comprehensive way before NMFS can consider them "reasonably foreseeable" in its analysis of cumulative effects.

An example of one such initiative is the Southern Resident Killer Whale Task Force, established through an executive order by the governor of Washington State to identify, prioritize, and

support the implementation of a longer term action plan need for Southern Resident killer whale recovery. The Task Force provided recommendations in a final report in November 2018. Although it is likely that several of the recommended actions will occur, it is currently uncertain which ones will be implemented. In response to recommendations of the Task Force, the Washington State Legislature provided approximately $13 million in funding "prioritized to increase prey abundance for southern resident orcas" (Engrossed Substitute House Bill 1109) for the 2019-2021 biennium (July 2019 through June 2021). The planned 2020 production associated with this legislative action is a release of an additional 13.5 million Chinook salmon (approximately 6.4 million from Puget Sound facilities, approximately 5.6 million from Washington coastal facilities, and approximately 1.5 million from Columbia River facilities). A similar level of Chinook salmon production funded by this legislative action is anticipated in the spring of 2021.

Washington State passed House Bill 1579 that addresses habitat protection of shorelines and waterways (Chapter 290, Laws of 2019 (2SHB 1579)), and funding was included for salmon habitat restoration programs and to increase technical assistance and enforcement of state water quality, water quantity, and habitat protection laws. Other actions included measures to increase survival through the hydropower system on the lower Snake and Lower Columbia rivers, passed legislation to decrease impacts of predatory fish on salmon (Chapter 290, Laws of 2019 (2SHB 1579)), passed the federal Endangered Salmon Predation Prevention Act (PL 115-329) to provide state and tribal managers more flexibility to manage sea lion predation on the Columbia River, provided funding to the Washington State Department of Transportation to complete fish barrier corrections, and funding to implement a lower Snake River dams stakeholder engagement process.

10 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the *Effects of the Action* (Section 8) to the *Environmental Baseline* (Section 7) and the *Cumulative Effects* (Section 9) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce appreciably the value of designated critical habitat for the conservation of the species. These assessments are made in full consideration of the *Status of the Species and Critical Habitat* (Section 6).

The following discussions separately summarize the probable risks the proposed action poses to threatened and endangered species and critical habitat that are likely to be exposed. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this opinion. Where stressors were determined to have insignificant or discountable effects to certain or all species earlier in this opinion, those stressors will not cause adverse effects to individuals of those species or cause a population or species level effect and thus cannot contribute to causing jeopardy or destruction or adverse modification of critical habitat

10.1 Blue Whale

The blue whale is endangered as a result of past commercial whaling. In the North Atlantic Ocean, at least 11,000 blue whales were taken from the late 19th to mid-20th centuries. In the North Pacific Ocean, at least 9,500 blue whales were killed between 1910 and 1965. Commercial whaling for blue whales no longer occurs, but blue whales are still affected by other stressors including anthropogenic noise, vessel strikes, entanglement in fishing gear, pollution, harassment due to whale watching, and reduced prey abundance and habitat degradation due to climate change. The minimum population size for Eastern North Pacific Ocean blue whales is 1,050; the more recent abundance estimate is 1,496 whales (Carretta et al. 2020). Current estimates indicate a growth rate of just under three percent per year for the Eastern North Pacific stock. Because populations appear to be increasing in size, the species appears to be somewhat resilient to current threats; however, the species has not recovered to pre-exploitation levels.

Blue whales are only present in the offshore portion of the NWTT action area. Blue whales may be exposed to acoustic stressors associated with training and testing activities throughout the year. Blue whales found in the offshore portion of the NWTT action area are recognized as part of the Eastern North Pacific stock. The acoustic effects analysis predicts that blue whales from the Eastern North Pacific stock may be exposed to sonar and other active acoustic sources associated with training activities in the offshore area that may result in two behavioral reactions annually and 11 behavioral reactions every seven years of the proposed action. Blue whales may also be exposed to sonar and other active acoustic sources associated with testing activities

throughout the year. The acoustic analysis predicts that blue whales of the Eastern North Pacific stock would be exposed to sonar and other active acoustic sources associated with testing activities in the offshore area that would result in three behavioral reactions and four TTS exposures annually, and 14 behavioral reactions and 21 TTS exposures every seven years of the proposed action. Acoustic modeling predicts that blue whales from the Eastern North Pacific stock would be exposed to impulses from explosive sources associated with testing activities in the offshore area that would result in one behavioral reaction annually and three behavioral reactions every seven years of the proposed action. Overall, NWTT training and testing activities would result in an estimated 0.022 behavioral disruptions annually per blue whale from the Eastern North Pacific stock (see Table 134).

The 1998 blue whale recovery plan does not outline downlisting or delisting criteria. The recovery plan does list several stressors potentially affecting the status of blue whales in the North Pacific Ocean that are relevant to NWTT activities including vessel strike, vessel disturbance, and military operations (including sonar). At the time the recovery plan was published, the effects of these stressors on blue whales in the Pacific Ocean were not well documented, their impact on recovery was not understood, and no attempt was made to prioritize the importance of these stressors on recovery. Anthropogenic noise associated with NWTT activities is not expected to impact the fitness of any individuals of this species. No mortality or serious injury of blue whales is expected to occur from NWTT activities.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy would conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA Rule from November 2020 to November 2027, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities and related incidental take specified in the MMPA Rule continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. NWTT stressors and related incidental take specified in the MMPA Rule would not affect the population dynamics, behavioral ecology, and social dynamics of individual blue whales in ways or to a degree that would reduce their fitness. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic or explosive stressors. We do not anticipate those behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect acoustic or explosive stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales compose (that is, we would not expect

reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

10.2 Fin Whale

Fin whale populations have decreased as a result of commercial whaling but are now protected under the IWC. Current threats to fin whales include entanglement in fishing gear, vessel strikes, pollution, and climate change. The best current abundance estimate for fin whales in California, Oregon, and Washington waters out to 300 nautical miles is 9,029 (CV=0.12) (Carretta et al. 2020; Nadeem et al. 2016); the minimum population estimate is 8,127 individuals (Carretta et al. 2020). Current estimates indicate approximately 10,000 fin whales in U.S. Pacific Ocean waters, with an annual growth rate of 4.8 percent in the Northeast Pacific stock, and a stable population abundance in the California/Oregon/Washington stock (Nadeem et al. 2016).

Fin whales could be present in all portions of the NWTT action area and may be exposed to acoustic sources associated with training and testing activities throughout the year. Fin whales found in the offshore portion of the action area and the Inland Waters portion that includes Puget Sound and the Strait of Juan de Fuca are recognized as part of the California, Oregon, and Washington stock. Fin whales found in Southeast Alaska (Western Behm Canal) are recognized as part of the Northeast Pacific stock.

The acoustic effects analysis predicts that fin whales of the California, Oregon, Washington stock and Northeast Pacific stock would be exposed to sonar and other active acoustic sources associated with training activities in the offshore area and Western Behm Canal that would result in 13 TTS and 41 behavioral reactions annually, and 92 TTS and 285 behavioral reactions every seven years of the proposed action. Fin whales would also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic effects analysis predicts that fin whales of the California, Oregon, Washington stock and Northeast Pacific stock would be exposed to sonar and other active acoustic sources associated with testing activities in the offshore area and Western Behm Canal that would result in 30 TTS and 45 behavioral reactions annually, and 145 TTS and 206 behavioral reactions every seven years of the proposed action. Acoustic modeling predicts that fin whales would be exposed to impulses from explosive sources associated with testing activities in the offshore area would result in two TTS and six behavioral reactions annually, and 9 TTS and 39 behavioral reactions every seven years of the proposed action. Overall, NWTT training and testing activities would result in an estimated 0.174 behavioral disruptions annually per fin whale from the California/Oregon/Washington stock (see Table 134). Based on our vessel strike analysis, we anticipate that no more than two fin whales would be struck by Navy vessels over the seven-year duration of the MMPA rule.

The 2010 fin whale recovery plan defines three recovery populations by ocean basin (the North Atlantic, North Pacific, and Southern Hemisphere) and sets criteria for the downlisting and

delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with fisheries, climate change, direct harvest, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. Anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. Two fin whale vessel strikes over seven years by Navy vessels during NWTT activities, based on our vessel strike analysis, would represent about 0.022 percent of the California, Oregon, and Washington stock. There is currently no reliable population estimate for the Northeast Pacific stock of fin whales, but the conservative estimate of two mortalities from this stock would likely represent a very small percentage of this stock as well. Downlisting criteria for fin whales includes the maintenance of at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on fin whale recovery populations.

NWTT acoustic and explosive stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual fin whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species from acoustic and explosive stressors as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated. Due to a lack of fitness consequences to individuals and the populations they represent, we also do not anticipate any reductions in survival rate or trajectory of recovery of fin whales as currently proposed from acoustic and explosive stressors.

Regarding vessel strikes, we anticipate an extremely small number of individual fin whales (up to two every seven years), relative to the population sizes of their aforementioned stocks, would be killed or seriously injured as a result of the proposed action. In our judgement, the removal of such a small proportion of the population would likely have a very minor negative impact on the population trend for these stocks. The loss (or serious injury) of these individuals is not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. For this reason, the effects of the proposed action from vessel strikes are not expected to appreciably reduce the likelihood of survival and recovery of fin whales.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects from stressors caused by training and testing activities the Navy would conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA Rule from November 2020 to November 2027, or cumulatively for the

reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities and related incidental take specified in the MMPA Rule continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species.

10.3 Humpback Whale – Central America and Mexico DPSs

Humpback whales have been threatened historically by whaling, while the most significant anthropogenic threats currently include entanglements and vessel strikes. The global, pre-exploitation estimate for humpback whales is 1,000,000 (Roman and Palumbi 2003). Based on surveys from 2004 to 2006, the Central America DPS is estimated to have just below 800 individuals, while the Mexico DPS is estimated to have just below 3,000 individuals (Wade 2017). However, sightings of humpbacks off the U.S. West Coast have been increasing in more recent years, and these DPS numbers are likely underestimates. Population growth rates are currently unavailable for the Central America DPS and Mexico DPS of humpback whales (Calambokidis 2017).

Humpback whales could be in all portions of the NWTT action area and may be exposed to sonar and other active acoustic sources associated with training activities throughout the year. Humpback whales found in the offshore portions of the action area and the Inland Waters of Puget Sound area are recognized as part of the California/Oregon/Washington stock. The humpback whales in the action area potentially belong to one of two ESA-listed DPSs: the threatened Mexico DPS or the endangered Central America DPS. Both of these DPSs may feed seasonally in the action area.

The acoustic effects analysis predicts that humpback whales of the Central America DPS and/or the Mexico DPS would be exposed to sonar and other active acoustic sources associated with training activities throughout the action area that would result in one TTS and three behavioral reactions annually, and 7 TTS and 21 behavioral reactions every seven years of the proposed action. The acoustic effects analysis also predicts that humpback whales of the Central America DPS and/or the Mexico DPS would be exposed to sonar and other active acoustic sources associated with testing activities throughout the action area that would result in a total of 51 TTS and 36 behavioral reactions annually, and a total of 257 TTS and 189 behavioral reactions every seven years of the proposed action. Acoustic modeling predicts that humpback whales from the Mexico DPS would be exposed to impulses from explosive sources associated with testing activities in the offshore area that would result in one TTS and one behavioral reaction annually, and five TTS and five behavioral reactions every seven years of the proposed action. Overall, NWTT training and testing activities would result in an estimated 0.123 behavioral disruptions

annually per humpback whale from the Central America and Mexico DPSs (see Table 134). Based on our vessel strike analysis, we anticipate that no more than two humpback whales, from the Central America DPS and Mexico DPS combined, would be struck by Navy vessels over the seven-year duration of the MMPA rule.

The 1991 humpback whale recovery plan does not outline specific downlisting and delisting criteria. The recovery plan does list several threats known or suspected of impacting humpback whale recovery including subsistence hunting, commercial fishing stressors, habitat degradation, loss of prey species, ship collision, and acoustic disturbance. Of these, ship collision and acoustic disturbance are relevant to NWTT activities. As described previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species.

NWTT acoustic and explosive stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual humpback whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species from acoustic and explosive stressors as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated. Due to a lack of fitness consequences to individuals and the populations they represent, we also do not anticipate any reductions in survival rate or trajectory of recovery of the endangered Central America DPS and threatened Mexico DPS of humpback whales as currently proposed from acoustic and explosive stressors.

Regarding vessel strikes, we anticipate at most around 0.26 percent of the humpback whale Central America DPS or 0.071 percent of the humpback whale Mexico DPS, may be killed or seriously injured over a seven-year period as a result of the proposed action. In our judgement, the removal of such a small proportion of the population would likely have a very minor negative impact on the population trend for these DPSs. The loss (or serious injury) of these two individuals over seven years from the Central America DPS (currently estimated at just under 800 whales) or the Mexico DPS (currently estimated at just under 3,000 whales) is not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. For this reason, the effects of the proposed action from vessel strikes are not expected to appreciably reduce the likelihood of survival and recovery of Central America DPS and Mexico DPS humpback whales.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects,* effects resulting from stressors caused by training and testing activities the Navy would conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA Rule from November 2020 to November 2027, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed*

Resources or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Central America DPS and Mexico DPS humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future and related incidental take specified in the MMPA Rule would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Central America DPS and Mexico DPS humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species.

10.3.1 Central America and Mexico DPSs Proposed Critical Habitat

On October 9, 2019, NMFS proposed to designate critical habitat for Central America and Mexico DPS humpback whales (84 FR 54354). Critical habitat for the Central America DPS would include 48,459 square nautical miles of marine habitat off the coasts of Washington, Oregon, and California. Critical habitat for the Mexico DPS would include 175,812 square nautical miles of marine habitat off the coasts of Alaska, Washington, Oregon, and California. About 1,570 square nautical miles of marine habitat are being proposed for exclusion from the critical habitat designation as a result of potential national security impacts. These areas include the Quinault Range Site and Buffer and SEAFAC.

These critical habitat designations include the physical and biological feature of prey species, defined as "primarily euphausiids and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth" (84 FR 54354).

We evaluated the effects of the proposed action on the prey essential feature of humpback whale proposed coastal critical habitat for the Central America and Mexico DPSs. We found that the effects of the following potential stressors on the quantity, quality, and availability of humpback whale prey would not likely reduce the conservation value of proposed coastal critical habitat: sonar and other transducers, in-water devices, and weapons firing and impact noise.

The use of Navy explosives within (or adjacent to) areas proposed as coastal critical habitat would likely result in the mortality and injury of schooling fishes preyed upon by humpback whales. With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore. Mine countermeasure and neutralization testing involving the use of explosives would occur in waters three nautical miles or greater from shore within the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nautical miles or greater from shore elsewhere in the NWTT offshore area. Some proportion of the explosions associated with this activity could occur either within the excluded Quinault Range Site or beyond the 200 meters depth contour.

The anticipated impacts on any euphausiids and schooling fishes, in terms of either total numbers of animals affected or percent of the population affected, are relatively minor. The number of euphausiids or fishes killed is anticipated to represent a small fraction of the total number of prey items needed per year by an individual humpback whale to maintain its energy requirements. Since the effects of reduced prey availability would likely be spread out over the DPSs as a whole (or some proportion of the DPSs), the impacts of prey mortalities as a result of NWTT offshore explosives on the nutritional needs of the Central America and Mexico DPSs would likely be relatively minor. Based on the reproductive strategies of the prey items affected, the effects on the current and future prey availability and reproductive potential of these prey items would also likely be relatively small. The anticipated extent of injury and mortality of humpback whale prey items resulting from NWTT explosives is extremely low relative to the number of prey items found throughout the action area and would not be detectable above baseline conditions.

In summary, we find that the Navy's use of explosives in the offshore portion of the NWTT action area would have a minor impact on the availability, quantity, or quality of humpback whale prey in proposed critical habitat. Thus, the effects of explosives on the prey essential PBF are not expected to appreciably reduce the conservation value of humpback whale proposed critical habitat for the Central America and Mexico DPSs. Therefore, based on our critical habitat effects analysis presented in Section 8.3.1.1, and the best available information regarding the risk of those effects, we do not anticipate the effects of the proposed action would appreciably diminish the value of Central America DPS and Mexico DPS humpback whale proposed critical habitat as a whole for the conservation of this species.

10.4 Southern Resident Killer Whale DPS

The Southern Resident killer whale DPS was listed as endangered under the ESA on November 18, 2005. The cumulative and synergistic effects of multiple threats have resulted in the continued decline of the Southern Resident killer whale population. Between 1967 and 1973 about 30 percent of the population was captured live for displays in oceanaria. The primary ongoing threats to the recovery of this population include quantity and quality of prey, toxic chemicals that accumulate in top predators, and disturbance from sound and vessels. Based on their population size and life history traits (i.e., slow-growing mammals that give birth to single calves with several years between births), we assume that Southern Resident killer whales would have elevated extinction probabilities due to a combination of exogenous anthropogenic threats (as discussed above in the Section 6.2.4 *Status of the Species* and Section 7 *Environmental Baseline*), natural phenomena (including vulnerability to disease), and endogenous threats resulting from their small population size.

A growing body of evidence documents how Southern Resident killer whales are affected by prey limitations, particularly Chinook salmon. Salmon populations in the Pacific Northwest have declined due to a combination of factors including land alteration associated with agriculture and

timber harvest practices, the construction of dams, urbanization, fishery harvest practices, hatchery operations, and increased predation from a growing population of pinnipeds. When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources, and as a chronic condition can lead to reduced body size and condition of individuals and lower birth and survival rates in a population. Indicators of nutritional stress include the poor condition individual Southern Resident killer whales are occasionally found in, and variable levels of the thyroid hormone triiodothyronine (Wasser et al. 2017). In addition, Southern Resident killer whale fecundity, death rates and rates of population increase have shown statistical correlations with some indices of Chinook salmon abundance (Hilborn et al. 2012).

Vessel traffic exposes Southern Resident killer whales to several threats that have consequences for the species' likelihood of survival and recovery. Three vessels strikes, two lethal and one sublethal, of Southern Resident killer whales have been documented in the past 15 years. In addition to strikes, the number and proximity of vessels, particularly whale-watch vessels in the inland areas occupied by Southern Resident killer whales, represents a source of chronic disturbance and stress for this population. With the disruption of feeding behavior that has been observed, it is estimated that the presence of vessels could result in an 18 percent decrease in energy intake, a consequence that could have a significant negative effect on an already prey-limited species (Lusseau et al. 2009; Williams et al. 2006a). Foraging behavior may also be impacted by sound that interferes with the whales' echolocation from vessels or other sounds sources. In addition to the disturbance associated with the presence of vessels, vessel traffic affects the acoustic landscape which may affect Southern Resident killer whale communication and social ecology. Vessels in the path of the whales can interfere with important social behaviors such as prey sharing (Ford and Ellis 2006) or nursing (Kriete 2007).

Exposure to contaminants may also harm Southern Resident killer whales. Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales are capable of accumulating high concentrations of contaminants. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants has been documented in Southern Resident killer whales (Krahn et al. 2007; Ross et al. 2000). Although the consequences of these pollutants on the fitness of individual killer whales and the population as a whole remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007).

In the mid- to late-1800s, the Southern Resident killer whale DPS was estimated to have numbered around 200 individuals. For the period between 1974 and the mid-1990s, when the population increased from 76 to 93 animals, the population growth rate was 1.8 percent. A delisting criterion for the Southern Resident killer whale DPS is an average growth rate of 2.3

percent for 28 years (NMFS 2008e). More recent data indicate the population is now in decline (Carretta 2019). The current population estimate of 74, including two new born calves, represents a decline from the recent past, when in 2012 there were 85 whales. As compared to stable or growing populations, the DPS reflects lower fecundity and has demonstrated little to no growth in recent decades (NMFS 2016k).

Given the low current population size, Southern Resident killer whales likely have a higher probability of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox 2007), including stochastic sex determination (Lande et al. 2003), and the effects of phenomena interacting with environmental variability. The very small estimated effective population size (about 26 individuals), the absence of gene flow from other populations, and documented breeding within pods may elevate the risk from inbreeding and other issues associated with genetic deterioration (Ford et al. 2018). These phenomena would likely amplify the potential consequences of anthropogenic stressors on this species.

The Navy's quantitative acoustic effects analysis estimates 49 exposures resulting in behavioral harassment and 2 exposures resulting in TTS of Southern Resident killer whales per year, and 232 behavioral reactions and 11 TTS exposures over seven years as a result of the proposed action (see Section 8.2.1.1 for details). No exposures resulting in PTS of Southern Resident killer whales were predicted. Overall, NWTT training and testing activities may result in an estimated 0.70 behavioral disruptions per Southern Resident killer whale annually (see Table 134). Over 90 percent of the predicted impacts on Southern Resident killer whales from sonar and other transducers are from testing activities in the offshore portion of the action area. The use of the Inland Waters portion of the NWTT action area by Southern Resident killer whales has declined in recent years as they shift their range in response to reduced prey availability in Puget Sound (Olson and Osborne 2017; Olson et al. 2018; Shields et al. 2018). If this trend continues (i.e., increased use of the offshore portion), this could result in more incidents of behavioral harassment and TTS for Southern Resident killer whales than predicted based on current offshore density models.

Southall et al. (2016) suggested that even minor, sub-lethal behavioral changes may still have significant energetic and physiological consequences given sustained or repeated exposure. The consequences of exposure to the anticipated acoustics effects would be more significant for whales that are already in poor condition, as such animals would be less likely to compensate for additional energy expenditures or lost foraging or reproductive opportunities. Southern Resident killer whale individuals are occasionally found in poor condition, which may indicate nutritional stress. However, sustained or repeated disturbance is unlikely for any individual Southern Resident killer whale given the relatively low estimated number of exposures predicted. With a current population size of 74 in the wild (including two calves born in September 2020), each individual Southern Resident killer whale would be exposed to Navy sonar at levels resulting in behavioral harassment, on average, less than once per year. While some individuals could experience more than one behavioral disruption per year, exposures would likely be infrequent

and short-term. Based on the available literature that indicates such infrequent exposures are unlikely to impact an individual's overall energy budget (Farmer et al. 2018; Harris et al. 2017b; King et al. 2015; NAS 2017; New et al. 2014; Southall et al. 2007a; Villegas-Amtmann et al. 2015), we do not expect this level of exposure to impact the fitness of exposed Southern Resident killer whales, even individuals that are already in poor condition.

Exposure resulting in TTS would be rare and, given the relatively narrow frequency range associated with most individual Navy sonar sources, we anticipate that any instances of TTS would likely be of minimum severity and short duration (Finneran 2015a). Given the infrequent number of instances (i.e., two per year across 74 individuals) and the brief amount of time individuals would be affected per instance, TTS is unlikely to significantly impair the ability of Southern Resident killer whales to communicate, forage, or breed, and is not expected to have long-term fitness consequences for the individuals affected. Additionally, we do not anticipate long duration or repeat exposures within a short period of time due to the species' ability to move quickly throughout its range and that long duration (i.e., more than one day) Navy activities also occur over large geographic areas (i.e., both the animal and the activity are moving within the action area, most likely not in the same direction). This decreases the likelihood that Southern Resident killer whales and Navy activities would co-occur for extended periods of time or repetitively over the duration of an activity. Although there is an increased chance that TTS resulting from explosives would affect frequencies utilized by animals for acoustic cues (as compared to TTS from sonar), the Navy's quantitative model predicts zero instances of TTS or behavioral harassment of Southern Resident killer whales from explosives. Therefore, as with behavioral harassment, we do not anticipate that the few anticipated instances of TTS from Navy activities involving sonar would result in long-term fitness consequences to individual Southern Resident killer whales in the action area. The Navy's continued implementation of procedural mitigation to avoid or reduce potential impacts on marine mammals from active sonar further decreases the likelihood of long-term fitness consequences resulting from these activities (see Section 3.5.1 *Procedural Mitigation* and Section 3.5.2.2 *Marine Species Mitigation Areas* for details).

The injury and mortality of salmonids that would result from NWTT stressors could have indirect effects on Southern Resident killer whales by reducing prey availability. A reduction in the availability of their prey may cause killer whales to forage for longer periods, travel to alternate locations, or abandon foraging efforts. Limitations in their prey availability is considered one of the primary threats affecting the survival and recovery of Southern Resident killer whales. Our analysis of the effects of the proposed action on Southern Resident killer whales via impacts to their prey focused on Chinook salmon, their primary prey throughout their range (Ford et al. 2010b; Ford et al. 2016; Hanson In prep; Hanson et al. 2010; Hilborn et al. 2012; NMFS 2019e), as well as Puget Sound/Georgia Strait coho and Puget Sound/Georgia Strait chum salmon, which may be important as substitute species in inland waters during certain months when the availability of Chinook salmon is reduced (Ford et al. 2016).

Other than explosives, we found that none of the stressors associated with the proposed action were likely to adversely affect salmonids (see Section 8.1.3 for details). With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore, and, therefore, outside of the areas where higher densities of many Chinook salmon, coho or chum salmon ESUs would typically be found. The use of Navy explosives would likely result in the mortality and injury of Chinook salmon, coho salmon, and chum salmon from populations preyed upon by Southern Resident killer whales. The mortality of adult salmon could have immediate impacts on the availability of prey for Southern Resident killer whales, as well as the reproductive potential of the salmon stock and population.

Based on our quantitative analysis, the estimated annual number of adult Chinook salmon (all ESUs) and Puget Sound/Georgia Strait coho and Puget Sound/Georgia Strait chum salmon killed during a maximum year of NWTT explosive activity levels represents an extremely small fraction of the total number of salmon needed per year by an individual Southern Resident killer whale to maintain their energy requirements. Since the effects of reduced prey availability would likely be spread out over the population as a whole (or some proportion of the population), the impacts of adult Chinook salmon, chum salmon, and coho salmon mortalities as a result of NWTT explosives on the nutritional needs of the Southern Resident killer whale DPS are anticipated to be minor. While the loss of an individual fish represents the loss of future reproductive potential, given the small anticipated mortality rates (i.e., less than 0.02 percent of each fish population), it is unlikely that salmon mortality from NWTT explosives would have an appreciable impact on the future reproductive potential of the salmon populations preyed upon by Southern Resident killer whales.

From our quantitative analysis, we expect a significantly larger number of salmon would be injured (as compared to killed) as a result of NWTT explosives. Several of the highest priority Chinook salmon ESUs for Southern Resident killer whale conservation, also had some of the highest estimated injury rates as a proportion of the population size among the ESUs analyzed. These include the Snake River fall run ESU, Snake River spring/summer ESU, Upper Columbia River fall run ESU, and Upper Columbia River spring run ESU (all with estimated adult injury rates of between 1.0 and 1.2 percent). For the Puget Sound Chinook salmon ESU, which is impacted by both offshore and inland explosives, we estimate a juvenile injury rates of nearly 1.2 percent for natural origin fish during a maximum year of NWTT explosive activity, although adult and hatchery injury rates were somewhat lower (Table 137). We also estimate an adult injury rate of 1.1 percent for the Puget Sound/Georgia Strait coho salmon ESU, which, as noted above, may be particularly important to the Southern Resident killer whale diet in the late summer during seasonal downward shifts in Chinook salmon abundance (Ford et al. 2016). As discussed previously, our fish effects analysis is based on a number of conservative assumptions that likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

We anticipate that while some injuries to Southern Resident killer whale prey species from explosives would be relatively minor, others may impact important life functions including predator avoidance, foraging, and migrations. Although we cannot quantify based on the available information, we expect that some proportion of salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected. Given the estimated number and proportional rate of injuries anticipated, particularly of high priority Chinook salmon ESUs, salmonid injuries resulting from the proposed action would likely have impacts on Southern Resident killer whale prey availability, as well as impacts on the future reproductive potential of important salmon prey populations. Therefore, based on the best available information, we conclude that Southern Resident killer whales would likely be adversely affected due to the anticipated reduction in the availability of their salmonid prey as a result of the proposed action.

While the use of Navy explosives would likely result in the mortality and injury of salmon populations preyed upon by Southern Resident killer whales, it should be noted that the estimated impacts presented above are based on a maximum annual activity level of offshore explosive use; in an average year the anticipated impacts would be somewhat lower. For the mine countermeasure neutralization activity, the annual average number of explosives used would be less than half that used in a maximum year. We also note that our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa), which likely result in conservatively high estimates of injury. The Navy would also continue to implement procedural mitigation measures to avoid or reduce potential impacts on marine mammals from activities involving explosives. The Navy has proposed mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months of October through June (see Section 3.5.1.9 for details). Since this timeframe includes months when peak densities of many Chinook salmon ESUs would be expected in the action area, this mitigation measure would likely result in reduced impacts to several of the Chinook salmon ESUs preyed upon by Southern Resident killer whales. Southern Residents are also more likely to occur in the offshore portion of the action area from October through June as compared to July through September, when they are more often seen in the inland waters. Mitigation measures proposed by the Navy would likely reduce the impacts of inland explosives on salmonid populations preyed upon by Southern Resident killer whales in the inland portion of their range. As a mitigation measure to minimize impacts to salmon in Hood Canal, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (> 0.5-2.5 pounds), and only conduct training with one ounce or less (E0) charges during August, September, and October. The Navy will also continue to implement procedural mitigation to avoid or reduce potential impacts on marine mammals from explosive mine countermeasure and neutralization activities. The use of Lookouts should reduce the

likelihood that Southern Resident killer whales would be actively feeding on salmon in areas where underwater explosives are being used.

As discussed in Section 8.1.1, all other potential stressors resulting from the proposed action would likely result in either insignificant or discountable effects on Southern Resident killer whales. These include most of the acoustic stressors (except sonar), and all of the energy stressors, physical disturbance stressors, entanglement stressors, ingestion stressors, and secondary stressors on Southern Resident killer whale habitat or prey species. While we anticipate some vessel strikes of large whale species (i.e., fin, humpback, and sperm), Southern Resident killer whales are not expected to be struck as a result of the proposed action. In general, the Navy has no records of any delphinids or small whales being struck by a vessel as a result of Navy activities in any OPAREA. Although Southern Resident killer whales have been struck in the past (three recorded incidents since 2005), these strikes did not involve Navy vessels, which make up an extremely small proportion of vessel traffic in the action area. Given this rate of past vessel strikes, the small number of Navy vessels in the action area compared to other vessels, and the fact that a Navy vessel strike involving a Southern Resident killer whale has not been recorded in the past, it is extremely unlikely that future strikes of Southern Resident killer whales would involve Navy vessels. In addition, The Navy will implement procedural mitigation to avoid or reduce the potential for vessel strikes of marine mammals (see Table 36 for details). Therefore, the likelihood of a Southern Resident killer whale vessel strike as a result of the proposed action is discountable.

The Southern Resident killer whale DPS would likely be adversely affected due to the anticipated reduction in the availability of their salmonid prey as a result of the proposed action. The most likely adverse effects would be increased time and energy expenditure searching for prey. It is possible that there is a measurable effect to the whales' behavior in terms of additional foraging effort due to the salmonid prey reductions anticipated from the proposed action. However, based on our effects analysis and considering the proposed mitigation measures, we do not expect these changes to persist or be so large that they result in more than a minor change to the overall health of any individual whale, or that they change the status of the population. Thus, even assuming a measurable effect, this would not rise to the level of an appreciable reduction in the likelihood of survival of any individual whale or the population as a whole.

In summary, we do not anticipate any effects from the proposed action would result in the mortality or serious injury of individual Southern Resident killer whales. Navy sonar as part of the proposed action would likely result in adverse effects on the Southern Resident killer whale DPS including behavioral harassment and TTS. However, given the small number of annual exposures estimated relative to the population size and short-term, temporary nature of the effects, we do not anticipate long-term fitness consequences for exposed individuals. Southern Resident killer whales would likely be adversely affected by the anticipated reduction in the availability of their salmonid prey as a result of NWTT explosives, resulting in more time and energy expended searching for prey. However, the magnitude of such impacts on individual

killer whales is expected to be relatively minor with no long-term fitness consequences anticipated. This conclusion is based on the relatively small number of estimated salmon mortalities (i.e., most impacts are in the form of salmon injuries), the relatively small proportion of each salmon population that would be injured, and the mitigation measures in place that would likely further reduce the estimated impacts from our effects analysis. An action that is not likely to reduce the long-term fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the Status of Listed Resources or Environmental Baseline), would not be expected to appreciably reduce the likelihood of the survival of the Southern Resident killer whale DPS in the wild by reducing the reproduction, numbers, or distribution of the DPS. We also conclude that effects from ongoing Navy training and testing activities and related incidental take specified in the MMPA Rule continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Southern Resident killer whale DPS in the wild by reducing the reproduction, numbers, or distribution of this DPS. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Southern Resident killer whale DPS.

10.4.1 Inland Designated Critical Habitat

Critical habitat for the Southern Resident killer whale DPS in inland waters was finalized in 2006 (71 FR 69054). The essential physical and biological features (formerly referred to as primary constituent elements or PCEs) of Southern Resident killer whale designated critical habitat in inland waters are: 1) water quality to support growth and development; 2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and 3) inter-area passage conditions to allow for migration, resting, and foraging. As discussed in the *Status of Species* and the *Environmental Baseline* sections of this opinion (Sections 6.2.4 and 7), the conservation value of these PBFs has already been diminished by numerous anthropogenic stressors, many of which are ongoing.

Degraded water quality has had large-scale effects on the Puget Sound – Georgia Basin marine ecosystems. Three main contaminants of concern are PCBs, PBDEs, and DDTs. These and several other contaminants of emerging concern pose threats to ESA-listed species, including Southern Resident killer whales and their primary prey, salmonids (Tian et al. 2019). Pollutants found in Puget Sound Chinook salmon have found their way into the food chain of the Sound.

The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants has been documented in Southern Resident killer whales (Krahn et al. 2007; Ross et al. 2000). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007).

Southern Resident killer whales are adversely affected by limitations of their primary prey, Chinook salmon (Matkin et al. 2017). Salmon have declined because of land alteration throughout the Pacific Northwest associated with agriculture, timber harvest practices, the construction of dams, urbanization, fishery harvest practices, and hatchery operations. These activities have negatively impacted the quantity, quality, and availability of prey to support Southern Resident killer whale individual growth, reproduction and development, as well as overall population growth (Chasco et al. 2017b; Matkin et al. 2017; NMFS 2019e; Wasser et al. 2017).

The large number and proximity of vessels, particularly whale-watch vessels, in the inland areas occupied by Southern Resident killer whales, can interfere with movements of the whales and impact the passage PBF. Vessel strikes are rare, but do occur and can result in serious injury or mortality. Vessel presence can cause killer whales to swim further and change direction more often as avoidance behaviors, which potentially increases energy expenditure and impacts foraging behavior. With the disruption of feeding behavior that has been observed, it is estimated that the presence of vessels could result in an 18 percent decrease in energy intake, a consequence that could have a significant negative effect on an already prey-limited species (Lusseau et al. 2009; Williams et al. 2006a).

As part of our effects analysis above, we evaluated the effects of the proposed action on the essential PBFs of Southern Resident killer whale inland critical habitat. We summarize our findings here. For details refer to Section 8.3.1.2.

Many of the stressors and resulting effects of the proposed NWTT activities would not overlap spatially with areas designated as Southern Resident killer whale inland critical habitat, either because they only occur in the offshore portion or only in inland areas that have been excluded from critical habitat designation for national security reasons. These include: weapons firing and impact noise (except small caliber gunnery using blanks); laser testing; medium and large decelerators and parachutes; chemical and biological simulants; and expended chaff and flares. Several other activities, which may overlap spatially with inland critical habitat, would have no pathway through which the resulting stressors could affect any of the essential features of critical habitat (i.e., water quality, prey, or passage conditions). Stressors that we found would have no effect on the three inland PBFs (i.e., water quality, prey, or passage conditions) include electromagnetic devices, expended wires and cables, small decelerators and parachutes, and

ingestion stressors including fragments from non-explosive practice munitions, explosives, and targets.

While seafloor devices could obstruct killer whale movements, considering the size of these devices, the mobility of killer whales to avoid or swim around them, and the temporary nature of deployment, any effects associated with obstruction from seafloor devices would likely be insignificant.

The effects of metals and chemicals introduced into seawater and sediments as a result of the proposed action were found to be discountable. Similarly, none of the essential features of Southern Resident killer whale inland critical habitat (i.e., water quality, prey, or inter-area passage) would likely be adversely affected by the use of biodegradable polymers in inland waters.

We evaluated the effects of acoustic stressors resulting from the proposed action on Southern Resident killer whale prey, which includes ESA-listed and non-listed salmonids. Most sounds sources proposed for use during training and testing activities overlapping or adjacent to critical habitat in the action area would not fall within the frequency range of salmonid hearing, thereby presenting no plausible route of effect on salmonids (Navy 2020e). We found that any effects of Navy sonar and other transducers, vessels, and in-water devices, on the quantity, quality, and availability of Southern Resident killer whales prey would not likely reduce the conservation value of Southern Resident killer whale inland critical habitat.

We evaluated the effects of acoustic stressors on the inter-area passage essential feature of Southern Resident killer whales inland critical habitat (i.e., inter-area passage conditions to allow for migration, resting, and foraging). Adverse habitat-related effects from acoustic stressors are more likely to result from chronic or long-lasting noise sources that degrade the value of habitat by interfering with the sound-reliant animal's ability to gain benefits from that habitat (i.e., altering the conservation value of the habitat) (NMFS 2019e). None of the sound sources resulting from the proposed action that may overlap with Southern Resident killer whale critical habitat are considered chronic or long-lasting. Instead, exposure to Navy vessel noise, aircraft noise, weapons noise, explosives and sonar as a result of the proposed action would be transient, brief and temporary, with no lasting or appreciably impact on an individual killer whale's ability to migrate, feed or rest. Proposed mitigation measures, as described in Section 3.5, would further reduce any potential effects of acoustic stressors on the Southern Resident killer whale's soundscape. As such, any effects of Navy acoustic stressors on inter-area passage conditions would not likely reduce the conservation value of Southern Resident killer whale inland critical habitat.

NWTT activities involving the use of explosives in inland waters would only occur within two Navy EOD Training Range locations (Hood Canal and Crescent Harbor), neither of which overlaps spatially with designated inland critical habitat for Southern Resident killer whales. Hood Canal was not included in the inland critical habitat designation due to the lack of

confirmed sightings of Southern Resident killer whales within this area. The EOD training range within Hood Canal is a sufficient distance from the entrance to Hood Canal such that any effects from the explosives proposed for use at this range (i.e., bin E0 and E3) would have no impact on any of the essential PBFs in areas that are included in the critical habitat designation (Navy 2020e).

The Crescent Harbor EOD Training Range was excluded from critical habitat designation because the benefits of exclusion for national security were found to outweigh the benefits of inclusion. However, sound from explosives during Navy mine neutralization training events at Crescent Harbor would likely overlap Southern Resident killer whale DPS designated critical habitat in the inland waters, as this training range is located outside of, but adjacent to, the Puget Sound portion of critical habitat (Navy 2020e). Thus, adverse effects on fish species that Southern Resident killer whales prey upon resulting from underwater detonations in Crescent Harbor could affect the prey species essential PBF of inland critical habitat (i.e., prey species of sufficient quantity, quality, and availability). The Navy has indicated that the shortest distance between designated inland critical habitat and the nearest detonation point within the EOD range site is well beyond the estimated maximum range to mortality for the largest explosives proposed in the Crescent Harbor EOD site. Therefore, we do not anticipate any mortality of salmon prey to occur within Southern Resident killer whale designated inland critical habitat. However, injury of salmon could occur within Southern Resident killer whale designated inland critical habitat since the average range to effects does extend into critical habitat.

Based on our effects analysis, any effects from explosives within the Crescent Harbor EOD Range on adult salmon within Southern Resident killer whale inland critical habitat are anticipated to be extremely minor. While this activity would likely impact a greater number of juvenile salmon inside critical habitat, Southern Resident killer whales typically seek out and feed on larger, adult salmon (Hilborn et al. 2012), which provide a much higher energy content compared to juveniles (O'Neill et al. 2014). Since Southern Resident killer whales are likely targeting adult fish, juvenile injuries are anticipated to have much less of an effect on the prey PBF in terms of immediate impacts on Southern Resident killer whales prey availability.

Aside from immediate effects, injuries to juvenile salmon could lead to fitness consequences, which could affect the future availability of prey for Southern Resident killer whales. The estimated number of juvenile Chinook salmon injuries from explosives in Crescent Harbor (both within critical habitat and outside of critical habitat) represent about 0.75 percent of the natural origin juvenile population and 0.05 percent of the hatchery origin juvenile population. Considering the distance between where explosives would likely occur and where critical habitat occurs, it is likely that a large number of these estimated injuries would occur outside of critical habitat. Further, more of the higher severity injuries that are likely to result in fitness consequences would be expected to occur closer to the detonation point (i.e., within the EOD range site, which is excluded from critical habitat). In addition, the Navy has proposed to continue procedural mitigations that may further reduce the estimated impacts of inland

explosives on the prey PBF of Southern Resident killer whale critical habitat. Therefore, based on the information available, we do not expect injuries to juvenile Puget Sound Chinook salmon within inland critical habitat to have a measurable effect on the future availability of prey for Southern Resident killer whales. Similarly, based on our effects analysis, we do not expect injuries to juvenile Puget Sound/Georgia Strait coho or chum within critical habitat to have a measurable effect on the future availability of prey for Southern Resident killer whales.

While explosives may result in injury to Southern Resident killer prey species within designated inland critical habitat, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of this endangered DPS. As such, we find the effects of inland explosives on the prey PBF of Southern Resident killer whale designated inland critical habitat to be insignificant.

Overall, we find that the Navy's use of explosives in the NWTT inland action area would have a relatively small impact on availability, quantity, or quality of Southern Resident killer whales prey in inland critical habitat. The estimated impact may be further reduced by the Navy's proposed continuance of procedural mitigation at Crescent Harbor to avoid adverse effects on juvenile salmon. Thus, the effects of explosives on the prey essential PBF are not expected to reduce the conservation value of Southern Resident killer whale inland critical habitat.

In summary, based on our critical habitat effects analysis presented in Section 8.3.1.2 , and the best available information regarding the risk of those effects, we do not anticipate the effects of the proposed action would appreciably diminish the value of Southern Resident killer whale inland designated critical habitat as a whole for the conservation of this species.

10.4.2 Coastal Proposed Critical Habitat

On September 19, 2019, NMFS proposed to revise the critical habitat designation for the Southern Resident killer whale DPS by expanding it to include areas along the U.S. West Coast (84 FR 49214), between the 6.1-meter depth contour and the 200-meter depth contour from the U.S. international border with Canada south to Point Sur, California. The essential physical and biological features of Southern Resident killer whale proposed critical habitat in coastal waters are: 1) water quality to support growth and development; 2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and 3) inter-area passage conditions to allow for migration, resting, and foraging.

From our Environmental Baseline (Section 7), anthropogenic threats to coastal water quality include contaminants and run-off, oil spills, marine debris, and climate change. While degraded water quality has had large-scale effects on areas designated as inland critical habitat (i.e., Puget Sound – Georgia Basin marine ecosystems), there is less information available on the effects of

potential stressors on water quality in proposed coastal critical habitat for Southern Resident killer whales.

Southern Resident killer whales are adversely affected by limitations of their primary prey, Chinook salmon (Matkin et al. 2017). Salmon have declined because of land alteration throughout the Pacific Northwest associated with agriculture, timber harvest practices, the construction of dams, urbanization, fishery harvest practices, and hatchery operations. These activities have negatively impacted the quantity, quality, and availability of prey to support Southern Resident killer whale individual growth, reproduction and development, as well as overall population growth (Chasco et al. 2017b; Matkin et al. 2017; NMFS 2019e; Wasser et al. 2017).

As noted above for inland critical habitat, commercial and recreational vessel activity can interfere with movements of the whales and impact the passage PBF. Vessel presence can cause killer whales to swim further and change direction more often, which potentially increases energy expenditure and impacts foraging behavior. Whereas vessel noise and disturbance is considered a major threat in inland waters, there is less information on the effects of vessels on Southern Resident killer whales in coastal areas. However, the lower anticipated levels of vessel activity offshore combined with more space for whales to maneuver and search for prey should lessen the impacts of this stressor in proposed offshore critical habitat.

As part of our effects analysis above, we evaluated the effects of the proposed action on the essential PBFs of Southern Resident killer whale proposed coastal critical habitat. We summarize our findings here. For details refer to Section 8.3.1.2.

Several of the activities that may overlap with Southern Resident killer whale proposed coastal critical habitat would have no pathway through which the resulting stressors could affect any of the essential features of critical habitat (i.e., water quality, prey, or passage conditions). These stressors include electromagnetic devices, expended wires and cables, small decelerators and parachutes, and ingestion stressors including fragments from non-explosive practice munitions, explosives, and targets. The effects of metals and chemicals introduced into seawater and sediments as a result of the proposed action were found to be discountable. Any secondary effects of potential stressors (including explosives) resulting from the proposed action on Southern Resident killer whales via impacts to water quality were found to be either insignificant or discountable. Similarly, we found that none of the essential features of Southern Resident killer whale proposed coastal critical habitat (i.e., water quality, prey, or inter-area passage) would likely be adversely affected by potential stressors from activities involving the following: lasers, decelerators and parachutes, chemical and biological simulants, or expended chaff and flares.

We evaluated the effects of acoustic stressors on the inter-area passage essential feature of Southern Resident killer whale proposed coastal critical habitat (i.e., inter-area passage conditions to allow for migration, resting, and foraging). The Navy's quantitative analysis

estimates 46 exposures resulting in behavioral harassment and two exposures resulting in TTS per year as a result of NWTT acoustic stressors in the offshore area. With an estimated population size of 72, each individual killer whale would be exposed to Navy sonar at levels resulting in behavioral harassment or TTS less than once per year, on average. Some proportion of these exposures would likely occur within the Quinault Range Site, which was excluded from proposed coastal critical habitat. Since the proposed use of sonar and other transducers during NWTT activities would be transient and temporary, any effects on Southern Resident killer whales would likely be temporary. Adverse habitat-related effects from acoustic stressors are more likely to result from chronic or long-lasting noise sources that degrade the value of habitat by interfering with the sound-reliant animal's ability to gain benefits from that habitat (i.e., altering the conservation value of the habitat) (NMFS 2019e). None of the sound sources resulting from the proposed action that may overlap with Southern Resident killer whale proposed critical habitat are considered chronic or long-lasting. Instead, exposure to vessel noise, aircraft noise, weapons noise, explosives and sonar as a result of the proposed action would be transient, brief, and temporary, with no lasting or appreciably impact on an individual killer whale's ability to migrate, feed or rest. Proposed mitigation measures, as described in Section 3.5, would further reduce any potential effects of acoustic stressors on the Southern Resident killer whale's soundscape. As such, any effects of Navy acoustic stressors on inter-area passage conditions would not likely reduce the conservation value of Southern Resident killer whale proposed coastal critical habitat.

While seafloor devices could obstruct killer whale movements, considering the size of these devices, the mobility of killer whales to avoid or swim around them, and the temporary nature of deployment, any effects on passage associated with obstruction from seafloor devices would likely be insignificant.

We evaluated the effects of the proposed action on the prey essential feature of Southern Resident killer whale proposed coastal critical habitat. We found that the effects of the following potential stressors on the quantity, quality, and availability of Southern Resident killer whale prey would not likely reduce the conservation value of proposed coastal critical habitat: sonar and other transducers, in-water devices, and weapons firing and impact noise.

The use of Navy explosives within (or adjacent to) areas proposed as coastal critical habitat would likely result in the mortality and injury of salmon populations preyed upon by Southern Resident killer whales. With the exception of a small number of explosive mine countermeasure and neutralization testing events, all explosive use in the offshore portion of the action area would occur greater than 50 nautical miles from shore, and, therefore, outside of the areas proposed as Southern Resident killer whale coastal critical habitat. Mine countermeasure and neutralization testing involving the use of explosives would occur in waters three nautical miles or greater from shore within the Quinault Range Site (outside the Olympic Coast National Marine Sanctuary) or 12 nautical miles or greater from shore elsewhere in the NWTT offshore area.

From our analysis above we found that an extremely small percentage of each Chinook salmon ESU would likely be killed from NWTT explosive activities during a maximum year of explosive activity levels (i.e., less than 0.02 percent across all ESUs, life stages, and hatchery/natural origin). We concluded that the impacts of salmon mortalities as a result of the proposed action on the nutritional needs of the Southern Resident killer whales population would likely be extremely small. From our quantitative analysis (Section 8.2.1.4), we expect a significantly larger number of salmon would be injured (as compared to killed) as a result of NWTT explosives. Several of the highest priority Chinook salmon ESUs for Southern Resident killer whale conservation also had some of the highest estimated injury rates (as a proportion of the population size) from offshore explosives among the ESUs analyzed. We expect that some proportion of salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected. Salmon that are injured (but not killed) by an explosion would still be available as prey and could be more easily captured by Southern Resident killer whales actively feeding within the proposed critical habitat area.

Whereas our species effects analysis considered the impacts to Southern Resident killer whale prey in all portions of the action area, our proposed critical habitat effects analysis is more narrowly focused only on those impacts that are likely to occur within proposed critical habitat. As noted above, some proportion of the explosions associated with this activity would likely occur outside of proposed coastal critical habitat (i.e., either within the excluded Quinault Range Site, the excluded buffer area, or beyond the 200 meters depth contour that defines the western edge of proposed critical habitat). Thus, any effects on prey occuring within the Quinault Range Site, the excluded buffer area, or beyond the 200 meter depth contour were not considered in our proposed coastal critical habitat effects analysis.

The Navy has proposed to conduct up to two mine countermeasure neutralization testing events per year. Based on information provided by the Navy, it is likely that at least one of these events, and possibly both events within a given year, would be conducted within the Quinault Range Site (Navy 2020d). Mine countermeasure and neutralization testing is limited by availability of certain depths, bottom types, and other bathymetric conditions (e.g., water clarity) which may only be found in certain portions of the Quinault Range Site (Navy 2020d). This testing activity is typically conducted in water depths of 300 ft., and in some cases may require testing in water depths of less than 100 feet (Navy 2020d). Given the likelihood that a large proportion of the mine countermeasure and neutralization explosives would occur within the Quinault Range Site, we expect the impacts of the proposed action on Southern Resident killer prey within proposed coastal critical habitat to be considerably smaller than the overall impacts to prey throughout the action area (as described in our species effects analysis in Section 8.2.1.4). In addition, as discussed previously (Section 8.2.3.1), our fish effects analysis is based on a number of conservative assumptions that likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

Therefore, while explosives may result in injury and mortality to Southern Resident killer whale prey species within proposed coastal critical habitat, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of this endangered DPS. The anticipated extent of injury and mortality of Southern Resident killer whale salmonid prey within proposed critical habitat resulting from NWTT explosives is extremely low, and would not be detectable above baseline conditions and assessed levels of take from other anthropogenic stressors including commercial and recreational fisheries. In addition, the Navy has proposed mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months of October through June (see Section 3.5.1.9 for details). Since this timeframe includes months when peak densities of many Chinook salmon ESUs would be expected in the action area, this mitigation measure would likely result in reduced impacts to several of the Chinook salmon ESUs preyed upon by Southern Resident killer whales. Southern Residents are also more likely to occur in the offshore portion of the action area from October through June as compared to July through September, when they are more often seen in the inland waters. These conservation measures would likely further reduce the impacts of offshore explosives on the prey PBF of Southern Resident killer whales. Therefore, based on the available information, we find the effects of explosives on the prey PBF of Southern Resident killer whale proposed coastal critical habitat to be insignificant.

In summary, many of the stressors associated with activities in the offshore portion of the action area would either not overlap with designated critical habitat or would have no pathway to affect the PBFs of Southern Resident killer whale proposed coastal critical habitat. The effects of all other stressors analyzed, including sonar and explosives, on the essential PBFs were found to be either insignificant or discountable and not likely to reduce the conservation value of proposed critical habitat. We conclude that the proposed action is not likely to adversely affect Southern Resident killer whale proposed coastal critical habitat.

Therefore, based on our critical habitat effects analysis presented in Section 8.3.1.2, and the best available information regarding the risk of those effects, we do not anticipate the effects of the proposed action would appreciably diminish the value of Southern Resident killer whale DPS proposed coastal critical habitat as a whole for the conservation of this species.

10.5 Sei Whale

Historically, whaling was the largest threat to sei whales; whaling ultimately led to sei whales being listed as endangered. Current threats to sei whales include commercial fishing, vessel strikes, and pollution. The best abundance estimate for sei whales for the waters of the U.S. West Coast is 519 (CV=0.40) (Carretta et al. 2020). Population growth rates for sei whales are not available at this time as there are little to no systematic survey efforts to study sei whales.

Sei whales are only present in the offshore portion of the NWTT action area. Sei whales may be exposed to acoustic stressors associated with training and testing activities throughout the year.

Sei whales found in the offshore portion of the NWTT action area are recognized as part of the Eastern North Pacific stock. The acoustic effects analysis predicts that sei whales from the Eastern North Pacific stock would be exposed to sonar and other active acoustic sources associated with training activities in the offshore area that would result in 16 behavioral reactions and 14 TTS exposures annually, and 111 behavioral reactions and 95 TTS exposures every seven years of the proposed action. Sei whales may also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic effects analysis predicts that sei whales from the Eastern North Pacific stock would be exposed to sonar and other active acoustic sources associated with testing activities in the offshore area that would result in 16 behavioral reactions and 35 TTS exposures annually, and 78 behavioral reactions and 168 TTS exposures every seven years of the proposed action. Acoustic modeling predicts that sei whales from the Eastern North Pacific stock would be exposed to impulses from explosive sources associated with testing activities in the offshore area that would result in one behavioral reaction and one TTS exposure annually and eight behavioral reactions and three TTS exposures every seven years of the proposed action. Overall, NWTT training and testing activities would result in an estimated 0.5 behavioral disruptions annually per sei whale from the Eastern North Pacific stock (see Table 134).

The 2011 sei whale recovery plan defines three recovery populations by ocean basin (the North Atlantic, North Pacific, and Southern Hemisphere) and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with fisheries, climate change, direct harvest, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. As described previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. Downlisting criteria for fin whales includes the maintenance of 1,500 mature, reproductive individuals with at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sei whale recovery populations.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by from training and testing activities the Navy will conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA regulations from November 2020 through November 2027, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future and related

incidental take specified in the MMPA Rule would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. NWTT stressors and related incidental take specified in the MMPA Rule will not affect the population dynamics, behavioral ecology, and social dynamics of individual sei whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

10.6 Sperm Whale

Sperm whale populations have been drastically reduced as a result of whaling. Current anthropogenic threats to sperm whale populations include ship strikes, entanglement in fishing gear, competition for resources due to overfishing, loss of prey and habitat due to climate change, and noise. The species' large global population size shows that it is somewhat resilient to current threats. There is insufficient data to evaluate trends in abundance and growth rates of sperm whales at this time. Sperm whales found in the offshore portion of the NWTT action area are recognized as part of the California/Oregon/Washington stock. The California/Oregon/Washington sperm whale stock is estimated to consist of 1,997 individuals (N_{min}=1,270) (Carretta et al. 2020).

Sperm whales are only present in the offshore portion of the NWTT action area. Sperm whales may be exposed to acoustic stressors associated with training and testing activities throughout the year. The acoustic effects analysis predicts that sperm whales of the California/Oregon/Washington stock would be exposed to sonar and other active acoustic sources associated with training activities in the offshore area that would result in 510 behavioral reactions and two TTS exposures annually, and 3,562 behavioral reactions and 12 TTS exposures every seven years of the proposed action. Sperm whales may also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic analysis predicts that sperm whales of the California/Oregon/Washington stock would be exposed to sonar and other active acoustic sources associated with testing activities in the offshore area that would result in 324 behavioral reactions and three TTS exposures annually, and 1,424 behavioral reactions and 19 TTS exposures every seven years of the proposed action. Overall, NWTT training and testing activities may result in an estimated 0.67 behavioral disruptions annually per sperm whale from the California/Oregon/Washington stock (see Table 134). Based on our vessel strike analysis, we anticipate that no more than one sperm whale would be struck by Navy vessels over the seven-year duration of the MMPA rule.

The 2010 sperm whale recovery plan defines three recovery populations by ocean basin (the Atlantic Ocean/Mediterranean Sea, Pacific Ocean, and Indian Ocean) and sets criteria for the

downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with fisheries, climate change, direct harvest, oil spills, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. As discussed previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. Downlisting criteria for sperm whales includes the maintenance of 1,500 mature, reproductive individuals with at least 250 mature females and 250 mature males in each recovery population. To qualify for downlisting, each recovery population must also have no more than a one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sperm whale recovery populations.

NWTT acoustic stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual sperm whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species from acoustic stressors as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated. Due to a lack of fitness consequences to individuals and the populations they represent, we also do not anticipate any reductions in survival rate or trajectory of recovery of sperm whales as currently proposed from acoustic stressors.

Regarding vessel strikes, we anticipate around 0.05 percent of sperm whales from the California/Oregon/Washington stock may be killed or seriously injured over a seven-year period as a result of the proposed action. In our judgement, the removal of such a small proportion of the population would likely have a very minor negative impact on the population trend for this stocks. The loss (or serious injury) of this individual from the California/Oregon/Washington stock (currently estimated at just under 2,000 whales) is not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. For this reason, the effects of the proposed action from vessel strikes are not expected to appreciably reduce the likelihood of survival and recovery of sperm whales.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy would conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA Rule from November 2020 to November 2027, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of sperm whales in the wild by reducing the reproduction, numbers, or

distribution of that species. We also conclude that effects from ongoing Navy training and testing activities and related incidental take specified in the MMPA Rule continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of sperm whales in the wild by reducing the reproduction, numbers, or distribution of that species.

10.7 Guadalupe Fur Seal

Guadalupe fur seal population numbers have been drastically reduced due to historic commercial hunting. Guadalupe fur seals are also under threat from additional anthropogenic stressors including commercial fishing and climate change. The current minimum population estimate is 31,019 individuals, which is estimated to be growing at approximately 13.7 percent per year (Carretta et al. 2020).

Guadalupe fur seals are present within the coastal margins of the offshore portion of the action area during the warm season (summer and early autumn). Guadalupe fur seals are considered "seasonal" migrants since they return to rookeries in Mexican waters in the cold season. The acoustic effects analysis predicts that Guadalupe fur seals would be exposed to sonar and other active acoustic sources associated with training activities in the offshore area that would result in 605 behavioral reactions and three TTS exposures annually, and 4,226 behavioral reactions and 21 TTS exposures every seven years of the proposed action. Additionally, Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with testing activities. The acoustic effects analysis predicts that Guadalupe fur seals would be exposed to sonar and other active acoustic sources associated with testing activities in the offshore area that would result in 877 behavioral reactions and ten TTS exposures annually, and 3,958 behavioral reactions and 64 TTS exposures every seven years of the proposed action. Overall, NWTT training and testing activities would result in an estimated 0.547 behavioral disruptions annually per Guadalupe fur seal from the Mexico stock (see Table 134). Acoustic modeling predicts that Guadalupe fur seals may be exposed to impulses from explosive sources associated with training and testing activities in the offshore area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds.

We do not anticipate any instances of PTS, injury or mortality from any stressors associated with training or testing activities. Pinnipeds in the water are tolerant of anthropogenic noise and activity. The available scientific information does not provide evidence that behavioral responses of pinnipeds to acoustic stressors from Navy training and testing activities will impact the fitness of any individuals of this species. Therefore, exposure to acoustic stressors will not have population or species level impacts.

The Guadalupe fur seal does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of individuals would occur and that effects from acoustic stressors would be temporary and not impact the fitness of individuals

or the population. In the absence of fitness consequences on individuals or the population to which those individuals belong, we do not expect an appreciable reduction in the ability of this species to recover.

Therefore, based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy would conduct in the NWTT action area on an annual basis, cumulatively over the seven year period of the MMPA Rule from November 2020 to November 2027, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Guadalupe fur seals in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities and related incidental take specified in the MMPA Rule continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Guadalupe fur seals in the wild by reducing the reproduction, numbers, or distribution of that species. NWTT stressors and related incidental take specified in the MMPA Rule will not affect the population dynamics, behavioral ecology, and social dynamics of individual seals in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual Guadalupe fur seals would not be likely to reduce the viability of the populations those individual Guadalupe fur seals represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

10.8 Leatherback Sea Turtle

The leatherback sea turtle was listed as endangered throughout its entire range on June 2, 1970, under the Endangered Species Conservation Act of 1969, a precursor to the ESA. The primary threats to leatherback sea turtles include fisheries bycatch, harvest of nesting females, and egg harvesting. Plastic ingestion is also common in leatherbacks and can block gastrointestinal tracts leading to death.

Pacific leatherbacks are split into western and eastern Pacific subpopulations based on their distribution and biological and genetic characteristics. Only western Pacific leatherbacks are expected to be found within the NWTT action area. Western Pacific leatherbacks nest in the Indo-Pacific, primarily in Indonesia, Papua New Guinea and the Solomon Islands. Spotila et al. (2000) estimated that the Pacific leatherback population declined from an estimated 81,000 adult turtles to 2,955 females (adult and subadult) in the two decades from 1980 to 2000. Martin et al. (2020) estimated the abundance of western Pacific leatherbacks for the two index beaches in Indonesia, which represent approximately 75 percent of all nesting individuals. Using the median value for imputed nest counts they estimated 790 total nesters (95 percent CI: 666–942). Jones et

al. (2018) used model-estimated annual female distributions for 2015 to 2017 to estimate an index of current total reproductive female abundance for the western Pacific leatherback population. This was computed as a 3-year run sum based on an assumed 3-year remigration interval. The estimates for 2015-2017 annual females ranged from 340 to 439 and the summed total reproductive female estimate was 1,180 (95 percent CI: 949–1,479) (Jones et al. 2018). Using this estimate, and assuming a 3:1 ratio of females to males, NMFS (2019g) estimated the current adult portion of the population is 1,851 (1,488-2,320). NMFS (2019g) used the proportion or change in the estimates derived from the information contained in Jones et al. (2018) to estimate the current population size of the West Pacific Ocean leatherback sea turtle. The total West Pacific Ocean population estimate is 175,000 leatherback sea turtles, but may range between 68,000 and 360,000 individuals (NMFS 2019g).

Population growth rates for leatherback sea turtles vary by ocean basin. Leatherbacks have been in decline in all major Pacific basin rookeries (nesting areas/groups) for at least the last two decades (Dutton et al. 2007; Gilman 2008; NMFS and USFWS 2007b; Sarti M. 1996; Spotila et al. 1996; Spotila et al. 2000; Tapilatu et al. 2013; TEWG 2007). Based on counts of leatherbacks at nesting beaches in the western Pacific, Tapilatu et al. (2013) estimated that the subpopulation has been declining at a rate of almost six percent per year since 1984. Based on a recent population assessment, Martin et al. (2020) reported a declining trend for western Pacific leatherback sea turtles of negative 6.1 percent annually. Estimated leatherback densities in the offshore portion of the action area are very low (i.e., 0.000114 per km^2) and nesting sites for the western Pacific subpopulation are far removed from the proposed action.

Based on our *Effects Analysis* (Section 8.2.2.1), we anticipate a small number of leatherback sea turtles (i.e., up to four estimated exposures per year, 26 exposures over seven years) could experience behavioral and physiological stress responses from exposure to explosives used during NWTT activities. The Navy's quantitative model predicts that no leatherbacks are likely to be exposed to levels of explosive sound and energy that could cause injury, PTS, or TTS. Behavioral responses from explosives are expected to be short-term, and are not anticipated to result in reduced fitness of individual turtles. In addition, since leatherback nesting does not occur within the action area (Frey et al. 2013; Kittinger et al. 2013; Summers et al. 2018) behavioral responses to explosives would have no impact on reproductive behavior or nesting success. The effects of all other potential stressors (i.e., acoustic, energy, physical disturbance and strike, entanglement, and ingestion) on sea turtles analyzed in this opinion were found to be either discountable or insignificant.

In summary, we anticipate an extremely small number of individual leatherbacks, relative to the population size, would be affected by the proposed action. Those effects would likely be limited to only minor, short-term behavioral responses with no resulting reductions in numbers, reproduction or individual fitness. We do not anticipate any effects from the proposed action would result in the mortality, injury or reduced fitness of individual leatherback sea turtles. The impacts expected to occur and affect leatherback sea turtles in the action area are not anticipated

to result in appreciable reductions in overall reproduction, numbers, or distribution of this species.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the leatherback sea turtle in the wild by reducing the reproduction, numbers, or distribution of the DPS. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the leatherback sea turtle in the wild by reducing the reproduction, numbers, or distribution of this species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the leatherback sea turtle.

10.8.1 Designated Critical Habitat

In Section 8.3.2 we evaluated the effects of the proposed action on the one identified essential physical and biological feature of leatherback designated critical habitat within the action area: *The occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (e.g., Chrysaora, Aurelia, Phacellophora, and Cyanea), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks*. We found that any impacts on leatherback critical habitat through effects to prey from acoustic stressors and physical disturbance and strike stressors resulting from the proposed action would likely be insignificant. We also found that although explosives may result in injury and mortality to leatherback prey species, we have no information to indicate that this stressor would have a measureable impact on the occurrence of prey species of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. Therefore, based on our critical habitat effects analysis presented in Section 8.3.2, we do not anticipate the effects of the proposed action will appreciably diminish the value of leatherback sea turtle designated critical habitat as a whole for the conservation of this species

10.9 Chinook Salmon

Within the action area, nine ESUs of Chinook salmon may be exposed to explosives associated with the Navy's proposed NWTT activities throughout the year. These include the endangered Sacramento River winter-run and Upper Columbia River spring-run ESUs, and threatened California coastal, Central Valley spring-run, Lower Columbia River, Puget Sound, Snake River fall-run, Snake River spring/summer-run, and Upper Willamette River ESUs. Listing dates for each of these Chinook salmon ESUs are provided in Section 6.2. Primary threats to Chinook salmon include blocked access to spawning grounds, habitat degradation caused by dams and

culverts, and commercial fishing. Further, impacts from recent draughts have also caused the species population numbers to decrease.

Any Chinook salmon located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E4 and E7). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast.

Adult and juvenile Chinook salmon both occur in offshore and inland blast sites within the NWTT action area. Although all nine Chinook salmon ESUs are likely to be adversely affected by impacts from NWTT offshore explosive activities shown in Table 149 (solely from blasts within bin E4 and E7), only Chinook salmon from the Puget Sound ESU are likely to be adversely affected by inland explosives in Hood Canal and Crescent Harbor. The maximum annual total number of estimated injuries and mortalities along with the proportion of Chinook salmon injured or killed from all nine ESUs likely to be adversely affected by NWTT explosive activities are presented in Tables 161-163. These estimates represent the maximum annual number of injuries and mortalities based on the maximum annual number of explosives proposed. Further, the average annual estimated injuries and mortalities over seven years of NWTT training and testing activities are presented in Tables 164-166. These tables also detail the annual proportion of Chinook salmon injured or killed from all nine ESUs likely to be adversely affected by NWTT explosive activities[25].

[25] The percentage of population killed or injured assumes the status of the species does not change over the course of course of the seven-year period.

Table 161. Percent of each Chinook salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central valley spring run - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	California coastal - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Snake River fall - T	4	476	40,653	0.0098%	1.1709%
	Juvenile		116	14,197	2,862,418	0.0041%	0.4960%
	Adult	Snake River spring/summer - T	-	14	1,263	-	1.1085%
	Juvenile		31	3,845	775,305	0.0040%	0.4959%
	Adult[1]	Lower Columbia River - T	2	273	115,782	0.0017%	0.2358%
	Juvenile		44	5,421	962,458	0.0046%	0.5632%
	Adult[1]	Upper Willamette River - T	9	1,033	94,428	0.0095%	1.0940%
	Juvenile		-	1	157	-	0.6369%
	Adult	Upper Columbia River spring - E	1	109	10,092	0.0099%	1.0801%
	Juvenile		15	1,828	368,642	0.0041%	0.4959%
	Adult[1]	Puget Sound - T	2	152	46,629	0.0051%	0.3268%
	Juvenile		330	33,823	7,271,130	0.0045%	0.4652%

[1] Includes mortality and injury estimates for hatchery fish with intact and clipped adipose fins.

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 162. Percent of each Chinook salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (adipose clipped)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	40	6,696	-	0.5974%
	Juvenile		10	1,199	200,000	0.0050%	0.5995%
	Adult	Central valley spring run - T	-	41	6,819	-	0.6013%
	Juvenile		106	13,005	2,169,329	0.0049%	0.5995%
	Adult	California coastal - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Snake River fall - T	3	440	46,524	0.0064%	0.9457%
	Juvenile		101	12,319	2,483,713	0.0041%	0.4960%
	Adult	Snake River spring/summer - T	-	77	7,161	-	1.0753%
	Juvenile		180	22,089	4,453,663	0.0040%	0.4960%
	Adult	Lower Columbia River - T	-	-	-	-	-
	Juvenile		1,441	176,612	31,353,395	0.0046%	0.5633%
	Adult	Upper Willamette River - T	-	-	-	-	-
	Juvenile		210	25,759	4,709,045	0.0045%	0.5470%
	Adult	Upper Columbia River spring - E	2	202	18,678	0.0107%	1.0815%
	Juvenile		25	3,084	621,759	0.0040%	0.4960%
	Adult	Puget Sound - T	-	-	-	-	-
	Juvenile		1,195	155,811	36,297,500	0.0033%	0.4293%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 163. Percent of each Chinook salmon ESU (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	4	630	-	0.6349%
	Juvenile		10	1,171	195,354	0.0051%	0.5994%
	Adult	Central valley spring run - T	1	67	11,181	0.0089%	0.5992%
	Juvenile		38	4,649	775,474	0.0049%	0.5995%
	Adult	California coastal - T	1	127	21,102	0.0047%	0.6018%
	Juvenile		63	7,662	1,278,078	0.0049%	0.5995%
	Adult	Snake River fall - T	3	364	31,011	0.0096%	1.1726%
	Juvenile		28	3,436	692,819	0.0040%	0.4959%
	Adult	Snake River spring/summer - T	3	416	38,394	0.0089%	1.0833%
	Juvenile		41	4,997	1,007,526	0.0041%	0.4960%
	Adult	Lower Columbia River - T	2	208	88,407	0.0023%	0.2353%
	Juvenile		540	66,159	11,745,027	0.0046%	0.5633%
	Adult	Upper Willamette River - T	3	335	30,609	0.0090%	1.0945%
	Juvenile		54	6,629	1,211,863	0.0045%	0.5470%
	Adult	Upper Columbia River spring - E	1	93	8,616	0.0089%	1.0833%
	Juvenile		19	2,325	468,820	0.0041%	0.4959%
	Adult	Puget Sound - T	1	124	67,194	0.0021%	0.1842%
	Juvenile		226	36,047	3,035,288	0.0074%	1.1876%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 164. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central valley spring run - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	California coastal - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Snake River fall - T	4	433	37,557	0.0087%	1.0645%
	Juvenile		50	6,084	2,862,418	0.0017%	0.2126%
	Adult	Snake River spring/summer - T	0	12	1,263	0.0081%	0.9834%
	Juvenile		13	1,648	775,305	0.0017%	0.2126%
	Adult[1]	Lower Columbia River - T	1	117	115,782	0.0009%	0.1009%
	Juvenile		19	2,323	962,458	0.0020%	0.2414%
	Adult[1]	Upper Willamette River - T	8	938	94,428	0.0081%	0.9936%
	Juvenile		-	-	157	-	-
	Adult	Upper Columbia River spring - E	1	99	10,092	0.0081%	0.9834%
	Juvenile		6	784	368,642	0.0017%	0.2126%
	Adult[1]	Puget Sound - T	2	114	46,629	0.0037%	0.2436%
	Juvenile		204	18,351	7,271,130	0.0028%	0.2524%

[1] Includes mortality and injury estimates for hatchery fish with intact and clipped adipose fins.

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 165. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (hatchery produced with adipose fin-clip) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (adipose clipped)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	17	6,696	0.0021%	0.2560%
	Juvenile		4	514	200,000	0.0021%	0.2569%
	Adult	Central valley spring run - T	-	18	6,819	0.0021%	0.2577%
	Juvenile		45	5,573	2,169,329	0.0021%	0.2569%
	Adult	California coastal - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Snake River fall - T	3	399	42,981	0.0070%	0.8586%
	Juvenile		43	5,279	2,483,713	0.0017%	0.2126%
	Adult	Snake River spring/summer - T	1	70	7,161	0.0081%	0.9834%
	Juvenile		77	9,467	4,453,663	0.0017%	0.2126%
	Adult	Lower Columbia River - T	-	-	-	-	-
	Juvenile		618	75,691	31,353,395	0.0020%	0.2414%
	Adult	Upper Willamette River - T	-	-	-	-	-
	Juvenile		90	11,040	4,709,045	0.0019%	0.2344%
	Adult	Upper Columbia River spring - E	2	184	18,678	0.0081%	0.9834%
	Juvenile		11	1,322	621,759	0.0017%	0.2126%
	Adult	Puget Sound - T	-	-	-	0.0021%	0.2560%
	Juvenile		564	78,577	36,297,500	0.0021%	0.2569%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 166. Average annual impacts every seven years to ESA-listed Chinook salmon ESUs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chinook salmon	Adult	Sac River winter run - E	-	2	630	-	0.2494%
	Juvenile		4	502	195,354	0.0021%	0.2569%
	Adult	Central valley spring run - T	-	29	11,181	-	0.2568%
	Juvenile		16	1,992	775,474	0.0021%	0.2569%
	Adult	California coastal - T	-	54	21,102	-	0.2573%
	Juvenile		27	3,284	1,278,078	0.0021%	0.2569%
	Adult	Snake River fall - T	3	330	31,011	0.0081%	1.0645%
	Juvenile		12	1,473	692,819	0.0017%	0.2126%
	Adult	Snake River spring/summer - T	3	378	38,394	0.0130%	0.9834%
	Juvenile		17	2,142	1,007,526	0.0017%	0.2126%
	Adult	Lower Columbia River - T	1	89	88,407	0.0008%	0.1010%
	Juvenile		231	28,354	11,745,027	0.0020%	0.2414%
	Adult	Upper Willamette River - T	2	304	30,609	0.0081%	0.9936%
	Juvenile		23	2,841	1,211,863	0.0019%	0.2344%
	Adult	Upper Columbia River spring - E	1	85	8,616	0.0081%	0.9834%
	Juvenile		8	997	468,820	0.0017%	0.2126%
	Adult	Puget Sound - T	1	68	67,194	0.0011%	0.1008%
	Juvenile		174	29,589	3,035,288	0.0057%	0.9748%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

As shown in Tables 161-163, an extremely small percentage of each Chinook salmon ESU would likely be killed from NWTT explosive activities during a maximum year of explosive activity levels (i.e., less than 0.02 percent across all ESUs, life stages, and hatchery/natural origin). Since these estimates are based on a maximum year, in many years an even smaller percentage of each population would likely be killed (See Tables 164-166). While the loss of an individual fish represents the loss of future reproductive potential, given the small anticipated

mortality rates, it is unlikely that salmon mortality from NWTT explosives would have an appreciable or measurable impact on survival or recovery of any of the affected Chinooks ESUs.

A larger number of adult Chinook salmon would likely be injured than killed as a result of NWTT explosives. The impacts due to injury, in terms of numbers and proportion of the population, vary substantially by ESU, life stage, and natal origin (Tables 161-163). The ESUs with the largest percentage of injury are the Snake River fall, Snake River spring/summer, Upper Columbia River spring, and Upper Willamette River ESUs as a result of being the only adult Chinook salmon populations that could incur effects from blasts during offshore testing and training activities that occur in waters greater than 50 nautical miles from shore. For these four Chinook salmon ESUs, the maximum annual adult injury rates varied between 0.94 and 1.17 percent of the population, with some variation based on natal origin (i.e., hatchery fin clip, hatchery fin intact, and natural). Average annual adult injury rates (Tables 164-166) over a seven year period were only slightly lower than the maximum annual rate for these ESUs.

It should be noted that the majority of explosive bins that Snake River fall, Snake River spring/summer, Upper Columbia River spring, and Upper Willamette River ESUs could be exposed to are from offshore training and testing activities that are proposed to occur in depths greater than 50 nautical miles from shore (See Table 145 and Table 149). Catch data and interviews with commercial fishermen indicate that maturing Chinook salmon are found in highest concentrations along the continental shelf within 60 km (37.3 mi) of the Washington, Oregon, and California coastlines (Pacific Fishery Management Council 2014a). Although Snake River fall, Snake River spring/summer, Upper Columbia River spring, and Upper Willamette River ESUs can occur at depths greater than 50 nautical miles (Sharma and Quinn 2012), fisheries data show that Chinook salmon occur in lower densities in these areas within the action area. However, due to a lack of available information on Chinook salmon densities in areas greater than 50 nm, we were unable to appropriately quantify the reduction in density for these deeper areas[26]. Due to lower densities of Chinook salmon in the far offshore marine environments, the injury and mortality estimates for adult Snake River, Upper Columbia River, and Willamette ESUs are likely conservatively high.

For juvenile Chinook, except for the Puget Sound ESU, the proportion of the population injured during a maximum year of explosive activity was less than 0.5 percent across all ESUs, life stages, and hatchery/natural origin. Average annual juvenile injury rates (Tables 164-166) over a seven year period were less than 0.3 percent across all Chinook salmon ESUs, except for Puget Sound. The Puget Sound ESU is the only Chinook salmon ESU to overlap with both offshore and inland NWTT explosive activities. For this ESU the maximum annual adult injury rates were around 0.5 percent of the hatchery origin fish population, and 1.2 percent of the naturally produced population. Average annual juvenile injury rates for Puget Sound ESU Chinook

[26] As noted in Section 2.4.2.1, we applied a uniform distribution density for Chinook salmon ESUs located in areas greater than 50nm from shore.

salmon were around 0.2 percent of the hatchery origin fish population and 1 percent of the naturally produced population.

Due to salmonids having higher rates of occurrence in the nearshore habitat, the Navy agreed, to the maximum extent practicable, to conduct mine countermeasure neutralization testing from July through September when operating within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area. In scenarios where this is not practicable, the Navy agreed to conduct no more than 60 percent of explosives (annually and over seven years) from October through June (see Section 3.5.1.9 for details). Since this timeframe includes months when peak densities of many Chinook salmon ESUs would be expected in the action area (specifically within the nearshore habitat), this mitigation measure would likely result in reduced impacts to several of the Chinook salmon ESUs analyzed. This is especially evident for adults from the Snake River spring/summer, Upper Columbia River spring, and Upper Willamette River ESUs which congregate at the mouth of the Columbia during this mitigation time period, significantly reducing the probability for higher rates of injury or mortality to occur. As stated in Section 6.2, the documented life histories for each of these ESUs show adults moving into the Columbia River from early February to late May (Myers et al. 1998b). Although we could not factor this mitigation into our quantitative analysis used for estimating injury and mortality, the Navy's proposal to move mine countermeasure neutralization testing events to months that would have less of an effect on spring- and fall-run Chinook salmon would likely reduce the estimated number of injuries and mortalities from our quantitative analysis. Due to this proposed mitigation and lower densities of Chinook in waters greater than 50 nautical miles from shore, the injury and mortality estimates for adult Snake River, Upper Columbia River, and Willamette ESUs are likely conservatively high. The estimated impacts to the Puget Sound ESU are also likely conservatively high because the Navy will avoid, to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (0.5-2.5 pounds) in Hood Canal, and only conduct training with one ounce or less (E0) charges during August, September, and October.

It is worth noting that, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis may also be conservatively high due to several conservative assumptions in our approach. The estimated number of Chinook salmon injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of salmonid fish injury from Navy explosives.

Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria for the most

sensitive fish species and life stages. This is likely a conservative approach for adult and subadult Chinook salmon which, given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on. If injured, large adult and subadult Chinook would also likely recover faster from sublethal injuries, as compared to juveniles, with a lower likelihood of fitness consequences or long-term effects on their survival or future reproductive potential.

Overall, the level of mortality and injury represents a reduction in abundance that may impact the future reproductive potential of Chinook populations but is not likely to appreciably reduce the likelihood of survival and recovery of any ESA-listed Chinook salmon ESU. As stated in 8.2.1.4, a considerably larger total estimated number of Chinook salmon would likely be injured than killed as a result of NWTT explosives. Although we cannot quantify based on the available information, we expect that some proportion of salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected. However, we conclude that the diversity of ESA-listed Chinook salmon ESUs will not be affected by this limited amount of mortality or injury because it is expected to be distributed across populations through the species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the NWTT action area would not appreciably reduce the likelihood of ESA-listed Chinook salmon surviving and recovering in the wild.

Some individual Chinook salmon may experience TTS as a result of Navy impulsive acoustic stressors. However, Chinook salmon lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in Chinook salmon migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because Chinook salmon are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of ESA-listed Chinook salmon ESUs in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of ESA-listed Chinook salmon ESUs in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the California coastal, Central Valley spring-run, Lower Columbia River, Puget Sound, Sacramento River winter-run, Snake River fall-run, Snake River spring/summer-run, Upper Columbia River spring-run, and Upper Willamette River ESUs of Chinook salmon.

10.9.1 Puget Sound Chinook Salmon ESU Critical Habitat

The nearshore marine habitat component of Puget Sound Chinook salmon critical habitat overlaps with the inland portion of the NWTT action area. We analyzed the effects of the proposed action on Puget Sound Chinook salmon critical habitat in Section 8.3.3.2 above. The identified estuarine and nearshore marine PBFs within the inland portion of the action area include the following: areas free of obstruction; water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh-and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation. There is no mechanism for the stressors associated with the proposed action to affect the water quantity or salinity PBFs. Many of the stressors resulting from NWTT activities would have no effect on water quality, salmonid natural cover, or salmonid forage within designated critical habitat. As discussed in Section 8.1.3.6 (*Stressors Resulting in Effects to Fish Habitat or Prey*), the secondary effects of stressors resulting from the proposed action on ESA-listed salmonids via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Sound from explosives detonated at Crescent Harbor could overlap ESA-listed salmon designated critical habitat in the inland waters, as this training range is located outside of but adjacent to critical habitat (Navy 2020e). Any effects of this activity would likely be minor, short-term and temporary, only occurring during activities involving explosives, with no lasting effect on prey availability or the pelagic food web expected. While seafloor devices could obstruct fish movements, considering the size of these devices, the

mobility of salmonids to avoid or swim around them, and the temporary nature of deployment, any effects associated with obstruction from seafloor devices would likely be insignificant. Therefore, based on our critical habitat effects analysis presented in Section 8.3.3.2, we do not anticipate the effects of the proposed action would appreciably diminish the value of Puget Sound ESU Chinook salmon designated critical habitat as a whole for the conservation of this species.

10.10 Chum Salmon

Within the action area, two ESUs of ESA-listed chum salmon may be exposed to explosives associated with NWTT activities throughout the year. These include the threatened Columbia River ESU and Hood Canal summer-run ESU which were both listed on March 25, 1999. Major threats to chum salmon include blocked access to spawning grounds and habitat degradation caused by dams and culverts.

As with other ESA-listed fishes in the action area, any chum salmon located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E4 and E7). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast.

Columbia River ESU of chum salmon are likely to be adversely affected by impacts from NWTT offshore explosive activities shown in Table 149 (solely from blasts within bin E4 and E7), while Hood Canal summer-run ESU of chum salmon are likely to be adversely affected by inland explosives at the Hood Canal range site. The maximum annual number of injuries and mortalities along with the proportion of chum salmon injured or killed from the two ESUs likely to be adversely affected by NWTT explosive activities are presented in Tables 167-168. These estimates and proportions represent the maximum number of injuries and mortalities anticipated during a maximum year of explosive activity level (i.e., in some years these estimates would be smaller). Further, the average annual estimated injuries and mortalities over every seven years of NWTT training and testing activities are presented in Tables 169-170. These tables also detail the proportion of ESA-listed chum salmon that are likely to be adversely affected by NWTT explosive activities[27]. All chum salmon assessed in this consultation had an intact adipose fin, therefore, there were no estimates of injury or mortality for hatchery produced chum salmon with clipped adipose fins.

[27] The percentage of population killed or injured assumes the status of the species does not change over the course of course of the seven year period.

Table 167. Percent of each ESA-listed chum salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (intact adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chum salmon	Adult	Hood Canal - T	2	96	3,601	0.0431%	2.6697%
	Juvenile		16	1,171	99,721	0.0157%	1.1744%
	Adult	Columbia River - T	-	-	1,278	0.0000%	0.0000%
	Juvenile		1	109	723,207	0.0001%	0.0151%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 168. Percent of each ESA-listed chum salmon ESU (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chum salmon	Adult	Hood Canal - T	22	1,377	76,193	0.0292%	1.8134%
	Juvenile		224	16,791	3,940,234	0.0057%	0.4326%
	Adult	Columbia River - T	-	5	31,932	-	0.0157%
	Juvenile		8	977	6,504,514	0.0001%	0.0150%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 169. Average annual impacts every seven years to ESA-listed chum salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, - indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (intact adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chum salmon	Adult	Hood Canal - T	2	96	3,601	0.0436%	2.6697%
	Juvenile		16	1,171	99,721	0.0156%	1.1744%
	Adult	Columbia River - T	-	-	1,278	-	-
	Juvenile		-	47	723,207	-	0.0064%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 170. Average annual impacts every seven years to ESA-listed chum salmon ESUs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and ESU that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Chum salmon	Adult	Hood Canal - T	22	1,382	76,193	0.0292%	1.8134%
	Juvenile		224	17,045	3,940,234	0.0057%	0.4326%
	Adult	Columbia River - T	-	2	31,932	-	0.0063%
	Juvenile		3	419	6,504,514	0.0001%	0.0064%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

As shown in Tables 167-168, an extremely small percentage of each chum salmon ESU would be killed from the Navy's NWTT explosive activities during a maximum year of explosives activity (i.e., less than 0.05 percent across all ESUs, life stages, and hatchery/natural origin). Since these estimates are based on a maximum year, in many years an even smaller percentage of each population would likely be killed (See Tables 169-170). While the loss of an individual fish represents the loss of future reproductive potential, given the small anticipated mortality rates, it is unlikely that salmon mortality from NWTT explosives would have an appreciable or measurable impact on survival or recovery of any of the affected chum ESUs.

A larger number of chum salmon would likely be injured than killed as a result of NWTT explosives. The impacts due to injury, in terms of numbers and proportion of the population, vary substantially by ESU, life stage, and natal origin (Tables 167-168). The ESU with the largest percentage of injury is the Hood Canal summer-run ESU as a result of it being the only population that would incur effects from blasts from inland explosive activities. We estimate that about 2.7 percent of adult and 1.2 percent of juvenile hatchery origin Hood Canal summer-run chum salmon may be exposed to injury during a maximum year of NWTT explosives activity level. Further, about 1.8 percent of adult and 0.4 percent of juvenile natural origin Hood Canal summer-run chum salmon may be exposed to injury during a maximum year of NWTT explosives activity level. The Columbia River ESU of chum salmon would be affected at much lower levels (i.e. less than 0.02 percent across all life stages and hatchery/natural origin).

It is worth noting that injury and mortality estimates for Hood Canal summer-run chum salmon are likely conservative because the Navy will avoid, to the maximum extent practicable, the use of larger charges at the Hood Canal site when adult salmon are migrating to spawn (August to October) which is the main timeframe when adults are within Hood Canal. At other times, adult presence in Hood Canal is extremely low (WDFW and Northwest Indian Fisheries Commission 1990). Further, Navy inland explosives are the only explosives expected to overlap with Hood Canal summer-run chum in the action area. These explosive events have a low likelihood to impact adult Hood Canal chum that may be congregating near their river or creek systems to spawn due to the location of the proposed activities. For example, EOD training activity occurs in waters toward the northern portion of Naval Base Kitsap-Bangor. Historically, the nearest Hood Canal summer-run chum stock to this EOD site was located in Big Beef Creek. However, this system has not seen summer-run chum for the last two season (C. Kinsel, WDFW, pers. comm. to C. Hunt, Navy, August 26, 2020). The nearest systems that actively support Hood Canal summer-run chum occur on the west side of the Canal and in the Dabob Bay tributaries which are further away from Navy EOD activities.

Furthermore, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis are also likely conservatively high due to several conservative assumptions in our approach. The estimated number of chum salmon injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of >207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of salmonid fish injury from Navy explosives. Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. This is likely a conservative approach for adult and subadult chum salmon which,

given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on. Due to their size, injured adult and subadult chum would also likely recover faster from sublethal injuries, as compared to juveniles, with a lower likelihood of fitness consequences or long-term effects on their survival or future reproductive potential.

Some individual chum salmon may experience TTS as a result of Navy impulsive acoustic stressors. However, chum salmon lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in chum salmon migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because chum salmon are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects on chum salmon resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Overall, the estimated level of mortality represents a reduction in abundance that may impact the future reproductive potential of chum but is not likely to appreciably reduce the likelihood of survival and recovery of either the Hood Canal summer-run ESU or Columbia River ESU of chum salmon. Less than 0.05 percent of each chum salmon ESU would be killed from the Navy's NWTT explosive activities during a maximum year of explosives activity. Less than 0.02 percent of each Columbia River ESU chum salmon would be injured during a maximum year of explosive activity levels. A considerably larger proportion of chum salmon from the Hood Canal summer-run ESU would likely be injured as a result of NWTT explosives. Although we cannot quantify based on the available information, we expect that some proportion of chum salmon injuries from explosives would likely result in fitness consequences, thus affecting the future survival and reproductive potential of the individual fish affected. As described in Section 8.2.3, the methodology used to quantify injury and mortality was based on several conservative assumptions which likely resulted in conservatively high estimates. Further, the estimated injury impacts to Hood Canal summer-run chum are likely conservatively high due to the Navy's

proposed location and timing of inland EOD events and the proposed mitigation measures discussed previously.

The two most recent status reviews (2011 and 2015) indicate some positive signs for the Hood Canal summer-run chum salmon ESU. Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; considered a good sign for viability in terms of spatial structure and diversity (Ford 2011b). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NWFSC 2015b). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (Ford 2011b). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years (NWFSC 2015b). Thus, while some proportion of chum salmon injuries from explosives would likely result in fitness consequences, the level of impacts anticipated would not appreciably affect the population abundance or trend of the Hood Canal summer-run chum salmon ESU at the population level.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of either the Hood Canal summer-run ESU or Columbia River ESU of chum salmon in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of either the Hood Canal summer-run ESU or Columbia River ESU of chum salmon in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Hood Canal summer-run ESU or Columbia River ESU of chum salmon.

10.10.1 Hood Canal Summer-run Chum Salmon ESU Critical Habitat

The nearshore marine habitat component of Hood Canal summer-run chum salmon critical habitat overlaps with the inland portion of the NWTT action area. We analyzed the effects of the proposed action on Hood Canal summer-run chum salmon critical habitat in Section 8.3.3.2 above. The identified estuarine and nearshore marine PBFs within the inland portion of the action area include the following: areas free of obstruction; water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh-and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic

invertebrates and fishes, supporting growth and maturation. There is no mechanism for the stressors associated with the proposed action to affect the water quantity or salinity PBFs. Many of the stressors resulting from NWTT activities would have no effect on water quality, salmonid natural cover, or salmonid forage within designated critical habitat. As discussed in Section 8.1.3.6 (*Stressors Resulting in Effects to Fish Habitat or Prey*), the secondary effects of stressors resulting from the proposed action on ESA-listed salmonids via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). The effects of explosives detonated at the Navy Hood Canal EOD site would not likley overlap with the nearshore marine habitat areas included in the Hood Canal summer-run chum salmon critical habitat designation. While seafloor devices could obstruct fish movements, considering the size of these devices, the mobility of salmonids to avoid or swim around them, and the temporary nature of deployment, any effects associated with obstruction from seafloor devices would likely be insignificant. Therefore, based on our critical habitat effects analysis presented in Section 8.3.3.2, we do not anticipate the effects of the proposed action would appreciably diminish the value of Hood Canal summer-run chum salmon designated critical habitat as a whole for the conservation of this species.

10.11 Coho Salmon

Within the action area, four ESUs of coho salmon may be exposed to explosives associated with NWTT activities throughout the year. These include the endangered Central California coast ESU and the threatened Lower Columbia River ESU, Oregon coast ESU, and Southern Oregon and Northern California coast ESU. Listing dates for each of these coho salmon DPSs are listed in Section 6.2. The main threats to coho salmon include blocked access to spawning grounds and habitat degradation caused by dams and culverts

As with other ESA-listed fishes in the action area, any coho salmon located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E4 and E7). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast. Adult and juvenile ESA-listed coho salmon will only occur in the offshore blast sites within the NWTT action area. The maximum annual number (based on a maximum year of explosive activity) of estimated injuries and mortalities numbers along with the proportion of coho salmon injured or killed from all four ESUs likely to be adversely affected by NWTT explosive activities are presented in Tables 171-173. Further, the average annual estimated injuries and mortalities over every seven years of NWTT training and testing activities are presented in Tables 174-176. These tables also detail

the proportion of ESA-listed coho salmon likely to be adversely affected by NWTT explosive activities[28].

Table 171. Percent of each ESA-listed coho salmon ESU (hatchery produced with adipose fin intact) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage[1]	ESU	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	0	4	654	0.0046%	0.5686%
	Juvenile		8	943	165,880	0.0046%	0.5686%
	Adult[1]	S. Oregon/N. California coast - T	2	194	21,868	0.0091%	0.8871%
	Juvenile		42	5,095	575,000	0.0073%	0.8861%
	Adult	Oregon coast - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Lower Columbia River - T	2	187	17,582	0.0114%	1.0636%
	Juvenile		22	2,657	249,784	0.0088%	1.0637%

[1] Includes mortality and injury estimates for natural and hatchery fish (intact and clipped numbers).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 172. Percent of ESA-listed coho salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult		-	-	-	-	-

[28] The percentage of population killed or injured assumes the status of the species does not change over the course of course of the seven year period.

Species	Life stage	ESU	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Juvenile	S. Oregon/N. California coast - T	14	1,772	200,000	0.0072%	0.8862%
	Adult	Oregon coast - T	0	10	1,118	0.0072%	0.8862%
	Juvenile		4	532	60,000	0.0072%	0.8862%
	Adult	Lower Columbia River - T	-	-	-	-	-
	Juvenile		632	77,508	7,287,647	0.0087%	1.0636%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 173. Percent of ESA-listed coho salmon ESU (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	0	22	3,864	0.0046%	0.5686%
	Juvenile		7	899	158,130	0.0046%	0.5686%
	Adult	S. Oregon/N. California coast - T	1	161	18,130	0.0055%	0.8880%
	Juvenile		146	17,844	2,013,593	0.0073%	0.8862%
	Adult	Oregon coast - T	14	1,672	188,640	0.0074%	0.8863%
	Juvenile		480	58,855	6,641,564	0.0041%	0.1478%
	Adult	Lower Columbia River - T	5	635	59,732	0.0050%	0.1825%
	Juvenile		57	7,035	661,468	0.0050%	0.1831%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 174. Average annual impacts every seven years to ESA-listed coho salmon ESUs (hatchery produced with adipose fin intact) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage[1]	ESU	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	-	2	654	-	0.2403%
	Juvenile		3	404	165,880	0.0020%	0.2436%
	Adult[1]	S. Oregon/N. California coast - T	1	83	21,868	0.0033%	0.3796%
	Juvenile		18	2,184	575,000	0.0031%	0.3798%
	Adult	Oregon coast - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Lower Columbia River - T	1	80	17,582	0.0041%	0.4558%
	Juvenile		9	1,139	249,784	0.0037%	0.4558%

[1] Includes mortality and injury estimates for natural and hatchery fish (intact and clipped numbers).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 175. Average annual impacts every seven years to coho salmon ESUs (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	S. Oregon/N. California coast - T	-	-	-	-	-
	Juvenile		43	5,317	200,000	0.0031%	0.3798%
	Adult	Oregon coast - T	-	30	1,118	-	0.3833%
	Juvenile		13	1,595	60,000	0.0031%	0.3798%
	Adult		-	-	-	-	-

Species	Life stage	ESU	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Juvenile	Lower Columbia River - T	1,897	232,524	7,287,647	0.0037%	0.4558%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 176. Average annual impacts every seven years to coho salmon ESUs (naturally produced) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -Indicates there are no fish at this lifestage and ESU/DPS that would be affected by mortality or injury.

Species	Life stage	ESU	Total		Number of fish (natural origin)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Coho salmon	Adult	Central California coast - E	-	9	3,864	0.0020%	0.2440%
	Juvenile		3	385	158,130	0.0020%	0.2437%
	Adult	S. Oregon/N. California coast - T	1	69	18,130	0.0032%	0.3798%
	Juvenile		62	7,647	2,013,593	0.0031%	0.3798%
	Adult	Oregon coast - T	6	716	188,640	0.0031%	0.3798%
	Juvenile		206	25,224	6,641,564	0.0031%	0.3798%
	Adult	Lower Columbia River - T	2	272	59,732	0.0038%	0.4558%
	Juvenile		25	3,015	661,468	0.0037%	0.4558%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

As shown in Tables 171-173, only a small annual percentage of each coho salmon ESU may be killed or injured by the Navy's NWTT explosive activities during a maximum year of explosives use. During the Navy's proposed annual activities, it is estimated that less than 1.1 percent of any coho salmon ESU would be exposed to injury and 0.0087 percent or less would be killed. Further, during seven years of the Navy's proposed testing and training activities it is estimated that less than 0.5 percent of hatchery and natural origin ESA-listed coho may be exposed to injury and less than 0.04 percent may be killed during the next seven years of NWTT activities.

It is worth noting that, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis may also be conservatively high due to several conservative assumptions in

our approach. The estimated number of coho salmon injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of salmonid fish injury from Navy explosives. Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. This is likely a conservative approach for adult and subadult coho salmon which, given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on. If injured, large adult and subadult coho would also likely recover faster from sublethal injuries, as compared to juveniles, with a lower likelihood of fitness consequences or long-term effects on their survival or future reproductive potential.

This level of mortality and injury anticipated represents a very small reduction in abundance that is not likely to appreciably reduce the likelihood of survival and recovery of any ESA-listed coho salmon ESU. Additionally, we conclude that the diversity of ESA-listed coho salmon ESUs will not be affected by this limited amount of mortality or injury because it is expected to be distributed across populations through the species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the NWTT action area would not appreciably reduce the likelihood of ESA-listed coho salmon surviving and recovering in the wild.

Some individual coho salmon may experience TTS as a result of Navy impulsive acoustic stressors. However, coho salmon lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in coho salmon migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because coho salmon are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt

normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of ESA-listed coho salmon ESUs in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of ESA-listed coho salmon ESUs in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in the survival rate or trajectory of recovery of the Central California coast, Lower Columbia River, Oregon coast, and Southern Oregon & Northern California coast ESUs of coho salmon.

10.12 Steelhead

Within the action area, ten DPSs of steelhead may be exposed to explosives associated with NWTT activities throughout the year. These include the threatened California Central Valley DPS, Central California coast DPS, Lower Columbia River DPS, Middle Columbia River, Northern California DPS, Puget Sound DPS, Snake River Basin DPS, South-Central California Coast DPS, Upper Columbia River DPS, and Upper Willamette River DPS. Listing dates for each of these steelhead DPSs are provided in Section 6.2. Primary threats to steelhead salmon include blocked access to spawning grounds, habitat degradation caused by dams and culverts, commercial fishing, and issues stemming from climate change (i.e., draught).

As with other ESA-listed fishes in the action area, any steelhead located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E8 and E11). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast. Adult and juvenile steelhead both occur in offshore and inland blast sites within the NWTT action area. While all ten steelhead DPSs are likely to be adversely affected by impacts from NWTT offshore explosive activities shown in Table 149, only steelhead from the Puget Sound DPS are likely to be adversely affected by inland explosives in Hood Canal and Crescent Harbor. The maximum annual (based on a maximum year of explosive use) estimated number of injuries and mortalities, along with the proportion of steelhead injured or killed using abundances from NMFS (2020c), from all ten steelhead DPSs likely to be adversely affected by NWTT explosive activities are presented in Tables 177-179. Further, the average annual estimated injuries and mortalities over every seven

years of NWTT training and testing activities are presented in Tables 180-182. These tables also detail the proportion of ESA-listed steelhead likely to be adversely affected by NWTT explosive activities[29].

Table 177. Percent of each steelhead DPS (hatchery produced with adipose fin intact) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	DPS	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Steelhead	Adult	South-Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	California Central Valley - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Northern California- T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Upper Columbia River - E	-	5	3,489	-	0.1433%
	Juvenile		2	157	138,601	0.0014%	0.1133%
	Adult	Snake River basin - T	-	51	48,411	-	0.1053%
	Juvenile		6	741	705,490	0.0009%	0.1050%
	Adult[1]	Lower Columbia River - T	-	70	66,891	-	0.1046%
	Juvenile		-	10	9,138	-	0.1094%
	Adult	Upper Willamette River - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Middle Columbia River - T	-	-	336	-	-
	Juvenile		1	120	110,469	0.0009%	0.1086%

[29] The percentage of population killed or injured assumes the status of the species does not change over the course of course of the seven year period.

Species	Life stage	DPS	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Adult	Puget Sound - T	-	-	-	-	-
	Juvenile		1	118	112,500	0.0009%	0.1049%

[1] Includes mortality and injury estimates for natural and hatchery fish (intact and clipped numbers).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 178. Percent of each steelhead DPS (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Steelhead	Adult	South-Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central California - T	0	12	11,598	0.0009%	0.1063%
	Juvenile		6	690	648,891	0.0009%	0.1063%
	Adult	California Central Valley - T	0	12	11,568	0.0009%	0.1063%
	Juvenile		14	1,702	1,600,653	0.0009%	0.1063%
	Adult	Northern California- T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Upper Columbia River - E	0	17	15,927	0.0009%	0.1063%
	Juvenile		6	731	687,567	0.0009%	0.1063%
	Adult	Snake River basin - T	2	254	238,530	0.0009%	0.1063%
	Juvenile		29	3,509	3,300,152	0.0009%	0.1063%
	Adult	Lower Columbia River - T	-	-	-	-	-
	Juvenile		10	1,273	1,197,156	0.0009%	0.1063%
	Adult	Upper Willamette River - T	-	-	-	-	-
	Juvenile		-	-	-	-	-

Species	Life stage	DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Adult	Middle Columbia River - T	0	1	1,344	0.0009%	0.1063%
	Juvenile		4	473	444,973	0.0009%	0.1063%
	Adult	Puget Sound - T	-	-	-	-	-
	Juvenile		1	117	110,000	0.0009%	0.1063%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 179. Percent of each steelhead DPS (naturally produced) that would be killed or injured in the offshore and inland portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (natural origin)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
-Steelhead	Adult	South-Central California - T	-	3	-	-	0.1439%
	Juvenile		-	99	79,057	-	0.1252%
	Adult	Central California - T	-	7	6,561	-	0.1067%
	Juvenile		2	271	248,771	0.0008%	0.1089%
	Adult	California Central Valley - T	-	5	5,058	-	0.0989%
	Juvenile		6	662	630,403	0.0010%	0.1050%
	Adult	Northern California- T	-	23	21,663	-	0.1062%
	Juvenile		8	877	821,389	0.0010%	0.1068%
	Adult	Upper Columbia River - E	-	8	5,793	-	0.1381%
	Juvenile		2	226	199,380	0.0010%	0.1134%
	Adult	Snake River basin - T	-	33	31,641	-	0.1043%
	Juvenile		6	839	798,341	0.0008%	0.1051%
	Adult	Lower Columbia River - T	-	40	38,760	-	0.1032%
	Juvenile		3	368	352,146	0.0009%	0.1045%
	Adult	Upper Willamette River - T	-	9	8,736	-	0.1030%
	Juvenile		2	165	140,396	0.0014%	0.1175%
	Adult		-	17	15,156	-	0.1122%

Species	Life stage	ESU/DPS	Total		Number of fish (natural origin)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Juvenile	Middle Columbia River - T	3	446	407,697	0.0007%	0.1094%
	Adult[1]	Puget Sound - T	2	100	26	0.0028%	0.1727%
	Juvenile		92	6,736	2,196,901	0.0042%	0.3066%

[1] Includes natural and hatchery fish (both clipped and intact adipose fins).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 180. Average annual impacts every seven years to ESA-listed steelhead DPSs (hatchery produced with adipose fin intact) killed or injured in the offshore and inland portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	DPS	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Steelhead	Adult	South-Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	California Central Valley - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Northern California- T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Upper Columbia River - E	-	4	3,489	-	0.1228%
	Juvenile		1	148	138,601	0.0009%	0.1071%
	Adult	Snake River basin - T	-	48	48,411	-	0.0989%
	Juvenile		6	698	705,490	0.0008%	0.0989%
	Adult[1]	Lower Columbia River - T	-	66	66,891	-	0.0985%
	Juvenile		-	9	9,138	-	0.0985%
	Adult		-	-	-	-	-

Species	Life stage	DPS	Total		Number of fish (intact adipose)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Juvenile	Upper Willamette River - T	-	-	-	-	-
	Adult	Middle Columbia River - T	-	-	336	-	-
	Juvenile		1	114	110,469	0.0009%	0.1029%
	Adult	Puget Sound - T	-	-	-	-	-
	Juvenile		1	111	112,500	0.0008%	0.0983%

[1] Includes mortality and injury estimates for natural and hatchery fish (intact and clipped numbers).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 181. Average annual impacts every seven years to ESA-listed steelhead DPSs (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Steelhead	Adult	South-Central California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Central California - T	-	12	11,598	-	0.0998%
	Juvenile		5	668	648,891	0.0008%	0.1030%
	Adult	California Central Valley - T	-	12	11,568	-	0.1013%
	Juvenile		13	1,584	1,600,653	0.0008%	0.0990%
	Adult	Northern California - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Upper Columbia River - E	-	19	15,927	-	0.1220%
	Juvenile		6	737	687,567	0.0009%	0.1072%
	Adult	Snake River basin - T	2	236	238,530	0.0008%	0.0989%
	Juvenile		27	3,263	3,300,152	0.0008%	0.0989%
	Adult		-	-	-	-	-

Species	Life stage	DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Juvenile	Lower Columbia River - T	10	1,178	1,197,156	0.0008%	0.0984%
	Adult	Upper Willamette River - T	-	-	-	-	-
	Juvenile		-	-	-	-	-
	Adult	Middle Columbia River - T	-	1	1,344	-	0.0957%
	Juvenile		4	458	444,973	0.0008%	0.1030%
	Adult	Puget Sound - T	-	-	-	-	-
	Juvenile		1	108	110,000	0.0008%	0.0983%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 182. Average annual impacts every seven years to ESA-listed steelhead DPSs (naturally produced) killed or injured in the offshore and inland portion of the NWTT action area in seven years. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (natural origin)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Steelhead	Adult	South-Central California - T	-	2	2,085	-	0.1165%
	Juvenile		1	95	79,057	0.0011%	0.1200%
	Adult	Central California - T	-	7	6,561	-	0.1002%
	Juvenile		2	256	248,771	0.0009%	0.1030%
	Adult	California Central Valley - T	-	5	5,058	-	0.1017%
	Juvenile		5	624	630,403	0.0008%	0.0990%
	Adult	Northern California- T	-	22	21,663	-	0.1002%
	Juvenile		7	827	821,389	0.0008%	0.1006%
	Adult	Upper Columbia River - E	-	7	5,793	-	0.1233%
	Juvenile		2	214	199,380	0.0009%	0.1072%
	Adult	Snake River basin - T	-	31	31,641	-	0.0989%
	Juvenile		7	789	798,341	0.0008%	0.0989%

Species	Life stage	ESU/DPS	Total		Number of fish (natural origin)[2]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
	Adult	Lower Columbia River - T	-	38	38,760	-	0.0984%
	Juvenile		3	347	352,146	0.0008%	0.0984%
	Adult	Upper Willamette River - T	-	9	8,736	-	0.1014%
	Juvenile		1	156	140,396	0.0009%	0.1113%
	Adult	Middle Columbia River - T	-	16	15,156	-	0.1046%
	Juvenile		4	420	407,697	0.0009%	0.1031%
	Adult[1]	Puget Sound - T	1	96	57,939	0.0020%	0.1657%
	Juvenile		90	6,600	2,196,901	0.0041%	0.3004%

[1] Includes natural and hatchery fish (both clipped and intact adipose fins).

[2] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

As shown in Tables 177-179, only a small annual percentage (less than 0.32 percent) of each steelhead DPS would be killed and/or injured by the Navy's annual NWTT explosive activities during a maximum year of explosives use. Further, as shown in Tables 180-182, only a small percentage (less than 0.31 percent) of each steelhead DPS is estimated to be killed and/or injured by the Navy's NWTT explosive activities for the next seven years. The DPS with the largest percentage of injury or mortality is the Puget Sound DPS as a result of it being the only steelhead DPS that would incur effects from blasts from both offshore and inland explosive activities. Other steelhead DPSs would be affected at much lower levels.

It is worth noting that, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis may also be conservatively high due to several conservative assumptions in our approach. The estimated number of steelhead injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of salmonid fish injury from Navy explosives. Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. This is likely a conservative approach for adult and subadult steelhead which, given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on. If injured, large adult and subadult steelhead would also likely

recover faster from sublethal injuries, as compared to juveniles, with a lower likelihood of fitness consequences or long-term effects on their survival or future reproductive potential.

The anticipated level of mortality and injury represents a very small reduction in abundance that is not likely to appreciably reduce the likelihood of survival and recovery of any ESA-listed steelhead. Additionally, we conclude that the diversity of ESA-listed steelhead populations will not be affected by this limited amount of take because it is expected to be distributed across populations through species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the NWTT action area would not appreciably reduce the likelihood of ESA-listed Pacific steelhead surviving and recovering in the wild.

Some individual steelhead may experience TTS as a result of Navy impulsive acoustic stressors. However, the steelhead lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in steelhead migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because these species are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects on steelhead resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of ESA-listed Pacific steelhead DPSs in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of ESA-listed Pacific steelhead DPSs in the wild by reducing the reproduction, numbers, or

distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the California Central Valley DPS, Central California coast DPS, Lower Columbia River DPS, Middle Columbia River, Northern California DPS, Puget Sound DPS, Snake River Basin DPS, South-Central California Coast DPS, Upper Columbia River DPS, and Upper Willamette River DPS of steelhead.

10.13 Sockeye Salmon

Within the action area, the endangered Snake River ESU of sockeye salmon may be exposed to explosives associated with NWTT activities throughout the year. The listing date for this sockeye salmon ESU is provided in Section 6.2. Threats to sockeye salmon include habitat impediments (dams), habitat degradation, habitat loss, commercial and recreational fishing, and impacts from climate change including drought.

As with other ESA-listed fishes in the action area, any sockeye salmon located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E4 and E7). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast. Adult and juvenile sockeye salmon only occur in offshore blast sites within the NWTT action area. The maximum annual number of estimated injuries and mortalities along with the proportion of sockeye salmon injured or killed using abundances from NMFS (2020c) from the Snake River ESU are presented in Tables 183-184. Further, the average annual estimated injuries and mortalities over every seven years of NWTT training and testing activities are presented in Tables 185-186. These tables also detail the proportion of Snake River ESU sockeye salmon likely to be adversely affected by NWTT explosive activities.

Table 183. Percent of each sockeye salmon ESU (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, - indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Sockeye salmon	Adult	Snake River - E	-	1	12,012	-	0.0121%
	Juvenile		-	29	242,610	-	0.0121%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 184. Percent of each sockeye salmon ESU (naturally produced) that would be killed or injured in the offshore portion of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (natural)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Sockeye salmon	Adult	Snake River - E	-	-	1,638	-	-
	Juvenile		-	2	19,181	-	0.0104%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 185. Average annual impacts every seven years to ESA-listed sockeye salmon (hatchery produced with adipose fin-clip) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (clipped adipose)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Sockeye salmon	Adult	Snake River - E	-	1	12,012	0.0000%	0.0048%
	Juvenile		0	14	242,610	0.0001%	0.0058%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

Table 186. Average annual impacts every seven years to ESA-listed sockeye salmon (naturally produced) killed or injured in the offshore portion of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected by mortality or injury.

Species	Life stage	ESU/DPS	Total		Number of fish (natural)[1]	Percentage of Population Killed	Percentage of Population Injured
			Mortality	Injury			
Sockeye salmon	Adult	Snake River - E	-	-	1,638	-	-
	Juvenile		-	1	19,181	-	0.0060%

[1] Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each ESU by the total number of spawners.

As shown in Tables 183-184, only a small annual percentage (less than 0.02 percent) of Snake River Sockeye Salmon may be killed or injured by the Navy's annual NWTT explosive activities during a maximum year of explosives use. Further, as shown in Tables 185-186, only a small

percentage (less than 0.0087 percent) of Snake River sockeye salmon is estimated to be killed and/or injured by the Navy's NWTT explosive activities for the next seven years.

It is worth noting that, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis may also be conservatively high due to several conservative assumptions in our approach. The estimated number of sockeye salmon injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of salmonid fish injury from Navy explosives. Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria for the most sensitive fish species and life stages. This is likely a conservative approach for adult and subadult sockeye salmon which, given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on. If injured, large adult and subadult sockeye salmon would also likely recover faster from sublethal injuries, as compared to juveniles, with a lower likelihood of fitness consequences or long-term effects on their survival or future reproductive potential.

The anticipated level of mortality and injury represents a very small reduction in abundance that is not likely to appreciably reduce the likelihood of survival and recovery of any ESA-listed sockeye salmon. Additionally, we conclude that the diversity of ESA-listed sockeye salmon populations will not be affected by this limited amount of take because it is expected to be distributed across populations through species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the NWTT action area would not appreciably reduce the likelihood of ESA-listed sockeye salmon surviving and recovering in the wild.

Some individual sockeye salmon may experience TTS as a result of Navy impulsive acoustic stressors. However, sockeye salmon lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in sockeye salmon migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because these species are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects on sockeye salmon resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the Snake River ESU of sockeye salmon in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Snake River ESU of sockeye salmon in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Snake River ESU of sockeye salmon.

10.14 Eulachon

Within the action area, Southern DPS eulachon may be exposed to explosives associated with NWTT activities throughout the year. Southern DPS eulachon was listed as threatened in October 20, 2011. The primary threats facing Southern DPS eulachon include habitat degradation, habitat impediments, water pollution, and fisheries interaction.

As with other ESA-listed fishes in the action area, any eulachon located within the blast radius of an explosion could be injured or killed, and sustain some degree of TTS or exhibit behavioral disruptions. Severity of injury would likely increase closer to the blast, where mortality is more probable within a close distance of the blast radius of larger bin sizes described in Section 8.2.3. (e.g., E4 and E7). In addition to bin size, factors such as the depth at which the explosive detonates also influences the total size of the blast.

In the NWTT action area, eulachon would primarily occur in offshore blast sites. In the waters of Puget Sound, eulachon have low abundance numbers relative to coastal waters, and typically occupy very deep waters (Donnelly and Burr 1995) outside of the Hood Canal and Crescent Harbor EOD range sites. As a result, the Navy's NWTT explosive activities were determined to likely adversely affect eulachon in the offshore portion of the action area whereas inland explosive activities were determined not likely to adversely affect eulachon. The maximum annual number of estimated injuries and mortalities, along with the proportion of Southern DPS

eulachon injured or killed using abundances from NMFS (2020c), are presented in Table 187. The average annual estimated injuries and mortalities of Southern DPS eulachon over every seven years of proposed NWTT activities are shown in Table 188.

Table 187. Percent of each Southern DPS eulachon that would be killed or injured in the offshore of the NWTT action area in a maximum year of explosives activity level. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected.

Species	DPS	Total		Number of fish	Percentage of Population Killed	Percentage of Population Injured
		Mortality	Injury			
Eulachon	Southern	341	2,520	75,184,360	0.0005%	0.00334%

[1]Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

Table 188. Average annual impacts every seven years to Southern DPS eulachon that would be killed or injured in the offshore of the NWTT action area. Unless noted otherwise, -indicates there are no fish at this lifestage and DPS that would be affected.

Species	DPS	Total		Number of fish	Percentage of Population Killed	Percentage of Population Injured
		Mortality	Injury			
Eulachon	Southern	147	1,080	75,184,360	0.00006%	0.0014%

[1]Derived from abundances presented in Table 11. Applied correction factor for adults that multiplied the average total number of years at sea for each DPS by the total number of spawners.

As shown in Table 187 and Table 188, only an extremely small annual percentage (less than 0.004 percent for maximum annual and less than 0.0015 percent for seven years of activity) of the Southern DPS eulachon may be killed or injured by the Navy's NWTT explosive activities. This level of mortality and injury represents a an extremely small reduction in abundance that is not likely to appreciably reduce the likelihood of survival and recovery Southern DPS eulachon. It is also worth noting that, as described in Section 8.2.3, the methodology used to quantify injury and mortality was conservative. Additionally, we conclude that the diversity of ESA-listed eulachon populations will not be affected by this limited amount of take because it is expected to be distributed across populations through species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the NWTT action area would not appreciably reduce the likelihood of ESA-listed eulachon surviving and recovering in the wild.

Some individual eulachon may experience TTS as a result of Navy impulsive acoustic stressors. However, eulachon lack notable hearing specialization, minimizing the likelihood of each

instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014a). Additionally, hearing is not thought to play a role in eulachon migration (e.g., Putnam et al. 2013). TTS is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006). Because these species are able to rely on alternative mechanisms for essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects on eulachon resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival and recovery of Southern DPS eulachon in the wild by reducing the reproduction, numbers, or distribution of the species or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Southern DPS eulachon in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Southern DPS eulachon.

10.15 Bocaccio – Puget Sound / Georgia Basin DPS

The bocaccio Puget Sound/Georgia Basin DPS was listed as endangered on April 28, 2010 (75 FR 22276). Historical overfishing has been recognized as the primary cause of the decline of ESA-listed rockfishes in Puget Sound (Drake et al. 2010; Palsson et al. 2009). As a long-lived, slow growing, and late maturing species bocaccio are particularly vulnerable to the effects of overfishing. Despite regulatory controls on fishing in recent years, past harvesting practices that targeted the largest and oldest individuals have likely hindered the capability of recovering population numbers for years to come (Love et al. 1998b). The development of nearshore areas also likely continues to degrade the quality of nearshore rearing habitats for juvenile bocaccio

(NMFS 2017h). Ongoing potential threats to adult bocaccio deepwater habitat include derelict fishing gear, dredging and sediment disposal, alternative energy structures, and cable laying (NMFS 2017h).

The rate of decline for rockfish in Puget Sound has been estimated at 3.1 to 3.8 percent annually for the period 1977 to 2014 (NMFS 2016m). This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climatic/oceanographic conditions (MacCall and He 2002). There is no current population abundance estimate for the Puget Sound/Georgia Basin DPS of bocaccio across the full DPS area. A survey conducted in 2008 produced an estimate of 4,606 adult bocaccio in the San Juan Island area (NMFS 2016m).

Based on our *Effects Analysis* (Section 8.2.3.1), we anticipate a relatively small number of juvenile and larval bocaccio would likely be exposed to NWTT explosives in inland waters at levels resulting in a range of adverse effects including mortality, injury, and TTS. Considering the relatively small areas within Puget Sound that would be affected by Navy explosives (i.e., Hood Canal and Crescent Harbor EOD range sites), as compared to the much larger area representing the range of this DPS, and the small number of explosive events proposed annually, only a very small proportion of larval and juvenile Puget Sound/Georgia Basin DPS bocaccio would likely be exposed. Larval rockfish have very high natural mortality rates and recruitment success is highly dependent on environmental conditions (NMFS 2014a). Poor larval survival in most years is balanced by the long lives of reproductive adults. For a long-lived, late maturing species with high natural mortality in early life stages, the loss of a very small number of individual larvae and juveniles is not anticipated to have a measureable effect on adult population abundance. Adult bocaccio, which prefer deep water areas, would not likely be affected by Navy underwater explosions within the shallower Hood Canal and Crescent Harbor EOD range sites. The effects of all other potential stressors (i.e., acoustic, energy, physical disturbance and strike, entanglement, and ingestion) on Puget Sound/Georgia Basin DPS bocaccio analyzed in this opinion were found to be either discountable or insignificant.

In summary, we anticipate an extremely small number of individual larval and juvenile Puget Sound/Georgia Basin DPS bocaccio, relative to the total population size of these life stages, would be killed or injured as a result of the proposed action. The loss (or serious injury) of these individuals is not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the Puget Sound/Georgia Basin bocaccio DPS in the wild by reducing the reproduction, numbers, or distribution of the DPS. We also conclude that effects from ongoing Navy training and testing activities continuing into the

reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Puget Sound/Georgia Basin bocaccio DPS in the wild by reducing the reproduction, numbers, or distribution of this DPS. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Puget Sound/Georgia Basin bocaccio DPS.

10.15.1 Puget Sound / Georgia Basin DPS Designated Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS of bocaccio was finalized in 2014 (79 FR 68041). In Section 8.3.3.1 we evaluated the effects of the proposed action on the following essential physical and biological features (PBFs) of Puget Sound/Georgia Basin DPS bocaccio designated critical habitat within the action area: 1) Quantity, quality, and availability of prey species to support individual growth, survival, and feeding opportunities; 2) Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities; and 3) For deepwater habitat, the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance. The secondary effects of stressors resulting from the proposed action on ESA-listed rockfish via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage).

The Navy's use of explosives at the Crescent Harbor EOD Range could impact nearshore (less than 30 meter depths) critical habitat designated for juvenile bocaccio by temporarily reducing the abundance of fish prey species near the detonation point. Although this EOD site was excluded from critical habitat designation, sound from explosives during Navy mine neutralization training events at Crescent Harbor could overlap ESA-listed rockfish designated critical habitat in the inland waters, as this training range is located outside of but adjacent to critical habitat (Navy 2020e). If this were to occur, the effects would likely be short-term and temporary, only occurring during the few proposed events involving explosives in inland waters, with no lasting effect on prey availability or the pelagic food web expected.

Precision anchoring could affect the critical habitat PBF associated with structure and rugosity in deepwater habitats occupied by adult bocaccio. However, impacts from precision anchoring would likely by highly localized, mainly restricted to the same few designated anchorages that have been used by the Navy for many years. The total bottom area affected by precision anchoring would be extremely minimal compared to the total area of deepwater critical habitat designated for this species.

Therefore, based on our critical habitat effects analysis presented in Section 8.3.3.1 , we do not anticipate the effects of the proposed action would appreciably diminish the value of Puget Sound/Georgia Basin DPS bocaccio designated critical habitat as a whole for the conservation of this species.

10.16 Yelloweye Rockfish – Puget Sound / Georgia Basin DPS

The Puget Sound/Georgia Basin DPS of yelloweye rockfish was listed under the ESA as threatened on April 28, 2010 (75 FR 22276). Historical overfishing has been recognized as the primary cause of the decline of ESA-listed rockfishes in Puget Sound (Drake et al. 2010; Palsson et al. 2009). As a long-lived, slow growing, and late maturing species, yelloweye rockfish are particularly vulnerable to the effects of overfishing. Despite regulatory controls on fishing in recent years, past harvesting practices that targeted the largest and oldest individuals have likely hindered the capability of recovering population numbers for years to come (Love et al. 1998b). Ongoing potential threats to juvenile and adult yelloweye rockfish deepwater habitat in Puget Sound include derelict fishing gear, dredging and sediment disposal, alternative energy structures, and cable laying (NMFS 2017h).

The rate of decline for rockfish in Puget Sound has been estimated at 3.1 to 3.8 percent annually for the period 1977 to 2014 (NMFS 2016m). This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climatic/oceanographic conditions (MacCall and He 2002). Based on the best available information, the current population abundance estimate for the Puget Sound/Georgia Basin DPS of yelloweye rockfish is nearly 67,000 adults (WDFW 2017).

Based on our *Effects Analysis* (Section 8.2.3.1), we anticipate a relatively small number of larval yelloweye rockfish would likely be exposed to NWTT explosives in inland waters at levels resulting in a range of adverse effects including mortality, injury, and TTS. Considering the relatively small areas within Puget Sound that would be affected by Navy explosives (i.e., Hood Canal and Crescent Harbor EOD range sites), as compared to the much larger area representing the range of this DPS, and the small number of explosive events proposed annually, only a very small proportion of larval Puget Sound/Georgia Basin DPS yelloweye rockfish would likely be exposed. Larval rockfish have very high natural mortality rates and recruitment success is highly dependent on environmental conditions (NMFS 2014a). Poor larval survival in most years is balanced by the long lives of reproductive adults. For a long-lived, late maturing species with high natural mortality in early life stages, the loss of a very small number of individual larvae and juveniles is not anticipated to have a measureable effect on adult population abundance. Adult and juvenile yelloweye rockfish, which prefer deep water areas, would not likely be affected by Navy underwater explosions within the shallower Hood Canal and Crescent Harbor EOD range sites. The effects of all other potential stressors (i.e., acoustic, energy, physical disturbance and strike, entanglement, and ingestion) on Puget Sound/Georgia Basin DPS yelloweye rockfish analyzed in this opinion were found to be either discountable or insignificant.

In summary, we anticipate an extremely small number of individual larval Puget Sound/Georgia Basin DPS yelloweye rockfish, relative to the total population size of this life stage, would be killed or injured as a result of the proposed action. The loss (or serious injury) of these

individuals is not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. Based on the evidence available, including the *Environmental Baseline, Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the Puget Sound/Georgia Basin yelloweye rockfish DPS in the wild by reducing the reproduction, numbers, or distribution of the DPS. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Puget Sound/Georgia Basin yelloweye rockfish DPS in the wild by reducing the reproduction, numbers, or distribution of this DPS. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Puget Sound/Georgia Basin yelloweye rockfish DPS.

10.16.1 Puget Sound / Georgia Basin DPS Designated Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS of yelloweye rockfish was finalized in 2014 (79 FR 68041). In Section 8.3.3.1 we evaluated the effects of the proposed action on the following essential physical and biological features of Puget Sound/Georgia Basin DPS yelloweye rockfish designated critical habitat within the action area: 1) Quantity, quality, and availability of prey species to support individual growth, survival, and feeding opportunities; 2) Water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities; and 3) For deepwater habitat, the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance. The secondary effects of stressors resulting from the proposed action on ESA-listed rockfish via impacts to their habitat or prey were all found to be either insignificant or discountable. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage).

Precision anchoring could affect the critical habitat PBF associated with structure and rugosity in deepwater habitats occupied by adult and juvenile yelloweye rockfish. However, impacts from precision anchoring would likely by highly localized, mainly restricted to the same few designated anchorages that have been used by the Navy for many years. The total bottom area affected by precision anchoring would be extremely minimal compared to the total area of deepwater critical habitat designated for this species.

Therefore, based on our critical habitat effects analysis presented in Section 8.3.3.1 , we do not anticipate the effects of the proposed action would appreciably diminish the value of Puget Sound/Georgia Basin DPS yelloweye rockfish designated critical habitat as a whole for the conservation of this species.

10.17 Green Sturgeon – Southern DPS

The Southern DPS of green sturgeon was listed as a threatened species under the ESA in 2006 (71 FR 17757). The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River caused primarily by impoundments. Green sturgeon also face threats related to water temperature, water flow, and from commercial and recreational fishing bycatch. Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015e).

Based on the best available information, the current population abundance estimate for the Southern DPS green sturgeon is 4,387 juveniles, 11,055 subadults, and 2,106 adults (Mora et al. 2018). No estimate of intrinsic growth rates are available for Southern DPS green sturgeon. Attempts to evaluate the status of Southern DPS green sturgeon have been met with limited success due to the lack of reliable long-term data.

With the exception of explosives, we found that the effects all other potential stressors (i.e., acoustic, energy, physical disturbance and strike, entanglement, and ingestion) analyzed in this opinion on Southern DPS green sturgeon were either discountable or insignificant (see Section 8.1.3). From our fish exposure analysis (Section 8.2.3), based on the best available information we conservatively estimate the following annual impacts on subadult/adult Southern DPS green sturgeon resulting from exposure to the proposed NWTT Phase III offshore explosives: up to 687.0 injuries and 5.7 mortalities based on a seven-year average annual explosive activity level; and up to 1,602.9 injuries and 13.2 mortalities based on a maximum year of explosive activity level. Although we did not quantify, some proportion of the population would also likely experience TTS and behavioral responses as a result offshore explosive detonations. The potential effects from the Navy's use of explosives in inland waters on Southern DPS green sturgeon were found to be discountable (Section 8.2.3.2).

It is worth noting that, as described in Section 8.2.3, the methodology used to quantify fish injury and mortality was conservative. Thus, estimated impacts from NWTT explosives from our quantitative analysis may also be conservatively high due to several conservative assumptions in our approach. The estimated number of Southern DPS green sturgeon injuries is likely a conservatively high estimate since our range to effects for injury are based on the more conservative acoustic injury criteria of > 207 dB peak re 1 µPa for fishes with a swim bladder (versus the Navy's proposed 220 dB peak re 1 µPa). The ranges to effects used in our analysis (i.e., based on 207 dB peak re 1 µPa) are also a conservative approach since the criterion they are based on is expressed in Popper et al. (2014a) as "greater than" the 207 dB threshold. These factors likely result in conservatively high estimates of Southern DPS green sturgeon injury from Navy explosives. Further, the ranges to effects used in our effects analysis are based on a zone of impact that would encompass the distance it would take for the sound wave to reach the criteria

for the most sensitive fish species and life stages. This is likely a conservative approach for subadult and adult green sturgeon which, given their large size, would likely be less sensitive to the effects of explosives than the smaller species and life stages these criteria were based on.

As noted, since we do not know the particular location or time of year of this proposed activity, the estimates of Southern DPS green sturgeon mortality and injury are based on the following conservative assumptions: 1) the proposed offshore mine countermeasure and neutralization events (two per year) would overlap in both time and space with the maximum monthly densities of Southern DPS green sturgeon in the action area, and 2) the maximum number of proposed explosives for this activity, including the maximum number of proposed explosives for the October through June seasonal time frame, would be used (i.e. both annual maximum and seven year total maximum). We anticipate that the estimates based on our quantitative analysis would be somewhat lower if some portion of detonations occurred farther offshore (i.e., in water depths greater than 80 meters) or during a time frame when estimated green sturgeon densities in the Quinault Range would be lower than those estimated above (Section 2.4.2.1).

In addition, the Navy has agreed to implement the following conservation measure: *The Navy will, to the maximum extent practicable, conduct mine countermeasure and neutralization testing from July 1 through September 30 when operating within 20 nautical miles from shore in the Marine Species Coastal Mitigation Area*. As discussed above, densities of Southern DPS green sturgeon are anticipated to be significantly lower from July through September as compared to other months when this activity would likely occur (e.g., five times lower based on our density estimation approach). Although it may not always be practicable for the Navy to conduct both mine countermeasure neutralization explosive events each year from July through September, this conservation measure would likely result in lower levels of Southern DPS green sturgeon injury and mortality than those estimated by our quantitative analysis.

As a slow growing, late maturing, and long-lived species, green sturgeon may be particularly susceptible to anthropogenic threats and less resilient to additional environmental perturbations (Kindsvater et al. 2016). A population model developed by Beamesderfer et al. (2007) highlighted the extreme sensitivity of green sturgeon populations to even small incremental increases in mortality where multiple ages are affected. The mortality of any individual fish from a population represents the loss of 100 percent of that fish's reproductive potential. Given their large body size and high fecundity, mortality of adult sturgeon can result in negative population levels impacts. For long-lived species, such as green sturgeon, mortality of subadults affects future reproductive potential and could have effects on a population's growth rate for decades (Heppell 2007). Because their natural mortality rate is so low, even small increases in the annual mortality rate of adults greatly decreases mean lifespan and, under density- independent conditions, lifetime egg production (Heppell 2007). Therefore, it is important to analyze the impact of subadult and adult anthropogenic mortality on the viability of sturgeon populations.

With an estimated subadult/adult population size of 13,161 (Mora et al. 2018), based on our conservative average annual mortality estimate (5.7 fish per year), less than 0.05 percent of the Southern DPS green sturgeon population could be killed on an annual basis as a result of the proposed action. Empirical studies evaluating the effects of varying levels of anthropogenic mortality on sturgeon populations are lacking. Previous model-based studies have used basic principles of fishery science to approximate tolerable mortality rates to ensure survival and/or recovery based on sturgeon life history parameters including female fecundity, age and growth, and natural mortality rate. Although these studies were addressing the effects of sturgeon bycatch in commercial fisheries, the basic concepts used to model a targeted mortality rate still apply to mortality from the proposed action (NMFS 2017b). Assuming a population is composed of equal numbers of males and females, each egg produced would need to survive and produce two viable eggs just to keep the population trend at zero (no increase or decrease). Goodyear (1993) suggests that to maintain a steady population at least 20 percent of the population must be of spawning age; to promote population growth and recovery, a more conservative estimate of 50 percent of the population must be of spawning age (Boreman 1984). Boreman (1997) evaluated the total lifetime potential egg production of an individual female Atlantic sturgeon from the Hudson River to determine the loss of potential fecundity caused by Atlantic sturgeon bycatch in commercial fisheries. Based on the approximate spawning population thresholds above (i.e., 20 percent for stability, 50 percent for recovery), Boreman estimated that a five percent fishing mortality level or lower would allow for population growth and recovery. This analysis assumes there are no other significant sources of mortality besides natural mortality and fishing mortality. Based on a yield and egg per recruit model, Kahnle et al. (1998) estimated that a commercial bycatch fishing mortality rate of three percent would keep the Hudson River Atlantic sturgeon population stable or increasing. As a caveat, both of these analyses were based on the Hudson River Atlantic sturgeon population; although we use this population as a surrogate, the information and assumptions used to develop these models are not necessarily valid for Southern DPS green sturgeon. In addition, these models were based on the limited sturgeon data available in the late 1990's; model outputs may differ if updated with more recent available data. Despite the identified caveats and shortcomings, these studies still provide us with some indication of the level of anthropogenic mortality sturgeon populations, in general, could endure while still allowing for population growth and recovery (NMFS 2017b).

Another approach for evaluating the relative impact of anthropogenic stressors on fish populations is to compare anthropogenic mortality with estimated natural mortality. The longevity of sturgeon is clearly associated with low natural mortality rates beyond the first few years of age (Beamesderfer et al. 2007). For their population model of Southern DPS green sturgeon, Beamesderfer et al. (2007) assumed an annual natural mortality rate of seven percent.

Mortality of subadult and adult Southern DPS green sturgeon resulting from the proposed action would likely have adverse effects on this threatened population. From above, we conservatively estimate an annual mortality rate of less than 0.05 percent as a result of the proposed action. This

rate is about two orders of magnitude lower than modeled target mortality levels (i.e., three to five percent) to allow for growth and recovery of the Hudson River Atlantic sturgeon population (Boreman 1997; Kahnle et al. 1998). The estimated annual mortality rate from the proposed action (less than 0.05 percent) is also more than two orders of magnitude lower than the estimated annual natural mortality rate (seven percent) for Southern DPS green sturgeon (Beamesderfer et al. 2007). In addition, given the conservative assumptions used in our effects analysis (discussed above), the 0.05 percent annual mortality rate is likely an upper limit on the Southern DPS green sturgeon average annual mortality rate that would be expected as a result of the proposed action. In summary, based on our risk analysis, we determine that Southern DPS green sturgeon mortality resulting from the proposed action would not reach levels that significantly affect the viability of this ESA-listed population. This conclusion is based on the following: 1) the estimated annual mortality rate is significantly below target mortality rates for sturgeon population growth and recovery as reported in the literature, and significantly below the estimated Southern DPS green sturgeon natural mortality rate, and 2) the estimated average annual mortality rate of 0.05 percent likely represents an upper limit based on several conservative assumptions in our effects analysis.

In addition to lethal effects, the proposed action is likely to result in sublethal effects on Southern DPS green sturgeon including behavioral responses, TTS, and sublethal injuries. As noted above (Section 8.2.3.2), because green sturgeon are not known to rely on hearing for essential life functions, and any effects from TTS would likely be short-term and temporary, and instances of TTS would not likely result in measurable effects on any individual's fitness. Similarly, behavioral effects on green sturgeon resulting from reactions to sound created by Navy explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. Some proportion of sub-lethal injuries from NWTT explosives would likely result in fitness consequences for individual green sturgeon exposed. With an estimated subadult/adult population size of 13,161 (Mora et al. 2018), based on our conservative average annual injury estimate (687 injuries per year), around five percent of the Southern DPS green sturgeon population could be injured on an annual basis as a result of the proposed action.

As discussed above, our estimated injury rate likely represents an upper limit based on several conservative assumptions in our quantitative effects analysis, and considering the additional mitigation measures in place to limit mine countermeasure explosives activity during months when green sturgeon densities are expected to be highest. In addition, considering their size, longevity and low rate of natural mortality, we would expect most subadult and adult green sturgeon to recover from sublethal injuries with little or no long-term effect on their survival or future reproductive potential. For a detailed discussion of the anticipated response of fish to injuries from explosives see Section 8.2.3.2.

In summary, we anticipate Southern DPS green sturgeon subadults and adults would be adversely affected as a result of the proposed action, with the likely effects including mortality,

sub-lethal injury, temporary hearing loss, and behavioral harassment. While the loss (or serious injury) of individuals would likely have adverse effects on this threatened population, the population level impacts are not anticipated to result in appreciable reductions in overall reproduction, numbers, or distribution of this species. In addition, the Navy has proposed conservation measures that would likely minimize the anticipated impacts of the proposed action on Southern DPS green sturgeon. In particular, the Navy has proposed, to the maximum extent practicable, to conduct mine countermeasure neutralization testing events with explosives during the time of year (i.e., July through September) when Southern DPS green sturgeon densities in the offshore portion of the action area are expected to lowest. Based on the evidence available, including the *Environmental Baseline*, *Effects of the Action* and *Cumulative Effects*, effects resulting from stressors caused by training and testing activities the Navy will conduct in the NWTT action area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the Southern DPS green sturgeon in the wild by reducing the reproduction, numbers, or distribution of the DPS. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Southern DPS green sturgeon in the wild by reducing the reproduction, numbers, or distribution of this DPS. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the Southern DPS green sturgeon.

10.17.1 Southern DPS Designated Critical Habitat

Critical habitat for the Southern DPS green sturgeon was finalized in 2009 (74 FR 52300). Offshore coastal marine areas and the Strait of Juan de Fuca co-occur with portions of the NWTT action area. Anthropogenic activities that threaten Southern DPS green sturgeon critical habitat include the application of pesticides, commercial shipping, industrial and residential activities generating point source pollution and non-point source pollution, the disposal of dredged materials, and bottom trawl fisheries that disturb the bottom.

In Section 8.3.3.3 we evaluated the effects of the proposed action on the following essential physical and biological features of estuarine and coastal marine Southern DPS green sturgeon designated critical habitat within the action area: food resources, migratory corridor, water quality (estuarine and coastal areas); sediment quality, water flow, and water depth (estuarine areas only). We found that, with the exception of explosives, none of the stressors associated with the proposed action would likely reduce the conservation value of the PBFs of Southern DPS green sturgeon critical habitat. Most of the proposed NWTT activities involving explosives would occur outside of areas designated as green sturgeon critical habitat (i.e., either farther offshore or within inland EOD range sites in Puget Sound). However, the Navy has proposed a new offshore mine countermeasure and neutralization testing activity that could overlap with Southern DPS green sturgeon critical habitat within the Quinault Range Site, where this activity would typically occur greater than three nautical miles from shore. The Navy has proposed two

such mine countermeasure and neutralization explosive testing events per year in the offshore area, with each event lasting up to 10 days in duration.

The migratory corridor PBF is defined as a migratory pathway necessary for the safe and timely passage of Southern DPS green sturgeon within marine and between estuarine and marine habitats (74 FR 52300). The Navy's proposed use of explosives in coastal marine critical habitat could overlap with green sturgeon migratory routes. Large explosives (e.g. bin E7), in particular, could result in behavioral responses which may temporarily disrupt green sturgeon migrations. Since explosions would be brief and occur sporadically over the course of an event, exposed green sturgeon would likely resume their normal migratory behavior shortly after the explosion occurs. Although explosives could result in behavioral effects within a radius extending over several thousand meters, as discussed above, the area impacted would still be relatively small compared to the total marine area designated as critical habitat within the green sturgeon's migratory corridor. While explosives may affect the safety and timeliness of green sturgeon coastal migrations, the magnitude of such effects would not likely compromise the survival or overall viability of the species. Seasonal mitigation proposed by the Navy, that limits the number of explosives that could be used in months when green sturgeon densities are expected to be highest, would likely further reduce the impact of explosives on the migratory corridor PBF. Therefore, the effects of explosives used in mine countermeasure and neutralization testing on the migratory corridor PBF of Southern DPS green sturgeon critical habitat are likely insignificant.

Underwater explosions may reduce available prey items by either directly killing or injuring green sturgeon benthic prey or by impacting the structure and complexity of the benthic environment where such prey occur. If green sturgeon prey items are killed or injured within critical habitat, it is likely that only a low number of individuals and therefore a small portion of benthic prey species populations would be affected. Although some prey items could be exposed within the described mortality ranges during an explosive activity, other prey items would likely be available to green sturgeon in the immediate (unaffected) area surrounding the activity. Given the small number of events and relatively short duration of the activity, the immediate impact of explosives on prey availability within critical habitat would likely be short-term. In addition, the Navy has proposed mitigation to limit the number of explosives used for mine countermeasure neutralization testing events during the months when peak densities of Southern DPS green sturgeon would be expected in the action area.

A longer term impact from this stressor could occur from the effects of explosives on the structure and complexity of the benthic environment that green sturgeon prey inhabit. Compared to pelagic prey species which would be more likely to repopulate an area shortly after an explosive activity is completed, benthic prey may be less likely to recolonize seafloor habitat areas that have been physically damaged by the blast of explosive detonations. Explosive mines (bin E7) associated with this activity would be moored in the water column at least 75 feet from the seafloor, and no explosive mines would be detonated on the ocean bottom. The force of a

large explosive mine detonated in the water column could result in impacts on features of benthic habitat that may be utilized by green sturgeon prey including effects on water quality from increased siltation and turbidity and physical disturbance of habitats utilized by benthic invertebrates. However, we have no specific information on the likelihood, duration, or potential magnitude of such impacts from an E7 explosive mine detonated 75 feet from the seafloor.

The only detonations occurring at or near the seafloor from this activity would be with mine neutralizers. The proposed mine neutralizers (bin E4 explosives) are shaped charges that would be used to penetrate live moored explosives mines, inert moored targets, or bottom-placed (inert only) targets. Unlike typical high explosives, mine neutralizers would have a directional force aimed at the target and focus a majority of the energy produced by the high explosive into a narrow beam. Given the relatively small net explosive weight of the proposed neutralizers, the unique properties of explosions from shaped charges hitting inert targets, the primarily soft bottom areas where this activity would likely occur, and the proposed mitigation to reduce impacts on areas of high seafloor complexity, we find that the impacts of this activity on benthic habitats, and associated green sturgeon prey, within Southern DPS green sturgeon designated critical habitat are likely insignificant.

In summary, we find that stressors associated with the proposed action would either have no effect or insignificant effects on the PBFs of Southern DPS green sturgeon designated critical habitat. Therefore, the proposed action is not likely to adversely affect Southern DPS green sturgeon designated critical habitat. Based on our critical habitat effects analysis presented in Section 8.3.3.3, we do not anticipate the effects of the proposed action will appreciably diminish the value of Southern DPS green sturgeon designated critical habitat as a whole for the conservation of this species.

11 CONCLUSION

We find that the proposed action is **not likely to adversely affect** the North Pacific right whale, gray whale – Western North Pacific DPS, steelhead – Southern California DPS, Steller sea lion – Western DPS, green sea turtle – East Pacific DPS, loggerhead sea turtle – North Pacific Ocean DPS, olive ridley sea turtle Mexico's Pacific Breeding Coast colonies, and olive ridley sea turtle – all other areas; thus, it is also not likely to jeopardize the continued existence of these species.

After reviewing the current status of the species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is **not likely to jeopardize** the continued existence of the following ESA-listed species: blue whale, fin whale, humpback whale (Central America DPS and Mexico DPSs), sei whale, killer whale – Southern Resident killer whales DPS, sperm whales, Guadalupe fur seal, leatherback sea turtle, bocaccio - Puget Sound/Georgia Basin DPS, yelloweye rockfish - Puget Sound/Georgia Basin DPS, Pacific eulachon - Southern DPS, green sturgeon – Southern DPS, Chinook salmon (Sacramento River winter-run, Central valley spring-run, California coastal, Snake River fall-run, Snake River spring/summer-run, Lower Columbia River, Upper Willamette River, Upper Columbia River spring-run, and Puget Sound ESUs), chum salmon (Hood Canal summer-run and Columbia River ESUs), coho salmon (Central California coast, Southern Oregon and Northern California coast, Lower Columbia River, and Oregon Coast ESUs), sockeye salmon (Snake River ESU), and steelhead (South-Central California Coast, Central California Coast, California Central Valley, Northern California, Upper Columbia River, Snake River Basin, Lower Columbia River, Upper Willamette River, Middle Columbia River, and Puget Sound DPSs).

After reviewing the current status of designated critical habitat, the environmental baseline within the action area, the effects of the proposed action on the essential physical and biological features and conservation value of designated critical habitat within the action area, and any cumulative effects, it is NMFS' biological opinion that the proposed action is **not likely to destroy or adversely modify** designated critical habitat for the following species: leatherback sea turtle, Chinook salmon – Puget Sound ESU, chum salmon – Hood Canal summer-run ESU, bocaccio - Puget Sound/Georgia Basin DPS, and yelloweye rockfish - Puget Sound/Georgia Basin DPS, and green sturgeon – Southern DPS.

After reviewing the current status of designated critical habitat, the environmental baseline within the action area, the effects of the proposed action on the essential physical and biological features and conservation value of designated critical habitat within the action area, and any cumulative effects, it is NMFS' conference opinion that the proposed action is **not likely to destroy or adversely modify** proposed critical habitat for the following species: humpback whale (Central America DPS and Mexico DPSs), and killer whale – Southern Resident killer whales DPS.

12 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. In the case of threatened species, section 4(d) of the ESA leaves it to the Secretary's discretion whether and to what extent to extend the statutory 9(a) ''take'' prohibitions, and directs the agency to issue regulations it considers necessary and advisable for the conservation of the species. At the time of this consultation, take prohibitions have not been extended to the following threatened species: yelloweye rockfish and eulachon. However, consistent with *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we assessed the amount or extent of take to these threatened species that is anticipated incidental to Navy training and testing activities and include this information in the ITS. Inclusion of these species in the ITS serves to assist the action agency with monitoring of take and provides a trigger for reinitiation if levels of estimated take are exceeded.

"Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. NMFS had not yet defined "harass" under the ESA in regulation, but has issued interim guidance on the term "harass," defining it as to "create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering." We considered NMFS' interim definition of harassment in evaluating whether the proposed activities are likely to result in harassment of ESA-listed species. Incidental take statements serve a number of functions, including providing reinitiation triggers for all anticipated take, providing exemptions from Section 9 liability for prohibited take, and identifying reasonable and prudent measures that will minimize the impact of anticipated incidental take.

Further, when an action will result in incidental take of ESA-listed marine mammals, ESA section 7(b)(4) requires that such taking be authorized under the MMPA section 101(a)(5) before the Secretary can issue an ITS for ESA-listed marine mammals and that an ITS specify those measures that are necessary to comply with Section 101(a)(5) of the MMPA. Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS, including those specified as necessary to comply with the MMPA, Section 101(a)(5). Accordingly, the terms of this ITS and the exemption from Section 9 of the ESA become effective only upon the issuance of MMPA authorization to take the marine mammals identified here. Absent such authorization, this ITS is inoperative for ESA-listed marine mammals.

12.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 C.F.R. §402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by the proposed action. The extent of take represents the "extent of land or marine area that may be affected by an action" and may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 FR 19953).

Table 189 lists the anticipated take from training and testing activities by species and the issuance of a seven-year regulation and LOAs by NMFS' Permits Division to authorize take of marine mammals pursuant to the MMPA. The amount of take resulting from NWTT Phase III activities was estimated based on the best information available. The numbers presented for marine mammals for vessel strike represent total exempted over a seven-year period; all other numbers presented are annual take levels. For marine mammals, as described in Section 8.2.1.4, a total of three large whale vessel strikes are exempted over the seven-year period of the MMPA rule (from December 2020 to December 2027). Of the three whale vessel strike takes exempted over seven years, no more than two may come from any of the following species: fin whale and humpback whale. Additionally, of those three whales over the seven years, no more than one may be a sperm whale.

When it is not possible or practicable to specify the amount or extent of take, a surrogate may be used if we: describe the causal link between the surrogate and take of the listed species, explain why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and set a clear standard for determining when the level of anticipated take has been exceeded. 50 C.F.R. 402.14(g)(7).

As described previously in Section 8.2.3, due to the lack of available density and abundance information on the life stages of ESA-listed rockfish exposed to explosives we cannot estimate the total number of individual fish (i.e., pelagic juvenile bocaccio, larval bocaccio, and larval yelloweye rockfish) that may be affected by NWTT activities using explosives. As such, it is not possible, nor would it be an accurate representation of potential effects, to express the amount of anticipated take (i.e., in the form of mortality, injury, TTS, and behavioral disruption) of bocaccio and yelloweye rockfish or to monitor take-related impacts in terms of individuals of these species. Since it is not practical to express the amount of anticipated take of ESA-listed rockfish, we instead use a surrogate measure to express the amount or extent of incidental take. The surrogate for the incidental take of ESA-listed rockfish is based on 1) the ranges to effects, which are the distances in the water column that correlate with each of the predicted effects (e.g., mortality, injury, TTS) from explosives in those areas occupied by pelagic juvenile bocaccio, larval bocaccio, and larval yelloweye rockfish, and 2) explosive activity level, which is the total number of explosives (by bin type) that would be detonated in the inland portion of the action

area (i.e. Hood Canal and Crescent Harbor EOD range sites) as part of the Navy's proposed action (see Section 8.2.3 for details on range to effects and activity levels).

Predicted ranges to effects for ESA-listed fish species are calculated with NAEMO based upon sound exposure criteria which are used to predict the onset of mortality, injury, or TTS within fishes within the NWTT action area. Ranges may vary greatly depending on factors such as the cluster size, location, depth, and season of the event, all of which are accounted for in NAEMO. Fishes within these ranges could be exposed above the effect threshold levels and a portion may experience those effects. Ranges to effects, in combination with explosive activity levels (i.e., the number of explosives used, by bin type, in portions of the action area, can be used to estimate the area within the water column within which fish would be exposed to an impulsive sound source at levels resulting in the anticipated effects (i.e., mortality, injury, TTS). Thus, there is a causal link between the surrogate factors above (i.e., range to effects and activity level) and take of ESA-listed rockfish as a result of the proposed action.

NAEMO predicted ranges to effects could change based on new model inputs, including changes in environmental data, new findings on the responses of fish to impulsive sound sources, or updated sound exposure criteria levels based on new information. Any increase in the predicted ranges to effects on fish from explosives used in the inland portion of the action area (i.e., bins E0 and E3) would result in a larger area within the water column within which ESA-listed rockfish would be exposed, and therefore, exceedance of the surrogate used to measure take of these species. Exceedance of an activity level as a trigger for consultation reinitiation is discussed below for any activity associated with take of an ESA-listed species.

In addition to the estimated level of take of Southern Resident killer whales anticipated from acoustic sources as shown in Table 189, based on our effects analysis (Section 8.2.1.4), we also anticipate take of Southern Resident killer whales from the indirect effects to this DPS resulting from impacts to their salmonid prey. Southern Resident killer whale take from the effects of explosives on their prey would likely be in the form of behavioral harassment, as reduced prey availability would result in whales spending more time and energy searching for prey, which can lead to nutritional stress. Whereas our analysis above estimates the quantity of prey (i.e., salmon by ESU and lifestage) that would likely be killed and injured, specifying the level of take of Southern Resident killer whales resulting from this estimated level of impact to their prey is neither possible nor practicable based on the information available. Predicting the spatio-temporal overlap between the whales' presence and impacts to their prey is extremely difficult. Southern Resident killer whales move quickly and frequently throughout their range, and both their presence within the action area and the explosive activities impacting their prey are highly variable through space and time. Further, even if the overlap between whale presence and prey reduction were known, we do not have information to reliably equate an estimated number of salmon killed (or injured) to a particular level of Southern Resident killer whale behavioral harassment take.

Since it is not possible or practicable to specify the amount or extent of take, we use the estimated impacts (i.e., mortalities and injuries) to their primary salmonid prey populations as a surrogate for take of Southern Resident killer whales from the indirect effects of the proposed action. These are shown in Table 135 and Table 136, for ESA-listed and non-listed salmon ESUs respectively. As impacts to these salmonid prey populations increase, Southern Resident killer whale prey availability decreases, resulting in increased energy expenditure searching for and capturing prey. Thus, there is a direct causal link between the level of salmon injury and mortality and the indirect take of Southern Resident killer whales via impacts to their prey. As discussed below, since impacts on salmonid prey species are directly related to explosive use, we use the link between explosive use and the level of take to determine when anticipated take levels have been exceeded.

Table 189. The estimated total number of lethal and non-lethal takes of threatened and endangered marine mammals, sea turtles, and fish reasonably certain to occur over a seven-year period as a result of the proposed Navy training and testing activities in the NWTT action area. Numbers in parentheses indicate the estimated maximum annual number of takes likely to occur based on the proposed maximum activity level (except in the case of mortality from vessel strike).

ESA-Listed Species	Impulsive and Non-Impulsive Acoustic Stressors					Vessel Strike[1] (over a 7 year period)	
	Harassment (TTS)	Harassment (Behavioral)	Harm (PTS)	Harm (lung injury)	Mortality	Mortality	Harm (non-lethal injuries)
Marine Mammals							
Blue Whale	21 (4)	28 (6)	-	-	-	-	-
Fin Whale[2]	246 (45)	530 (92)	-	-	-	2	-
Humpback Whale (Central America DPS)	19 (4)	19 (4)	-	-	-	2	-
Humpback Whale (Mexico DPS)	250 (49)	196 (36)	-	-	-		
Southern Resident killer whale	11 (2)	232 (49)	-	-	-	-	-
Sei Whale	266 (50)	197 (33)	-	-	-	-	-
Sperm Whale	31 (5)	4,986 (834)	-	-	-	1	-
Guadalupe Fur Seal	85 (13)	8,184 (1,482)	-	-	-	-	-
Sea Turtles							
Leatherback Sea Turtle	-	26 (4)	-	-	-	-	-

Fishes – Effects of Explosives								
Species	Life stage	ESU/DPS	Natural		Hatchery: adipose clip[3]		Hatchery: adipose intact	
			Mortality	Injury	Mortality	Injury	Mortality	Injury
Chinook salmon	Adult	Sac River winter run - E	- (-)	11 (4)	1 (-)	120 (40)	- (-)	- (-)
	Juvenile		29 (10)	3,513 (1,171)	29 (10)	3,597 (1,199)	- (-)	- (-)
	Adult	Central valley spring run - T	2 (1)	201 (67)	1 (-)	123 (41)	- (-)	- (-)
	Juvenile		114 (38)	13,946 (4,649)	318 (106)	39,014 (13,005)	- (-)	- (-)
	Adult	California coastal - T	3 (1)	380 (127)	- (-)	- (-)	- (-)	- (-)
	Juvenile		188 (63)	22,985 (7,662)	- (-)	- (-)	- (-)	- (-)
	Adult	Snake River fall - T	19 (3)	2,311 (364)	23 (3)	2,796 (440)	25 (4)	3,029 (476)
	Juvenile		84 (28)	10,309 (3,436)	302 (101)	36,956 (12,319)	348 (116)	42,591 (14,197)
	Adult	Snake River spring/summer - T	22 (3)	2,643 (416)	4 (0)	493 (77)	1 (-)	87 (14)
	Juvenile		122 (41)	14,991 (4,997)	541 (180)	66,267 (22,089)	94 (31)	11,536 (3,845)
	Adult[4]	Lower Columbia River - T	5 (2)	625 (208)	- (-)	- (-)	7 (2)	818 (273)
	Juvenile		1,620 (540)	198,477 (66,159)	4,324 (1,441)	529,836 (176,612)	133 (44)	16,264 (5,421)
	Adult[4]	Upper Willamette River - T	17 (3)	2,129 (335)	- (-)	- (-)	54 (9)	6,568 (1033)
	Juvenile		162 (54)	19,887 (6,629)	631 (210)	77,277 (25,759)	- (-)	3 (1)
	Adult	Upper Columbia River spring - E	5 (1)	593 (93)	11 (2)	1,286 (202)	6 (1)	695 (109)
	Juvenile		57 (19)	6,976 (2,325)	75 (25)	9,251 (3,084)	45 (15)	5,485 (1,828)
	Adult[4]	Puget Sound - T	5 (1)	474	- (-)	- (-)	12 (2)	795 (152)
	Juvenile		1,216 (226)	207,122 (36,047)	3,951 (1,195)	550,040 (155,811)	1,426 (330)	128,460 (33,823)
Coho salmon	Adult	Central California coast - E	22 (7)	2697 (899)	- (-)	- (-)	- (-)	11 (4)
	Juvenile		4 (1)	482 (161)	- (-)	- (-)	23 (8)	2829 (943)
	Adult[4]		4 (1)	482	- (-)	- (-)	5 (2)	581 (194)

	Juvenile	S. Oregon/N. California coast - T	437 (146)	53,531 (17,844)	43 (14)	5,317 (1,772)	125 (42)	15,286 (5,095)
	Adult	Oregon coast - T	41 (14)	5,015 (1,672)	- (-)	30 (10)	- (-)	- (-)
	Juvenile		1,441 (480)	176,566 (58,855)	13 (4)	1,595 (532)	- (-)	- (-)
	Adult	Lower Columbia River - T	16 (5)	1,906 (635)	- (-)	- (-)	5 (2)	561 (187)
	Juvenile		172 (57)	21,105 (7,035)	1,897 (632)	232,524 (77,508)	65 (22)	7,970 (2,657)
Chum salmon	Adult	Hood Canal summer run	156 (22)	9,672 (1,377)	- (-)	- (-)	11 (2)	673 (96)
	Juvenile		1,568 (224)	119,313 (16,791)	- (-)	- (-)	109 (16)	8,198 (1,171)
	Adult	Columbia River - T	- (-)	14 (5)	- (-)	- (-)	- (-)	1 (-)
	Juvenile		24 (8)	2,932 (977)	- (-)	- (-)	3 (1)	326 (109)
Sockeye salmon	Adult	Snake River - E	- (-)	1 (0)	- (-)	4 (1)	- (-)	- (-)
	Juvenile		- (-)	8 (2)	1 (0)	98 (29)	- (-)	- (-)
Steelhead	Adult	South-Central California - T	- (-)	17 (3)	- (-)	- (-)	- (-)	- (-)
	Juvenile		6 (0)	664 (99)	- (-)	- (-)	- (-)	- (-)
	Adult	Central California - T	- (-)	46 (7)	- (-)	81 (12)	- (-)	- (-)
	Juvenile		15 (2)	1,793 (271)	38 (6)	4678 (690)	- (-)	- (-)
	Adult	California Central Valley - T	- (-)	36 (5)	- (-)	82 (12)	- (-)	- (-)
	Juvenile		36 (6)	4,367 (662)	91 (14)	11,088 (1,702)	- (-)	- (-)
	Adult	Northern California - T	1 (-)	152 (23)	- (-)	- (-)	- (-)	- (-)
	Juvenile		47 (8)	5,787 (877)	- (-)	- (-)	- (-)	- (-)
	Adult	Upper Columbia River - E	- (-)	50 (8)	2 (-)	136 (17)	- (-)	30 (5)
	Juvenile		12 (2)	1,496 (226)	43 (6)	5,158 (731)	9 (2)	1039 (157)
	Adult	Snake River basin - T	2 (-)	219 (33)	14 (2)	1,651	3 (-)	335 (51)
	Juvenile		46 (6)	5,526 (839)	187 (29)	22,842 (3,509)	40 (6)	4,883 (741)
	Adult[4]	Lower Columbia River - T	2 (-)	267 (40)	- (-)	- (-)	3 (-)	461 (70)
	Juvenile		20 (3)	2,426 (368)	68 (10)	8,248 (1,273)	- (-)	63 (10)
	Adult	Upper Willamette River - T	- (-)	62 (9)	- (-)	- (-)	- (-)	- (-)
	Juvenile		9 (2)	1,094 (165)	- (-)	- (-)	- (-)	- (-)
	Adult	Middle Columbia River - T	1 (-)	111 (17)	- (-)	9 (1)	- (-)	3 (-)
	Juvenile		25 (3)	2,941 (446)	26 (4)	3209 (473)	7 (1)	796 (120)

	Adult[5]	Puget Sound - T	8 (2)	672 (100)	- (-)	- (-)	- (-)	- (-)
	Juvenile		627 (92)	46,203 (6,736)	6 (1)	757 (117)	6 (1)	774 (118)
Eulachon	Adult	Southern - T	1,023 (341)	37,104 (12,368)	-	-	-	-
Green Sturgeon	Subadult and Adult (combined)	Southern - T	40 (14)	4,809 (1,603)				

[1] No more than three vessel strikes in total (fin, humpback and sperm whales combined) are estimated over seven years. Of the three whales that have the potential to be struck over seven years, no more than two may come from any of the following species: fin whale and humpback whale. Additionally, of those three whales over the seven years, no more than one may be a sperm whale.
[2] Includes California, Oregon, and Washington and Northeast Pacific stocks.
[3] It should be noted that ESA take prohibitions do not apply to hatchery fish with clipped adipose fins from threatened ESUs or DPSs
[4] Hatchery intact adipose mortality and injury estimates comprise of hatchery fish with intact and clipped adipose fins.
[5] Includes natural and hatchery (clipped and intact adipose fish) estimates.

Activity Levels as Indicators of Take for Marine Mammals, Sea Turtles, and Fishes

As discussed in this opinion, the estimated take of ESA-listed sea turtles, marine mammals, and fish[30] from acoustic stressors is based on Navy modeling, which represents the best available means of numerically quantifying take. As the level of modeled sonar or explosive use increases, the level of take is likely to increase as well. For take from acoustic sources specified above, feasible monitoring techniques for detecting and calculating actual take at the scale of NWTT activities do not exist. We are not aware of any other feasible or available means of determining when estimated take levels may be exceeded. Therefore, we must rely on Navy modeling, and the link between sonar or explosive use and the level of take, to determine when anticipated take levels have been exceeded. As such, we established a term and condition of this Incidental Take Statement that requires the Navy to report to NMFS any exceedance of activity levels specified in the preceding opinion and in the final MMPA rule before the exceedance occurs if operational security considerations allow, or as soon as operational security considerations allow after the relevant activity is conducted. Exceedance of an activity level for any activity associated with take of an ESA-listed species will require the Navy to reinitiate consultation.

12.2 Effects of the Take

In this opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to jeopardize the continued existence or recovery of any ESA-listed species or result in the destruction or adverse modification of designated (or proposed) critical habitat.

[30] For estimating take of fish, NMFS only used the ranges to effects modeled by the Navy's NAEMO model.

12.3 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species.

To minimize the impacts of incidental take, necessary and appropriate reasonable and prudent measures, and terms and conditions to implement the measures, must be provided. Only incidental takes resulting from the agency actions carried out consistent with the specified reasonable and prudent measures and terms and conditions identified in the ITS are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

Reasonable and prudent measures, and implementing terms and conditions, are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). The reasonable and prudent measures and terms and conditions are specified as required by 50 C.F.R. 402.14 (i)(1)(ii) and (iv) to document the incidental take by the proposed action and minimize the impact of that take on ESA-listed species. The reasonable and prudent measures are nondiscretionary, and must be undertaken by the Navy and NMFS' Permits Division so that they become binding conditions for the exemption in section 7(o)(2) to apply.

NMFS has determined the following reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take of threatened and endangered species during the proposed action:

1. The Navy and NMFS Permits Division shall minimize effects to ESA-listed marine mammals, sea turtles, and fishes from the use of active sonar, explosives, and vessels during training and testing activities. This includes adherence to the mitigation measures specified in the final MMPA rule and LOA and those measures described in Section 3.5 of the preceding opinion.

2. The Navy and NMFS Permits Division shall monitor and report to NMFS' Office of Protected Resources ESA Interagency Cooperation Division on impacts to ESA-listed marine mammals, sea turtles, and fishes from the use of sonar and other transducers, explosives, and vessels during training and testing activities. This includes adherence to the monitoring and reporting measures specified in the final MMPA rule and LOA.

12.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Navy and NMFS Permits Division must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). These terms and conditions are non-discretionary. If the Navy or NMFS Permits Division fail to ensure

compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

1) The following terms and conditions implement reasonable and prudent measure 1:

   a) The Navy shall implement all mitigation measures as specified in the final MMPA rule and LOA, and as described in this opinion in Section 3.5.

   b) NMFS Permits Division shall ensure that all mitigation measures as prescribed in the final rule and LOA, and as described in Section 3.5 of this opinion are implemented by the Navy.

   c) The Navy shall continue technical assistance/adaptive management efforts with NMFS to help inform future consultations on Navy training and testing in the action area. Adaptive management discussions should include review of Navy's exercise and monitoring reports, review of ESA section 7 reinitiation triggers (described in Section 14 below), and potential new measures to increase mitigation effectiveness (e.g., thermal detection of protected species).

2) The following terms and conditions implement reasonable and prudent measure 2:

   a) The Navy shall monitor training and testing activities and submit reports annually to NMFS Permits Division and NMFS ESA Interagency Cooperation Division including the location and total hours and counts of active sonar hours and in-water explosives used, and an assessment if activities conducted in the action area exceeded levels of training and testing analyzed in this opinion annually and over the seven-year period of the MMPA regulations and LOAs.

   b) NMFS Permits Division shall review the reports submitted by the Navy described above in 2(a). Within two months of receipt of each Navy report, NMFS Permits Division will submit written documentation to NMFS ESA Interagency Cooperation Division assessing if Navy activities conducted in the action area exceeded levels of training and testing analyzed in this opinion annually and over the seven-year period of the MMPA regulations and LOAs.

   c) The Navy shall provide a report summarizing the status of and/or providing a final assessment on the Navy's Lookout Effectiveness Study following the end of Calendar Year (CY) 2021. The report will be submitted no later than 90 days after the end of CY2021. The report will provide a statistical assessment of the data available to date characterizing the effectiveness of Navy lookouts relative to trained marine mammal observers for the purposes of implementing the mitigation measures required in this biological opinion.

   d) The Navy shall report to the NMFS ESA Interagency Cooperation Division all observed injury or mortality of any ESA-listed species resulting from the proposed training and

testing activities within the action area. The Navy shall report when enough data are available to determine if the dead or seriously injured ESA-listed species may be attributable to these activities, including but not limited to, the use of explosives and vessel strike.

e) In the event that Navy personnel (uniformed military, civilian, or contractors while conducting Navy work) discover a live or dead stranded marine mammal or sea turtle within the action area or on Navy property, the Navy shall comply with the stranding Notification and Reporting Plan.

f) If NMFS personnel determine that the circumstances of any of the strandings reported in 2(f) suggest investigation of the associated Navy activities is warranted (see stranding and notification document for example circumstances), and an investigation into the stranding is being pursued, NMFS personnel will submit a written request to the Navy asking that they provide the status of all sound sources and explosive use in the 48 hours preceding and within 50 km (27 NM) of the discovery/notification of the stranding by NMFS, or estimated time of stranding. Navy will submit this information as soon as possible, but no later than seven business days after the request.

13 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

1. The Navy should monitor and report sighting, location, and stranding data for MMPA- and ESA-listed marine mammals, sea turtles, and fish in the NWTT action area.

2. As practicable, the Navy should develop procedures to aid any individual ESA-listed marine mammal or sea turtle that has been impacted by NWTT activities and is in a condition requiring assistance to increase likelihood of survival.

3. The Navy should continue to model potential impacts to ESA-listed marine mammals and sea turtles using NAEMO and other relevant models. The Navy should validate assumptions used in risk analyses and seek new information and higher quality data for use in such efforts.

4. The Navy should work towards developing a model-based approach for quantitatively analyzing the effects of acoustic stressors (i.e., sonar and explosives) on ESA-listed fishes. The Navy should coordinate with NMFS' regional science centers or other entities to collect additional information on the density and distribution of ESA-listed fishes within the NWTT action area in order to incorporate into density models in the future.

5. The Navy should continue the development of autonomous marine mammal detection technologies to reduce the risk of vessel strike.

6. The Navy should continue to conduct behavioral response studies aimed at obtaining response data that is more consistent with the received sound levels, distances, and durations of exposure that animals are likely to receive incidental to actual training and testing activities.

7. The Navy should coordinate with NMFS on the collection of information for better understanding the effectiveness of mitigation measures proposed by the Navy during NWTT sonar and explosives use. This should include an assessment of the effectiveness of Navy Lookouts for minimizing impacts to ESA-listed species (see Oliveira et al. 2019). Findings should be incorporated into the Navy's approach to quantitatively evaluating the effects of acoustics stressors on ESA-listed species.

8. As practicable, the Navy should supplement the proposed visual monitoring mitigation measures described in Section 3.5.1 with passive and active acoustic monitoring for activities that could cause cetacean injury or mortality.

9. We recommend the Navy consider using the potential standards for towed array passive acoustic monitoring in the Towed Array Passive Acoustic Operations for Bioacoustic

Applications: ASA/JNCC Workshop Summary March 14-18, 2016 Scripps Institution of Oceanography, La Jolla, California, USA (Thode 2017).

10. The Navy should continue to conduct research on thermal detection monitoring systems, as a supplement to visual monitoring, to further minimize the impacts of Navy acoustic stressors on ESA-listed marine mammals.

11. To the extent practicable, the Navy should extend the active sonar mitigation zone if a Southern Resident killer whale is observed to avoid or reduce the potential for exposure to TTS or behavioral harassment.

12. To the extent practicable, the Navy should cease transmission of mid-frequency active sonar and cease detonations (for activities involving explosives) if a Southern Resident killer whale is observed by Navy Lookouts or by Navy personnel on additional platforms at any distance.

13. The Navy should coordinate with state and federal resource managers to identify research priorities and carry out actions that aid in the recovery and management of Southern Resident killer whales.

14. The Navy should conduct additional aerial surveys in the Pacific Northwest region to detect Southern Resident killer whales.

In order for NMFS' Office of Protected Resources ESA Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, the Navy and NMFS Permits Division should notify the ESA Interagency Cooperation Division of any conservation recommendations they implement in their final action.

14 REINITIATION NOTICE

This concludes formal consultation on the Navy's proposed NWTT Phase III activities and NMFS' promulgation of regulations and issuance of incidental take authorizations pursuant to the MMPA. As 50 C.F.R. §402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if:

(1) The amount or extent of taking specified in the ITS is exceeded.
(2) New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered.
(3) The identified action is subsequently modified in a manner that causes an effect to ESA-listed species or designated critical habitat that was not considered in this opinion.
(4) A new species is listed, or critical habitat designated under the ESA that may be affected by the action.

15 REFERENCES

57 FR 14658. Endangered and Threatened species: Threatened status for Snake River spring/summer Chinook salmon, threatened status for Snake River fall Chinook salmon. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

58 FR 68543. Designated critical habitat; Snake River sockeye salmon, Snake River spring/summer Chinook salmon, and Snake River fall Chinook salmon. Final Rule. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

59 FR 440. Endangered and Threatened Species; Status of the Sacramento River Winter-run Chinook Salmon. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

64 FR 57399. Designated critical habitat: revision of critical habitat for Snake River spring/summer Chinook salmon. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

70 FR 37160. Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

70 FR 52488. Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

71 FR 17757. Endangered and Threatened Wildlife and Plants: Threatened Status for Southern Distinct Population Segment of North American Green Sturgeon. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

79 FR 20802. Endangered and Threatened Wildlife; Final Rule To Revise the Code of Federal Regulations for Species Under the Jurisdiction of the National Marine Fisheries Service. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

84 FR 54354. Endangered and Threatened Wildlife and Plants: Proposed Rule to Designate Critical Habitat for the Central America, Mexico, and Western North Pacific Distinct Population Segments of Humpback Whales. Pages 54354-54391 *in* N. O. A. A. National Marine Fisheries Service, Commerce, editor.

85 FR 53763. Guidelines for Safely Deterring Marine Mammals. Pages 53763-53785 *in* N. O. A. A. National Marine Fisheries Service, Commerce, editor.

58 FR 33212. Designated Critical Habitat; Sacramento River Winter-run Chinook Salmon. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

FR 64 50394. Endangered and Threatened Species; Threatened Status for Two Chinook Salmon Evolutionarily Significant Units (ESUs) in California. Pages FR 64 50394 *in* N. O. A. A. National Marine Fisheries Service, Commerce, editor.

Abdul-Aziz, O. I., N. J. Mantua, and K. W. Myers. 2011. Potential climate change impacts on thermal habitats of Pacific salmon (*Oncorhynchus* spp.) in the North Pacific Ocean and adjacent seas. Canadian Journal of Fisheries and Aquatic Science 68:1660-1680.

Aburto, A., D. J. Rountry, and J. L. Danzer. 1997. Behavioral responses of blue whales to active signals. Naval Command, Control and Ocean Surveillance Center, RDT&E Division, Technical Report 1746, San Diego, CA.

Acevedo-Whitehouse, K., A. Rocha-Gosselin, and D. Gendron. 2010. A novel non-invasive tool for disease surveillance of free-ranging whales and its relevance to conservation programs. Animal Conservation 13(2):217-225.

Acevedo, A. 1991. Interactions between boats and bottlenose dolphins, Tursiops truncatus, in the entrance to Ensenada de la Paz, Mexico. Aquatic Mammals 17(3):120-124.

Ackerman, R. A. 1997. The nest environment and the embryonic development of sea turtles. Pages 83-106 *in* P. L. M. Lutz, J. A. , editor. The biology of sea turtles. CRC Press, Boca Raton.

Adams, J., J. Felis, J. W. Mason, and J. Y. Takekawa. 2014a. Pacific Continental Shelf Environmental Assessment (PaCSEA): Aerial Seabird and Marine Mammal Surveys off Northern California, Oregon, and Washington, 2011-2012 (OCS Study BOEM 2014-003). Bureau of Ocean Energy Management, Camarillo, California.

Adams, J., J. Felis, J. W. Mason, and J. Y. Takekawa. 2014b. Pacific Continental Shelf Environmental Assessment (PaCSEA): Aerial Seabird and Marine Mammal Surveys off Northern California, Oregon, and Washington, 2011–2012. Bureau of Ocean Energy Management, Camarillo, CA.

Adams, P. 2000. Status review update for the steelhead Northern California Evolutionary Significant Unit. Southwest Fisheries Science Center, Santa Cruz/Tiburon Laboratory, Tiburon, California.

Adams, P. B., C. B. Grimes, S. T. Lindley, and M. L. Moser. 2002. Status review for North American green sturgeon, Acipenser medirostris. National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz, CA.

Agler, B. A., and coauthors. 1990. Fin whale (*Balaenoptera physalus*) photographic identification: Methodology and preliminary results from the western North Atlantic. Report of the International Whaling Commission Special Issue 12:349-356.

Aguayo, L. A. 1974. Baleen whales off continental Chile. Pages 209-217 *in* W. E. Schevill, editor. The Whale Problem: A Status Report. Harvard University Press, Cambridge, Massachusetts.

Aguilar, A. 1983. Organochlorine pollution in sperm whales, *Physeter macrocephalus*, from the temperate waters of the eastern North Atlantic. Marine Pollution Bulletin 14(9):349-352.

Aguilar, A., and A. Borrell. 1988. Age- and sex-related changes in organochlorine compound levels in fin whales (*Balaenoptera physalus*) from the Eastern North Atlantic. Marine Environmental Research 25(1988?):195-211.

Aguilar, A., and C. H. Lockyer. 1987. Growth, physical maturity, and mortality of fin whales (*Balaenoptera physalus*) inhabiting the temperate waters of the northeast Atlantic. Canadian Journal of Zoology 65:253-264.

Aguilar Soto, N., and coauthors. 2006. Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? Marine Mammal Science 22(3):690-699.

Aguirre, A., and coauthors. 2007. Assessment of sea turtle health in Peconic Bay of eastern Long Island. Pages 108 *in* Twenty-Fourth Annual Symposium on Sea Turtle Biology and Conservation.

Akamatsu, T., K. Nakamura, H. Nitto, and M. Watabe. 1996. Effects of underwater sounds on escape behavior of Steller sea lions. Fisheries Science 62(4):503–510.

Al-Humaidhi, A. 2011. Analysis of green sturgeon bycatch by sector and time in the West Coast Groundfish Fishery. 3pp. Included as Attachment 2 to: National Marine Fisheries Service. 2011. Endangered Species Act Section 7.

Allen, A. N., J. J. Schanze, A. R. Solow, and P. L. Tyack. 2014. Analysis of a Blainville's beaked whale's movement response to playback of killer whale vocalizations. Marine Mammal Science 30(1):154-168.

Allen, B. M., and R. P. Angliss. 2010. Alaska Marine Mammal Stock Assessments, 2009. U.S. Department of Commerce.

Allen, J. K., M. L. Peterson, G. V. Sharrard, D. L. Wright, and S. K. Todd. 2012. Radiated noise from commercial ships in the Gulf of Maine: Implications for whale/vessel collisions. Journal of the Acoustical Society of America 132(3):EL229-EL235.

Allen, M. R., H. de Coninck, O. P. Dube, and D. J. Heogh-Guldberg Ove; Jacob, Kejun; Revi, Aromar; Rogelj, Joeri; Roy, Joyashree; Shindell, Drew; Solecki, William; Taylor, Michael; Tschakert, Petra; Waisman, Henri; Halim, Sharina Abdul; Antwi-Agyei, Philip; Aragón-Durand, Fernando; Babiker, Mustafa; Bertoldi, Paolo; Bindi, Marco; Brown, Sally; Buckeridge, Marcos; Camilloni, Ines; Cartwright, Anton; Cramer, Wolfgang; Dasgupta, Purnamita; Diedhiou, Arona; Djalante, Riyanti; Dong, Wenjie; Ebi, Kristie L.; Engelbrecht, Francois; Fifita, Solomone; Ford, James; Forster, Piers; Fuss, Sabine; Hayward, Bronwyn; Hourcade, Jean-Charles; Ginzburg, Veronika; Guiot, Joel; Handa, Collins; Hijioka, Yasuaki; Humphreys, Stephen; Kainuma, Mikiko; Kala, Jatin; Kanninen, Markku; Kheshgi, Haroon; Kobayashi, Shigeki; Kriegler, Elmar; Ley, Debora; Liverman, Diana; Mahowald, Natalie; Mechler, Reinhard; Mehrotra, Shagun; Mulugetta, Yacob; Mundaca, Luis; Newman, Peter; Okereke, Chukwumerije; Payne, Antony; Perez, Rosa; Pinho, Patricia Fernanda; Revokatova, Anastasia; Riahi, Keywan; Schultz, Seth; Séférian, Roland; Seneviratne, Sonia I.; Steg, Linda; Suarez Rodriguez, Avelino G.; Sugiyama, Taishi; Thomas, Adelle; Vilariño, Maria Virginia; Wairiu, Morgan; Warren, Rachel; Zhou, Guangsheng; Zickfeld, Kirsten. 2018. Technical Summary. In: *Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)].

Alves, A., and coauthors. 2014. Vocal matching of naval sonar signals by long-finned pilot whales (Globicephala melas). Marine Mammal Science 30(3):1248-1257.

Amador-Capitanachi, M. J., and coauthors. 2020. Ecological implications of unprecedented warm water anomalies on interannual prey preferences and foraging areas of Guadalupe fur seals. Marine Mammal Science:1-17.

Amaral, K., and C. Carlson. 2005a. Scientific basis for whale watching guidelines. A review of current research. Unpublished paper to the IWC Scientific Committee. 17 pp. Ulsan, Korea, June (SC/57/WW1).

Amaral, K., and C. Carlson. 2005b. Summary of non-lethal research techniques for the study of cetaceans. United Nations Environment Programme UNEP(DEC)/CAR WG.27/REF.5. 3p. Regional Workshop of Experts on the Development of the Marine Mammal Action Plan for the Wider Caribbean Region. Bridgetown, Barbados, 18-21 July.

Anderwald, P., and coauthors. 2013. Displacement responses of a mysticete, an odontocete, and a phocid seal to construction-related vessel traffic. Endangered Species Research 21(3):231-240.

Andrady, A. L. 2011. Microplastics in the marine environment. Marine Pollution Bulletin, 62(8), 1596–1605. doi: 10.1016/j.marpolbul.2011.05.030.

André, M., M. Terada, and Y. Watanabe. 1997. Sperm whale (*Physeter macrocephalus*) behavioural responses after the playback of artificial sounds. Report of the International Whaling Commission 47:499-504.

Andrew, R. K., B. M. Howe, and J. A. Mercer. 2002. Ocean ambient sound: Comparing the 1960s with the 1990s for a receiver off the California coast. Acoustics Research Letters Online 3(2):65-70.

Andrews, A. H., and coauthors. 2005. Bomb radiocarbon and lead-radium disequilibria in otoliths of bocacio rockfish (*Sebstes paucispinis*): a determination of age and longevity for a difficult-to-age fish. Marine and Freshwater Research 56:517-528.

Andrews, R. C. 1916. The sei whale (*Balaenoptera borealis* Lesson). Memoirs of the American Museum of Natural History, New Series 1(6):291-388.

Angliss, R. P., and R. B. Outlaw. 2005. Alaska marine mammal stock assessments, 2005. U.S. Department of Commerce, NMFSAFSC-161.

Angliss, R. P., and R. B. Outlaw. 2008. Alaska marine mammal stock assessments, 2007. Department of Commerce, NMFS-AFSC-180.

Angradi, A. M., C. Consiglio, and L. Marini. 1993. Behaviour of striped dolphins (*Stenella coeruleoalba*) in the central Tyrrhenian Sea in relation to commercial ships. European Research on Cetaceans 7:77-79. Proceedings of the Seventh Annual Conference of the European Cetacean Society, Inverness, Scotland, 18-21 February.

Antonelis, G. A., J. D. Baker, T. C. Johanos, R. C. Braun, and A. L. Harting. 2006. Hawaiian monk seal (*Monachus schauinslandi*): status and conservation issues. Atoll Research Bulletin 543:75-101.

Anttila, C. K., C. C. Daehler, N. E. Rank, and D. R. Strong. 1998. Greater male fitness of a rare invader (Spartina alterniflora, Poaceae) threatens a common native (Spartina foliosa) with hybridization. American Journal of Botany 85:1597-1601.

Antunes, R., and coauthors. 2014. High thresholds for avoidance of sonar by free-ranging long-finned pilot whales (Globicephala melas). Marine Pollution Bulletin 83(1):165-180.

AquaMaps. 2016. Reviewed distribution maps for Acipenser medirostris (Green sturgeon), with modelled year 2100 native range map based on IPCC A2 emissions scenario.

Aquatic Mammals. 2015. Supplemental tables: Biologically important areas for selected cetaceans within U.S. Waters – West Coast region. Aquatic Mammals 41(1):30–32.

Arcangeli, A., and R. Crosti. 2009. The short-term impact of dolphin-watching on the behaviour of bottlenose dolphins (*Tursiops truncatus*) in western Australia. Journal of Marine Animals and their Ecology 2(1):3-9.

Archer, F. I., and coauthors. 2019. Revision of fin whale Balaenoptera physalus (Linnaeus, 1758) subspecies using genetics. Journal of Mammalogy 100(5):1653-1670.

Archer, F. I., and coauthors. 2013. Mitogenomic phylogenetics of fin whales (Balaenoptera physalus spp.): genetic evidence for revision of subspecies. PLoS ONE 8(5):e63396.

Arfsten, D., C. Wilson, and B. Spargo. 2002. Radio frequency chaff: The effects of its use in training on the environment. Ecotoxicology and Environmental Safety 53:11-Jan.

Arias-del-Razo, A., G. Heckel, Y. Schramm, and M. A. Pardo. 2016. Terrestrial habitat preferences and segregation of four pinniped species on the islands off the western coast of the Baja California Peninsula, Mexico. Marine Mammal Science 32(4):1416-1432.

Arnbom, T., V. Papastavrou, L. S. Weilgart, and H. Whitehead. 1987. Sperm whales react to an attack by killer whales. Journal of Mammalogy 68(2):450-453.

Ashe, E., J. Wray, C. R. Picard, and R. Williams. 2013. Abundance and Survival of Pacific Humpback Whales in a Proposed Critical Habitat Area. PLoS ONE 8(9):e75228.

ASSRT. 2007. Status Review of Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*). Atlantic Sturgeon Status Review Team Report to National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Atkinson, S., D. Crocker, D. Houser, and K. Mashburn. 2015. Stress physiology in marine mammals: How well do they fit the terrestrial model? Journal of Comparative Physiology B Biochemical, Systemic and Environmental Physiology 185(5):463-486.

Atkinson, S., D. P. Demaster, and D. G. Calkins. 2008. Anthropogenic causes of the western Steller sea lion *Eumetopias jubatus* population decline and their threat to recovery. Mammal Review 38(1):1-18.

Atlantic Leatherback Turtle Recovery Team. 2006. Recovery Strategy for Leatherback Turtle (Dermochelys coriacea) in Atlantic Canada. Fisheries and Oceans Canada, Ottawa, Canada.

Attard, C. R. M., and coauthors. 2010. Genetic diversity and structure of blue whales (*Balaenoptera musculus*) in Australian feeding aggregations. Conservation Genetics 11(6):2437–2441.

Au, D., and W. Perryman. 1982. Movement and speed of dolphin schools responding to an approaching ship. Fishery Bulletin 80:371-379.

Au, W., J. Darling, and K. Andrews. 2001. High-frequency harmonics and source level of humpback whale songs. Journal of the Acoustical Society of America 110(5 Part 2):2770.

Au, W. W. L., J. K. B. Ford, J. K. Horne, and K. A. Newman Allman. 2004. Echolocation signals of free-ranging killer whales (*Oricinus orca*) and modeling of foraging for chinook salmon (*Oncorhynchus tshawytscha*). Journal of the Acoustical Society of America 115(2).

Au, W. W. L., and M. Green. 2000. Acoustic interaction of humpback whales and whale-watching boats. Marine Environmental Research 49(5):469-481.

Au, W. W. L., and coauthors. 2006a. Acoustic properties of humpback whale songs. Journal of Acoustical Society of America 120(August 2006):1103-1110.

Au, W. W. L., and coauthors. 2006b. Acoustic properties of humpback whale songs. Journal of the Acoustical Society of America 120(2):1103-1110.

Aurioles-Gamboa, D., and F. J. Camacho-Rios. 2007. Diet and feeding overlap of two otariids, *Zalophus californianus* and *Arctocephalus townsendi*: Implications to survive environmental uncertainty. Aquatic Mammals 33(3):315-326.

Aurioles-Gamboa, D., F. Elorriaga-Verplancken, and C. J. Hernandez-Camacho. 2010. The current population status of Guadalupe fur seal (*Arctocephalus townsendi*) on the San Benito Islands, Mexico. Marine Mammal Science 26(2):402-408.

Aurioles-Gamboa, D., C. J. Hernandez-Camacho, and E. Rodriguez-Krebs. 1999. Notes on the southernmost records of the Guadalupe fur seal, *Arctocephalus townsendi*, in Mexico. Marine Mammal Science 15(2):581-583.

Aurioles-Gamboa, D., and D. Szteren. 2019. Lifetime coastal and oceanic foraging patterns of male Guadalupe fur seals and California sea lions. Marine Mammal Science n/a(n/a).

Avens, L., and K. J. Lohmann. 2003. Use of multiple orientation cues by juvenile loggerhead sea turtles, *Caretta caretta*. Journal of Experiential Biology 206(23):4317–4325.

Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8(3):165-177.

Avila, I. C., K. Kaschner, C.F. Dormann. 2018. Current global risks to marine mammals: Taking stock of the threats. Biological Conservation 221:44-58.

Ayres, K. L., and coauthors. 2012. Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (Orcinus orca) population. PLoS ONE 7(6):e36842.

Bailey, H., and coauthors. 2010. Behavioural estimation of blue whale movements in the northeast Pacific from state-space model analysis of satellite tracks. Endangered Species Research 10(1):93-106.

Bailey, K. M., and E. D. Houde. 1989. Predation on eggs and larvae of marine fishes and the recruitment problem. Advances in Marine Biology 25:1-83.

Bain, D. E. 2002. A model linking energetic effects of whale watching to killer whale (Orcinus orca) population dynamics. Friday Harbor Laboratories, University of Washington, Friday Harbor, Washington.

Bain, D. E., D. Lusseau, R. Williams, and J. C. Smith. 2006a. Vessel traffic disrupts the foraging behavior of southern resident killer whales (*Orcinus* spp.). International Whaling Commission.

Bain, D. E., J. C. Smith, R. Williams, and D. Lusseau. 2006b. Effects of vessels on behavior of southern resident killer whales (*Orcinus* spp.).

Baird, R., and H. Whitehead. 2000. Social organization of mammal-eating killer whales: Group stability and dispersal patterns. Canadian Journal of Zoology-revue Canadienne De Zoologie - CAN J ZOOL 78:2096-2105.

Baird, R. W. 2001. Status of killer whales, *Orcinus orca*, in Canada. The Canadian Field-Naturalist 115(4):676-701.

Baird, R. W., and coauthors. 2013. Odontocete Studies Off the Pacific Missile Range Facility in February 2013: Satellite-Tagging, Photo Identification, and Passive Acoustic Monitoring for Species Verification. U.S. Navy Pacific Fleet.

Baird, R. W., and coauthors. 2017. Final Report for Commander, U.S. Pacific Fleet. Odontocete studies on the Pacific Missile Range Facility in February 2016: satellite tagging, phot-identification, and passive acoustic monitoring. .

Baird, R. W., and coauthors. 2016. Final Report: Odontocete Studies on the Pacific Missile Range Facility in February 2015: Satellite-Tagging, Photo-Identification, and Passive Acoustic Monitoring. In U. S. P. F. Commander (Ed.), Naval Facilities Engineering Command Pacific under HDR Environmental, Operations and Construction, Inc. Contract No. N62470-10-D-3011, CTO KB28. Olympia, WA: HDR Inc.

Baker, C. S., and P. J. Clapham. 2004. Modelling the past and future of whales and whaling. Trends in Ecology and Evolution 19(7):365-371.

Baker, C. S., and L. M. Herman. 1983. Seasonal contrasts in the social behavior of the North Pacific humpback whale. Pages 5 *in* Fifth Biennial Conference on the Biology of Marine Mammals, New England Aquarium, Boston, Massachusetts.

Baker, C. S., and L. M. Herman. 1989. Behavioral responses of summering humpback whales to vessel traffic: Experimental and opportunistic observations (*Megaptera novaeangliae*). Tech. Rep. No. NPS-NR-TRS-89-01. 50 pgs. Final report to the National Park Service, Alaska Regional Office, Anchorage, Alaska [Available from the U.S. Dept. Interior, NPS, Alaska Reg. Off., Room 107, 2525 Gambell St., Anchorage, AK 99503.

Baker, C. S., L. M. Herman, B. G. Bays, and G. B. Bauer. 1983. The impact of vessel traffic on the behavior of humpback whales in southeast Alaska: 1982 season. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, National Marine Mammal Laboratory.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research 2:21-30.

Barbieri, E. 2009. Concentration of heavy metals in tissues of green turtles (*Chelonia mydas*) sampled in the Cananéia Estuary, Brazil. Brazilian Journal of Oceanography 57(3):243-248.

Barlow, J. 1994. Abundance of large whales in California coastal waters: A comparison of ship surveys in 1979/80 and in 1991. Report of the International Whaling Commission 44:399-406.

Barlow, J. 1995. Abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Fishery Bulletin 93(1):1-14.

Barlow, J. 1997. Preliminary estimates of cetacean abundance off California, Oregon, and Washington based on a 1996 ship survey and comparisons of passing and closing modes. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Admin. Rept. LJ-97- 11, La Jolla, California.

Barlow, J. 2003. Preliminary estimates of the abundance of cetaceans along the U.S. West Coast: 1991-2001. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center LJ-03-03.

Barlow, J. 2010. Cetacean Abundance in the California Current Estimated from a 2008 Ship-Based Line-Transect Survey. Southwest Fisheries Science Center, La Jolla, CA.

Barlow, J. 2016. Cetacean Abundance in the California Current Estimated from Ship-based Line-transect Surveys in 1991–2014. Southwest Fisheries Science Center, La Jolla, CA.

Barlow, J., and coauthors. 2011. Humpback whale abundance in the North Pacific estimated by photographic capture-recapture with bias correction from simulation studies. Marine Mammal Science 27(4):793–818.

Barlow, J., and coauthors. 1997. U.S. Pacific Marine Mammal Stock Assessments: 1996 Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, NOAA-TM-NMFS-SWFSC-248.

Barlow, J., and B. L. Taylor. 2001. Estimates of large whale abundance off California, Oregon, Washington, and Baja California based on 1993 and 1996 ship surveys. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, LJ-01-03.

Barnard, B. 2016. Carriers stick with slow-steaming despite fuel-price plunge. The Journal of Commerce. The JOC Group Inc., New York, NY.

Barnhart, R. A. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest): steelhead. . Technical Report, TR-EL-82-4/872-11-60. Humboldt State University, Arcada, California.

Bartholomew Jr., G. A. 1950. A male Guadalupe fur seal on San Nicholas Island, California. Journal of Mammalogy 31(2):175-180.

Bartol, S. M., J. A. Musick, and M. Lenhardt. 1999. Auditory Evoked Potentials of the Loggerhead Sea Turtle (*Caretta caretta*). Copeia 3:836-840.

Bass, A. L. 2010. Juvenile coho salmon movement and migration through tide gates.

Bassett, C., B. Polagye, M. M. Holt, and J. Thomson. 2012. A vessel noise budget for Admiralty Inlet, Puget Sound, Washington (USA). Journal of the Acoustical Society of America 6(132):3706-3719.

Bassett, C., J. Thomson, and B. Polagye. 2010. Characteristics of underwater ambient noise at a proposed tidal energy site in Puget Sound. Pages 8 *in* Oceans 2010 MTS/IEEE Conference, Seattle, Washington.

Bauer, G., and L. M. Herman. 1986. Effects of vessel traffic on the behavior of humpback whales in Hawaii. National Marine Fisheries Service, Honolulu, Hawaii.

Bauer, G. B. 1986. The behavior of humpback whales in Hawaii and modifications of behavior induced by human interventions. (Megaptera novaeangliae). University of Hawaii. 314p.

Baulch, S., and C. Perry. 2014. Evaluating the impacts of marine debris on cetaceans. Marine Pollution Bulletin 80(1-2):210-221.

Baumann-Pickering, S., and coauthors. 2012. Appendix; Marine species monitoring for the U.S. Navy's Gulf of Alaska temporary maritime activities area annual report 2012.

Baylis, H. A. 1928. Parasites of whales. Natural History Magazine 1(2):55-57.

Beale, C. M., and P. Monaghan. 2004. Human disturbance: people as predation-free predators? Journal of Applied Ecology 41:335-343.

Beamer, E., A. McBride, R. Henderson, and K. Wolf. 2003. The importance of non-natal pocket estuaries in Skagit Bay to wild Chinook salmon: an emerging priority for restoration. Skagit System Cooperative Research Department.

Beamer, E. M., B. Hayman, and D. Smith. 2005. Appendix C: Linking freshwater habitat to Skagit Chinook salmon recovery. Skagit River System Cooperative and Washington Department of Fish and Wildlife.

Beamesderfer, R. C., M. L. Simpson, and G. J. Kopp. 2007. Use of life history information in a population model for Sacramento green sturgeon. Environmental Biology of Fishes 79(3-4):315-337.

Beamish, R. J. 1993. Climate and exceptional fish production off the west coast of North American. Canadian Journal of Fisheries and Aquatic Sciences 50(10):2270-2291.

Beamish, R. J., M. Folkes, R. Sweeting, and C. Mahnken. 1998. Intra-annual changes in the abundance of coho, Chinook, and chum salmon in Puget Sound in 1997. Puget Sound Research'98, Puget Sound Water Quality Action Team, Olympia, Washington.

Becker, E. A., and coauthors. 2016. Moving Towards Dynamic Ocean Management: How Well Do Modeled Ocean Products Predict Species Distributions? Remote Sensing 8(2):149.

Becker, E. A., and coauthors. 2018. Predicting cetacean abundance and distribution in a changing climate. Diversity and Distributions 25(4):626-643.

Becker, E. A., and coauthors. 2017. Habitat-Based Density Models for Three Cetacean Species off Southern California Illustrate Pronounced Seasonal Differences. Frontiers in Marine Science 4(121):1–14.

Bejder, L., S. M. Dawson, and J. A. Harraway. 1999. Responses by Hector's dolphins to boats and swimmers in Porpoise Bay, New Zealand. Marine Mammal Science 15(3):738-750.

Bejder, L., and D. Lusseau. 2008. Valuable lessons from studies evaluating impacts of cetacean-watch tourism. Bioacoustics 17-Jan(3-Jan):158-161. Special Issue on the International Conference on the Effects of Noise on Aquatic Life. Edited By A. Hawkins, A. N. Popper & M. Wahlberg.

Bejder, L., A. Samuels, H. Whitehead, H. Finn, and S. Allen. 2009. Impact assessment research: Use and misuse of habituation, sensitisation and tolerance to describe wildlife responses to anthropogenic stimuli. Marine Ecology Progress Series 395:177-185.

Bejder, L., and coauthors. 2019. Low energy expenditure and resting behaviour of humpback whale mother-calf pairs highlights conservation importance of sheltered breeding areas. Scientific Reports 9(1):771.

Belcher, R. I., and T. E. Lee, Jr. 2002. *Arctocephalus townsendi*. Mammalian Species 700:1–5.

Belcher, R. L., and T.E. Lee, Jr. 2002. Arctocephalus townsendi. Mammalian Species 700(1):1-5.

Bellinger, M. R., and coauthors. 2015. Geo-Referenced, Abundance Calibrated Ocean Distribution of Chinook Salmon (Oncorhynchus tshawytscha) Stocks across the West Coast of North America. PLoS ONE 10(7).

Benson, A., and A. W. Trites. 2002. Ecological effects of regime shifts in the Bering Sea and eastern North Pacific Ocean. Fish and Fisheries 3(2):95-113.

Benson, S. R., and coauthors. 2007. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology 6(1):150-154.

Benson, S. R., and coauthors. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere 2(7).

Berchok, C. L., D. L. Bradley, and T. B. Gabrielson. 2006. St. Lawrence blue whale vocalizations revisited: Characterization of calls detected from 1998 to 2001. Journal of the Acoustical Society of America 120(4):2340–2354.

Bergman, P., J. Merz, and B. Rook. 2011. Memo: green sturgeon observations at Daguerre Point Dam, Yuba River, CA. Auburn (CA): Cramer Fish Sciences.

Bergmann, M., L. Gutow, and M. Klages. 2015. Marine anthropogenic litter. Springer.

Bergström, L., and coauthors. 2014. Effects of offshore wind farms on marine wildlife—a generalized impact assessment. Environmental Research Letters 9(3):034012.

Berman-Kowalewski, M., and coauthors. 2010. Association between blue whale (*Balaenoptera musculus*) mortality and ship strikes along the California coast. Aquatic Mammals 36(1):59-66.

Bernaldo De Quiros, Y., and coauthors. 2012. Decompression vs. decomposition: Distribution, amount, and gas composition of bubbles in stranded marine mammals. Frontiers in Zoology 3:177.

Bernaldo De Quiros, Y., and coauthors. 2013. Compositional discrimination of decompression and decomposition gas bubbles in bycaught seals and dolphins. PLoS ONE 8(12):e83994.

Bernardi, G., S. R. Fain, J. P. Gallo-Reynoso, A. L. Figueroa-Carranza, and B. J. Le Boeuf. 1998. Genetic variability in Guadalupe fur seals. Journal of Heredity 89(4):301-305.

Berzin, A. A. 1971. The sperm whale. (*Physeter macrocephalus*). Pishchevaya Promyshlennost Moscow, NTIS No. TT-71-50152.

Best, B. D., C. H. Fox, R. Williams, P. N. Halpin, and P. C. Paquet. 2015. Updated marine mammal distribution and abundance estimates in British Columbia. Journal of Cetacean and Research Management 15:9–26.

Best, P. B., P. A. S. Canham, and N. Macleod. 1984. Patterns of reproduction in sperm whales, *Physeter macrocephalus*. Report of the International Whaling Commission Special Issue 6:51-79. Reproduction in Whales, Dolphins and Porpoises. Proceedings of the

Conference Cetacean Reproduction Estimating Parameters For stock Assessment and Management.

Best, P. B., and C. H. Lockyer. 2002. Reproduction, growth and migrations of sei whales *Balaenoptera borealis* off the west coast of South Africa in the 1960s. South African Journal of Marine Science 24:111-133.

Bester, M. N., J. W. H. Ferguson, and F. C. Jonker. 2002. Population densities of pack ice seals in the Lazarev Sea, Antarctica. Antarctic Science 14(2):123–127.

Bettridge, S., and coauthors. 2015a. Status Review of the Humpback Whale (*Megaptera novaeangliae*) under the Endangered Species Act. Southwest Fisheries Science Center, La Jolla, CA.

Bettridge, S. O. M., and coauthors. 2015b. Status review of the humpback whale (Megaptera novaeangliae) under the Endangered Species Act.

Bi, H., W. T. Peterson, J. Lamb, and E. Casilas. 2011. Copepods and salmon: characterizing the spatial distribution of juvenile salmon along the Washington and Oregon coast, USA. Fisheries Oceanography 20:125-138.

Bi, H., R. E. Ruppel, and W. T. Peterson. 2007. Modeling the pelagic habitat of salmon off the Pacific Northwest (USA) coast using logistic regression. Marine Ecology Progress Series 336:249-265.

Bickham, J. W., T. R. Loughlin, J. K. Wickliffe, and V. N. Burkanov. 1998. Geographic variation in the mitochondrial DNA of Steller sea lions: Haplotype diversity and endemism in the Kuril Islands. Biosphere Conservation 1(2):107-117.

Biggs, D. C., R. R. Leben, and J. G. Ortega-Ortiz. 2000. Ship and satellite studies of mesoscale circulation and sperm whale habitats in the northeast Gulf of Mexico during GulfCet II. Gulf of Mexico Science 18(1):15-22.

Binckley, C. A., J. R. Spotila, K. S. Wilson, and F. V. Paladino. 1998. Sex determination and sex ratios of Pacific leatherback turtles, *Dermochelys coriacea*. Copeia 2(291-300).

Bjorkstedt, E. P., and coauthors. 2005. An analysis of historical population structure for evolutionarily significant units of Chinook salmon, coho salmon, and steelhead in the North-Central California Coast Recovery Domain. U.S. Department of Commerce, NOAA-TM-NMFS-SWFSC-382.

Bjorndal, K. A. 1997. Foraging ecology and nutrition of sea turtles. Pages 199–231 *in* The biology of sea turtles. CRC Press, Boca Raton, Florida.

Black, N. A. 2011. Fish-eating (Resident) Killer Whales Sighted in Monterey Bay on Feb. 10, 2011. Monterey Bay Whale Watch.

Blackwell, S. B., J. W. Lawson, and M. T. Williams. 2004. Tolerance by ringed seals (*Phoca hispida*) to impact pipe-driving and construction sounds at an oil production island. The Journal of the Acoustical Society of America 115(5 [Pt. 1]):2346–2357.

Blackwell, S. B., and coauthors. 2013. Effects of airgun sounds on bowhead whale calling rates in the Alaskan Beaufort Sea. Marine Mammal Science.

Blackwell, S. B., and coauthors. 2015. Effects of airgun sounds on bowhead whale calling rates: evidence for two behavioral thresholds. PLoS ONE 10(6):e0125720.

Blane, J. M., and R. Jaakson. 1994. The impact of ecotourism boats on the St. Lawrence beluga whales (*Delphinapterus leucas*). Environmental Conservation 21(3):267-269.

Bleakney, J. S. 1965. Reports of Marine Turtles from New England and Eastern Canada. The Canadian Field-Naturalist 79:120-128.

Blum, J. P. 1988. Assessment of factors affecting sockeye salmon (*Oncorhynchus nerka*) production in Ozette Lake, WA. Masters Thesis, University of Washington, Seattle, Washington.

Blundell, G. M., and G. W. Pendleton. 2015. Factors affecting haul-out behavior of harbor seals (*Phoca vitulina*) in tidewater glacier inlets in Alaska: Can tourism vessels and seals coexist? PLoS ONE 10(5):e0125486.

Bonney, J., and P. T. Leach. 2010. Slow Boat From China. Maritime News.

Boreman, J. 1997. Sensitivity of North American sturgeons and paddlefish to fishing mortality. Environmental Biology of Fishes 48(1-4):399-405.

Boreman, J., W.J. Overholtz, and M.P. Sissenwine. 1984. A preliminary analysis of the effects of fishing on shortnose sturgeon.

Borin, J. M., and coauthors. 2017. Energetic requirements of green sturgeon (Acipenser medirostris) feeding on burrowing shrimp (Neotrypaea californiensis) in estuaries: importance of temperature, reproductive investment, and residence time. Environmental Biology of Fishes 100(12):1561-1573.

Borrell, A. 1993. PCB and DDTs in blubber of cetaceans from the northeastern North Atlantic. Marine Pollution Bulletin 26(3):146.

Borrell, A., and A. Aguilar. 1987. Variations in DDE percentage correlated with total DDT burden in the blubber of fin and sei whales. Marine Pollution Bulletin 18(2):70-74.

Bostrom, B. L., T. T. Jones, M. Hastings, and D. R. Jones. 2010. Behaviour and Physiology: The Thermal Strategy of Leatherback Turtles. PLoS ONE 5(11):e13925.

Bousman, W. G., and R. M. Kufeld. 2005. UH-60A Airloads Catalog. National Aeronautics and Space Administration, Moffett Field, CA.

Bowles, A. E., M. Smultea, B. Wursig, D. P. Demaster, and D. Palka. 1994. Relative abundance and behavior of marine mammals exposed to transmissions from the Heard Island feasibility test. Journal of the Acoustical Society of America 96(4):2469-2484.

Boyd, I., D. Claridge, C. Clark, B. Southall, and P. Tyack. 2008. Behavioral Response Study 2007 BRS-07 cruise report.

Branch, T. A. 2007. Abundance of Antarctic blue whales south of 60 S from three complete circumpolar sets of surveys. Journal of Cetacean Research and Management 9(3):253–262.

Brandt, M. J., A. Diederichs, K. Betke, and G. Nehls. 2011. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Marine Ecology Progress Series 421:205–216.

Branstetter, B. K., and J. J. Finneran. 2008. Comodulation masking release in bottlenose dolphins (Tursiops truncatus). Journal of the Acoustical Society of America 124(1):625-633.

Branstetter, B. K., and coauthors. 2017a. Killer whale (Orcinus orca) behavioral audiograms. The Journal of the Acoustical Society of America 141(4):2387-2398.

Branstetter, B. K., and coauthors. 2017b. Killer whale (*Orcinus orca*) behavioral audiograms. The Journal of the Acoustical Society of America 141:2387-2398.

Brennan, J. S., K. F. Higgins, J. R. Cordell, and V. A. Stamatiou. 2004. Juvenile salmon composition, timing, distribution, and diet in the marine nearshore waters of central Puget Sound in 2001-2002 King County Department of Natural Resources and Parks, Seattle, Washington:164 p.

Briggs, C., S. M. Shjegstad, J. Silva, and M. Edwards. 2016. Distribution of chemical warfare agent, energetics, and metals in sediments at a deep-water discarded military munitions site. Deep Sea Research Part II: Topical Studies in Oceanography, 128, 63–69.

Brocksen, R. W., and H. T. Bailey. 1973. Respiratory Response of Juvenile Chinook Salmon and Striped Bass Exposed to Benzene, a Water-Soluble Component of Crude Oil. International Oil Spill Conference Proceedings 1973(1):783-791.

Brown, L. R., P. B. Moyle, and R. M. Yoshiyama. 1994. Historical decline and current status of coho salmon in California. North American Journal of Fisheries Management 14(2):237-261.

Brownell Jr., R. L., P. J. Clapham, T. Miyashita, and T. Kasuya. 2001. Conservation status of North Pacific right whales. Journal of Cetacean Research and Management (Special Issue 2):269-286.

Brownell Jr., R. L., A. R. Lang, A. M. Burdin, A. B. Bradford, and D. W. Weller. 2009. The western gray whale population is distinct: A response to SC/61/BRG22. International Whaling Commission Scientific Committee, Madeira, Portugal.

Browning, L. J., and E. J. Harland. 1999. Are bottlenose dolphins disturbed by fast ferries? European Research on Cetaceans 13:92-98. Proceedings of the thirteenth Annual Conference of the European Cetacean Society. P. G. H. Evans, J. Cruz & J. A. Raga-Eds.). Valencia, Spain, 5-8 April.

Brumm, H., and H. Slabbekoorn. 2005. Acoustic communication in noise. Advances in the Study of Behavior 35:151-209.

Bryant, P. J., C. M. Lafferty, and S. K. Lafferty. 1984. Reoccupation of Laguna Guerrero Negro, Baja California, Mexico, by gray whales. (*Eschrichtius robustus*). M. L. Jones, S. L. Swartz, and S. Leatherwood, editors. The Gray Whale, *Eschrichtius robustus*. Academic Press, New York.

Buckman, A. H., and coauthors. 2011. PCB-Associated Changes in mRNA Expression in Killer Whales (Orcinus orca) from the NE Pacific Ocean. Environmental Science & Technology 45(23):10194-10202.

Budelmann, B. U. 2010. Cephalopoda. R. Hubrecht, and J. Kirkwood, editors. The UFAW Handbook on the Care and Management of Laboratory and Other Research Animals, Eighth Edition. Wiley Blackwell, Oxford, UK.

Bullock, T. H., D. A. Bodznick, and R. G. Northcutt. 1983. The phylogenetic distribution of electroreception - evidence for convergent evolution of a primitive vertebrate sense modality. Brian Research Reviews 6(1):25-46.

Burdin, A. M., A. L. Bradford, G. A. Tsidulko, and M. Sidorenko. 2011. Status of western gray whales off northeastern Sakhalin Island and eastern Kamchatka, Russia in 2010. International Whaling Commission-Scientific Committee, Tromso, Norway.

Burgner, R. L. 1991. The life history of sockeye salmon (Oncorhynchus nerka). C. a. L. Margolis, editor. Life history of Pacific salmon. University of British Columbia Press, Vancouver, British Columbia, Canada.

Burke, B. J., M. C. Liermann, D. J. Teel, and J. J. Anderson. 2013. Environmental and geospatial factors drive juvenile Chinook salmon distribution during early ocean migration. NRC Research Press 70:1167-1177.

Burtenshaw, J. C., and coauthors. 2004. Acoustic and satellite remote sensing of blue whale seasonality and habitat in the Northeast Pacific. Deep-Sea Research Part Ii-Topical Studies in Oceanography 51(10-11):967-986.

Busby, P. J., and coauthors. 1996. Status review of steelhead from Washington, Oregon, and California. U.S. Department of Commerce, Northwest Fisheries Science Center, NMFS-NWFSC-27, Seattle, Washington.

Busch, D. S., C. J. Harvey, and P. McElhany. 2013. Potential impacts of ocean acidification on the Puget Sound food web. ICES Journal of Marine Science 70(4):823-833.

Byron, C. J., and B. J. Burke. 2014. Salmon ocean migration models suggest a variety of population-specific strategies. Reviews in Fish Biology and Fisheries 24(3):737-756.

Calambokidis, J. 2012. Summary of ship-strike related research on blue whales in 2011.

Calambokidis, J., and J. Barlow. 2004. Abundance of blue and humpback whales in the eastern North Pacific estimated by capture-recapture and line-transect methods. Marine Mammal Science 20(1):63–85.

Calambokidis, J., and J. Barlow. 2013. Updated Abundance Estimates of Blue and Humpback Whales off the U.S. West Coast Incorporating Photo-Identifications from 2010 and 2011. Cascadia Research and Southwest Fisheries Science Center, Olympia, WA and La Jolla, CA.

Calambokidis, J., J. Barlow, K. Flynn, E. Dobson, and G. H. Steiger. 2017. Update on abundance, trends, and migrations of humpback whales along the U.S. West Coast. International Whaling Commission, Cambridge, United Kingdom.

Calambokidis, J., J. Barlow, J. K. B. Ford, T. E. Chandler, and A. B. Douglas. 2009a. Insights into the population structure of blue whales in the eastern North Pacific from recent sightings and photographic identification. Marine Mammal Science 25(4):816-832.

Calambokidis, J., T. Chandler, E. Falcone, and A. Douglas. 2004a. Research on Large Whales off California, Oregon, and Washington in 2003. Cascadia Research, Contract number 50ABNF100065, Olympia, WA.

Calambokidis, J., and T. Chandler. 2000. Marine mammal observations and mitigation associated with USGS seismic surveys in the Southern California Bight in 2000. Cascadia Research Collective report. Prepared for the U.S. Geological Survey. 13pp.

Calambokidis, J., and coauthors. 2002. Abundance, range and movements of a feeding aggregation of gray whales (*Eschrichtius robustus*) from California to southeastern Alaska in 1998. Journal of Cetacean Research and Management 4(3):267-276.

Calambokidis, J., and coauthors. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Cascadia Research, Olympia, WA.

Calambokidis, J., J. Laake, and A. Perez. 2017c. Updated analysis of abundance and population structure of seasonal gray whales in the Pacific Northwest, 1996–2015. International Whaling Commission, Cambridge, United Kingdom.

Calambokidis, J., J. L. Laake, and A. Klimek. 2010. Abundance and Population Structure of Seasonal Gray Whales in the Pacific Northwest, 1998–2008. International Whaling Commission Scientific Committee, Washington, DC.

Calambokidis, J., B. R. Mate, E. M. Oleson, L. Irvine, and C. E. H. A. B. Douglas. 2009b. The Costa Rica Dome as a winter feeding and breeding areas for North Pacific blue whales. Pages 47 *in* Eighteenth Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada.

Calambokidis, J., and coauthors. 2003. Feeding and vocal behavior of blue whales determined through simultaneous visual-acoustic monitoring and deployment of suction-cap attached tags. Pages 27 *in* Abstracts of the 15th Biennial Conference on the Biology of Marine Mammals, Greensboro, North Carolina.

Calambokidis, J., and coauthors. 1990. Sightings and movements of blue whales off central California 1986-88 from photo-identification of individuals. Report of the International Whaling Commission (Special Issue 12):343-348.

Calambokidis, J., and coauthors. 2015. Biologically Important Areas for Selected Cetaceans Within U.S. Waters – West Coast Region. Aquatic Mammals (Special Issue) 41(1):39–53.

Calambokidis, J., G. H. Steiger, D. K. Ellifrit, B. L. Troutman, and C. E. Bowlby. 2004b. Distribution and abundance of humpback whales (*Megaptera novaeangliae*) and other marine mammals off the northern Washington coast. Fishery Bulletin 102(4):563-580.

Calambokidis, J., and coauthors. 1996. Interchange and isolation of humpback whales off California and other North Pacific feeding grounds. Marine Mammal Science 12(2):215-226.

Calkins, D. G. 1986. Marine Mammals. Pages 527−558 *in* D. W. Hood, and S. T. Zimmerman, editors. In The Gulf of Alaska, Physical Environment and Biological Resources. Government Printing Office, Washington, D.C.

Callier, M. D., and coauthors. 2018. Attraction and repulsion of mobile wild organisms to finfish and shellfish aquaculture: a review. Reviews in Aquaculture 10.4:924-949.

Campbell, G. S., and coauthors. 2015. Inter-annual and seasonal trends in cetacean distribution, density and abundance off southern California. Deep Sea Research Part II: Topical Studies in Oceanography 112:143-157.

Campbell, G. S., D. W. Weller, and J. A. Hildebrand. 2010. SIO small boat based marine mammal surveys in Southern California: Report of Results for August 2009–July 2010: Annual Range Complex Monitoring Report for Hawaii and Southern California. Draft submission to NMFS 15 September 2010. U.S. Department of the Navy.

Canadian Press. 2007. Most of B.C. diesel slick dissipated: Coast Guard. The Globe and Mail, Toronto, Canada.

Candy, J. R., and coauthors. 2015. Population differentiation determined from putative neutral and divergent adaptive genetic markers in Eulachon (Thaleichthys pacificus, Osmeridae), an anadromous Pacific smelt. Molecular Ecology Resources 15(6):1421-1434.

Carder, D. A., and S. Ridgway. 1990. Auditory brainstem response in a neonatal sperm whale. Journal of the Acoustic Society of America 88(Supplement 1):S4.

Carr, A. 1987. Impact of nondegradable marine debris on the ecology and survival outlook of sea turtles. Marine Pollution Bulletin 18(6):352-356.

Carretta, J., and coauthors. 2002. U.S. Pacific marine mammal stock assessments: 2002. U.S. Department of Commerce, NMFS-SWFSC-346.

Carretta, J. V., and J. Barlow. 2008. Acousitc pingers eliminate beaked whale bycatch in a gill net fishery. Marine Mammal Science 24(4):2053-2073.

Carretta, J. V., and J. Barlow. 2011. Long-term effectiveness, failure rates, and "dinner bell" properties of acoustic pingers in a gillnet fishery. Marine Technology Society Journal 45(5):7–19.

Carretta, J. V., J. Barlow, K. A. Forney, M. M. Muto, and J. Baker. 2001. U.S. Pacific marine mammal stock assessments: 2001. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA-TM-NMFS-SWFSC-317.

Carretta, J. V., and S. J. Chivers. 2004. Preliminary estimates of marine mammal mortality and biological sampling of cetaceans in California gillnet fisheries for 2003. Unpublished paper to the IWC Scientific Committee. 20 pp. Sorrento, Italy, July (SC/56/SM1).

Carretta, J. V., and coauthors. 2007. U.S. Pacific marine mammal stock assessments: 2007.

Carretta, J. V., and coauthors. 2010. U.S. Pacific marine mammal stock assessments: 2009. NOAA.

Carretta, J. V., and coauthors. 2005. U.S. Pacific marine mammal stock assessments: 2004. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, NOAA-TM-NMFS-SWFSC-358.

Carretta, J. V., and coauthors. 2012. U.S. Pacific Marine Mammal Stock Assessments: 2011. Southwest Fisheries Science Center, La Jolla, CA.

Carretta, J. V., and coauthors. 2020. U.S. Pacific Marine Mammal Stock Assessments: 2019. U. S. D. o. Commerce, editor.

Carretta, J. V., and coauthors. 2018a. U.S. Pacific Marine Mammal Stock Assessments: 2017. Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, NOAA-TM-NMFS-SWFSC-602, La Jolla, California.

Carretta, J. V., and coauthors. 2018b. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2012-2016. U. S. D. o. Commerce, editor.

Carretta, J. V., K.A. Forney, E.M. Oleson, D.W. Weller, A.R. Lang, J. Baker, M.M. Muto, B. Hanson, A.J. Orr, H. Huber, M.S. Lowry, J. Barlow, J.E. Moore, D. Lynch, L. Carswell, R.L. Brownell. 2019. U.S. Pacific Marine Mammal Stock Assessments: 2018. U.S. Department of Commerce.

Carretta, J. V., and coauthors. 2017a. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2011–2015. Southwest Fisheries Science Center, La Jolla, CA.

Carretta, J. V., and coauthors. 2016a. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2010-2014. Southwest Fisheries Science Center, La Jolla, California.

Carretta, J. V., and coauthors. 2016b. Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2010–2014. Southwest Fisheries Science Center, La Jolla, CA.

Carretta, J. V., and coauthors. 2013. U.S. Pacific marine mammal stock assessments: 2012. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., and coauthors. 2016c. U.S. Pacific marine mammal stock assessments: 2015.

Carretta, J. V., and coauthors. 2017b. U.S. Pacific Marine Mammal Stock Assessments: 2017. Southwest Fisheries Science Center, La Jolla, California.

Carretta, J. V., and coauthors. 2017c. U.S. Pacific Marine Mammal Stock Assessments: 2017. Southwest Fisheries Science Center, La Jolla, CA.

Carroll, A. G., R. Przeslawski, A. Duncan, M. Gunning, and B. Bruce. 2017. A critical review of the potential impacts of marine seismic surveys on fish and invertebrates. Marine Pollution Bulletin 114:9-24.

Cascadia Research. 2012. Record number of blue whales off Washington in December 2011.

Cascadia Research. 2017a. Examination of fin whale reveals it was killed by collision with ship.

Cascadia Research. 2017b. Return of humpback whales to the Salish Sea. Cascadia Research Collective, Olympia, WA.

Cascadia Research. 2017c. "Sounders", the local gray whales that feed in northern Puget Sound each spring, staying longer than usual and update on boat-struck gray whale.

Castellote, M., C. W. Clark, and M. O. Lammers. 2012. Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. Biological Conservation 147(1):115-122.

Cates, K., and A. Acevedo-Gutiérrez. 2017. Harbor Seal (*Phoca vitulina*) tolerance to vessels under different levels of boat traffic. Aquatic Mammals 43(2):193–200.

CDFG. 2003. September 2002 Klamath River fish kill: preliminary analysis of contributing factors. ?????? ??????:??-??

CDFW. 2000. Lower American River Pilot Salmon and Steelhead Spawning Habitat Improvement Project. Quarterly Status Report July 1999-March 2000.

Celewycz, A. G., E. A. Fergusson, J. H. Moss, and J. A. Orsi. 2014. High seas salmonid coded-wire tag recovery data, 2013. N. NOAA, Alaska Fish. Sci. Cent., Auke Bay Laboratories, Ted Stevens Marine Res. Institute, Northwest Fish. Sci. Cent., editor. North Pacific Anadromous Fish Commission, Juneau, AK.

Cerchio, S., S. Strindberg, T. Collins, C. Bennett, and H. Rosenbaum. 2014. Seismic surveys negatively affect humpback whale singing activity off northern Angola. PLoS ONE 9(3):e86464.

Chaloupka, M., P. Dutton, and H. Nakano. 2004. Status of sea turtle stocks in the Pacific. Pages 135-164 *in* Expert Consultation on Interactions Between Sea Turtles and Fisheries within an Ecosystem Context, volume FAO Fisheries Report. No. 738, Rome.

Chamberlin, J. W., A. N. Kagley, K. L. Fresh, and T. P. Quinn. 2011. Movements of Yearling Chinook Salmon during the First Summer in Marine Waters of Hood Canal, Washington. Transactions of the American Fisheries Society 140(2):429-439.

Chapman, D. J., and B. E. Julius. 2005. The use of preventative projects as compensatory restoration. Journal of Coastal Research 40(Special Issue):120-131.

Chapman, D. W. 1962. Aggressive behavior in juvenile coho salmon as a cause of emigration. Journal of the Fisheries Board of Canada 19(6):1047-1080.

Chapman, N. R., and A. Price. 2011. Low frequency deep ocean ambient noise trend in the Northeast Pacific Ocean. Journal of the Acoustical Society of America 129(5):EL161-EL165.

Charif, R., and coauthors. 2015. Development of Statistical Methods for Assessing Changes in Whale Vocal Behavior in Response to Mid-Frequency Active Sonar. Final Report. Prepared for U.S. Fleet Forces Command. Submitted to Naval Facilities Engineering Command Atlantic, Norfolk, Virginia, under Contract No. N62470-10-3011, Task Order 39, issued to HDR Inc., Virginia Beach, Virginia. 20 March 2015.

Charif, R. A., D. K. Mellinger, K. J. Dunsmore, K. M. Fristrup, and C. W. Clark. 2002. Estimated source levels of fin whale (*Balaenoptera physalus*) vocalizations: Adjustments for surface interference. Marine Mammal Science 18(1):81-98.

Chasco, B. 2019. Dissertation Abstract: Combining Disparate Data Sources with Ecosystem and Integrated Models: Case Studies for Improving Management of Protected Species. Oregon State University.

Chasco, B., and coauthors. 2017a. Estimates of Chinook salmon consumption in Washington State inland waters by four marine mammal predators from 1970 to 2015. Canadian Journal of Fisheries and Aquatic Sciences 74(8):1173-1194.

Chasco, B. E., and coauthors. 2017b. Competing tradeoffs between increasing marine mammal predation and fisheries harvest of Chinook salmon. Scientific Reports 7:14.

Cheung, W. W. L., R. D. Brodeur, T. A. Okey, and D. Pauly. 2015. Projecting future changes in distributions of pelagic fish species of Northeast Pacific shelf seas. Progress in Oceanography 130:19-31.

Childers, A. R., T. E. Whitledge, and D. A. Stockwell. 2005. Seasonal and interannual variability in the distribution of nutrients and chlorophyll a across the Gulf of Alaska shelf: 1998-2000. Deep-Sea Research II 52:193-216.

Cholewiak, D., and coauthors. 2018. Communicating amidst the noise: Modeling the aggregate influence of ambient and vessel noise on baleen whale communication space in a national marine sanctuary. Endangered Species Research 36:59–75.

Cholewiak, D., A. I. Deangelis, D. Palka, P. J. Corkeron, and S. M. Van Parijs. 2017. Beaked whales demonstrate a marked acoustic response to the use of shipboard echosounders. Royal Society Open Science 4(12).

Christal, J., and H. Whitehead. 1997. Aggregations of mature male sperm whales on the Galápagos Islands breeding ground. Marine Mammal Science 13(1):59-69.

Christal, J., H. Whitehead, and E. Lettevall. 1998. Sperm whale social units: Variation and change. Canadian Journal of Zoology 76(8):1431-1440.

Christian, E., and J. Gaspin. 1974. Swimmer Safe Standards from Underwater Explosions. Navy Science Assistance Program Project No. PHP-11-73. White Oak, MD: Naval Ordnance Laboratory.

Christiansen, F., A. M. Dujon, K. R. Sprogis, J. P. Y. Arnould, and L. Bejder. 2016. Noninvasive unmanned aerial vehicle provides estimates of the energetic cost of reproduction in humpback whales. Ecosphere, 7(10), e01468. .

Christiansen, F., D. Lusseau, E. Stensland, and P. Berggren. 2010. Effects of tourist boats on the behaviour of Indo-Pacific bottlenose dolphins off the south coast of Zanzibar. Endangered Species Research 11(1):91-99.

Christiansen, F., M. Rasmussen, and D. Lusseau. 2013. Whale watching disrupts feeding activities of minke whales on a feeding ground. Marine Ecology Progress Series 478:239-251.

Christiansen, F., M. H. Rasmussen, and D. Lusseau. 2011. Whalewatching boats disrupt the foraging activities of minke whales in Faxaflói bay, Iceland. Pages 61 *in* Nineteenth Biennial Conference on the Biology of Marine Mammals, Tampa, Florida.

Chua, T. H. 1988. On the road to extinction: The leatherback turtle (*Dermochelys coriacea*) in Terengganu, Malaysia. Pages 153-158 *in* A. Sasekumar, R. D'Cruz, and S. L. H. Lim, editors. Eleventh Annual Seminar, Malaysian Society of Marine Sciences, Kuala Lumpur, Malaysia.

Clapham, P. J. 1994. Maturational changes in patterns of association in male and female humpback whales, Megaptera novaeangliae. Journal of Zoology 234(2):265-274.

Clapham, P. J. 1996. The social and reproductive biology of humpback whales: an ecological perspective. Mammal Review 26:27-49.

Clapham, P. J., S. B. Young, and R. L. Brownell Jr. 1999. Baleen whales: Conservation issues and the status of the most endangered populations. Mammal Review 29(1):35-60.

Claridge, D., and J. Durban. 2009. Monitoring beaked whale movements during submarine commanders course using satellite telemetry tags. Office of Naval Research.

Claridge, D. E., C. A. Dunn, and J. W. Durban. 2009. Photographic mark-recapture reveals turnover of beaked whales on an active sonar range. Pages 57 *in* Eighteenth Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada.

Clark, C. 2006. Acoustic communication in the great whales: The medium and the message. Presentation at the 86th Annual Conference of the American Society of Mammalogists.

Clark, C., and coauthors. 2009. Acoustic masking of baleen whale communications: Potential impacts from anthropogenic sources. Pages 56 *in* Eighteenth Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada.

Clark, C. W., J. F. Borsani, and G. Notarbartolo-di-Sciara. 2002. Vocal activity of fin whales, *Balaenoptera physalus*, in the Ligurian Sea. Marine Mammal Science 18(1):286-295.

Clark, C. W., and P. J. Clapham. 2004. Acoustic monitoring on a humpback whale (Megaptera novaeangliae) feeding ground shows continual singing into late spring. Proceedings of the Royal Society of London Series B Biological Sciences 271(1543):1051-1057.

Clark, C. W., and K. M. Fristrup. 2001. Baleen whale responses to low-frequency human-made underwater sounds. Journal of the Acoustical Society of America 110(5 Part 2):2751.

Clark, C. W., and G. J. Gagnon. 2004. Low-frequency vocal behaviors of baleen whales in the North Atlantic: Insights from Integrated Undersea Surveillance System detections, locations, and tracking from 1992 to 1996. Journal of Underwater Acoustics (USN) 52(3):48.

Clarke, C. W., and R. A. Charif. 1998. Acoustic monitoring of large whales to the west of Britain and Ireland using bottom mounted hydrophone arrays, October 1996-September 1997.

Clarke, J. T., and S. E. Moore. 2002. A note on observations of gray whales in the southern Chukchi and northern Bering Seas, August-November, 1980-89. Journal of Cetacean Research and Management 4(3):283-288.

Clarke, M. R. 1976. Observations on sperm whale diving. Journal of the Marine Biological Association of the United Kingdom 56(3):809-810.

Clarke, M. R. 1986. Cephalopods in the diet of odontocetes. Research on Dolphins. M. M. Bryden and R. J. Harrison (eds.). Oxford Univ. Press, Oxford, England. ISBN 0-19-857606-4. p.281-321.

Clarke, R. 1956. A giant squid swallowed by a sperm whale. Proceedings of the Zoological Society of London 126:645.

Clavero, M., and E. Garcıa-Berthou. 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology and Evolution 20(3):110.

Clemens, W. A., and G. V. Wilby. 1961. Fishes of the Pacific Coast of Canada, Second edition, volume 68. Fisheries Research Board of Canada.

Clement, D. 2013. Effects on Marine Mammals. Ministry for Primary Industries. Literature review of ecological effects of aquaculture. Report prepared by Cawthron Institute, Nelson, New Zealand.

Clyne, H., R. Leaper, and J. Kennedy. 1999. Computer simulation of interactions between the North Atlantic right whale (Eubalaena glacialis) and shipping. European Research on Cetaceans 13:458.

CMC, and NMFS. 1988. Proceedings of a workshop to review and evaluate whale watching programs and management needs. Center for Marine Conservation and National Marine Fisheries Service, Monterey, California.

Cogan, J. 2015. 2015 Whale Sightings in the Salish Sea: Central Salish Sea and Puget Sound. Center for Whale Research, Friday Harbor, WA.

Cole, T. V. N., D. L. Hartley, and R. L. Merrick. 2005. Mortality and serious injury determinations for large whales stocks along the eastern seaboard of the United States, 1999-2003. NOAA, NMFS, NEFSC.

Collard, S. B. 1990. The influence of oceanographic features on post-hatchling sea turtle distribution and dispersion in the pelagic environment. Pages 111-114 *in* T. H. Richardson, J. I. Richardson, and M. Donnelly, editors. Tenth Annual Workshop on Sea Turtle Biology and Conservation.

Collin, S. P., and D. L. Whitehead. 2004. The functional roles of passive electroreception in non-electric fishes. Animal Biology 54(1):25-Jan.

Cominelli, S., and coauthors. 2018. Noise exposure from commercial shipping for the southern resident killer whales population. Marine Pollution Bulletin 136:177-200.

Commerce. 2001. Joint Interim Report Bahamas Marine Mammal Stranding Event of 15–16 March 2000. Washington, DC: U.S. Department of Commerce, & U.S. Department of the Navy. .

Commission, I. W. 2004. Report of the Workshop on the Western Gray Whale: Research and Monitoring Needs, 22-25 October 2002, Ulsan, Korea. Journal of Cetacean Research and Management 6(Supplement):487-500.

Commission, W. S. C. 2003. Salmon and steelhead habitat limiting factors: Water resource inventory area 16 Dosewallips-Skokomish Basin.

Conn, P. B., and G. K. Silber. 2013. Vessel speed restrictions reduce risk of collision-related mortality for North Atlantic right whales. Ecosphere 4(4).

Connor, W. P., and coauthors. 2014. Research, monitoring, and evaluation of emerging issues and measures to recover the Snake River fall Chinook salmon ESU, 1/1/2012–12/31/2013: Annual report, 1991-029-00. Bonneville Power Administration.

Connor, W. P., J. G. Sneva, K. F. Tiffan, R. K. Steinhorst, and D. Ross. 2005. Two alternative juvenile life history types for fall Chinook salmon in the Snake River basin. Transactions of the American Fisheries Society 134(2):291-304.

Corey, F. C. G., R. H. Draeger, O. A. Brines, J. S. Shaver, and E. L. 1943. An experimental study of concussion. United States Naval Medical Bulletin 41(1):339-352.

Corkeron, P. J. 1995. Humpback whales (*Megaptera novaeangliae*) in Hervey Bay, Queensland: Behaviour and responses to whale-watching vessels. Canadian Journal of Zoology 73(7):1290-1299.

COSEWIC. 2002. COSEWIC assessment and update status report on the blue whale *Balaenoptera musculus* (Atlantic population, Pacific population) in Canada.1–32.

Costa, D., and coauthors. 2016a. A bioenergetics approach to understanding the population consequences of disturbance: elephant seals as a model system The Effects of Noise on Aquatic Life II (pp. 116–169). New York: Springer.

Costa, D. P., and coauthors. 2003. The effect of a low-frequency sound source (acoustic thermometry of the ocean climate) on the diving behavior of juvenile northern elephant seals, *Mirounga angustirostris*. The Journal of the Acoustical Society of America 113(2):1155–1165.

Costa, D. P., and coauthors. 2016b. Assessing the exposure of animals to acoustic disturbance: Towards an understanding of the population consequences of disturbance.

Costidis, A. M., and S. A. Rommel. 2016. The extracranial venous system in the heads of beaked whales, with implications on diving physiology and pathogenesis. Journal of Morphology 277(1):34-64.

Coulson, T., and coauthors. 2006. Estimating individual contributions to population growth: Evolutionary fitness in ecological time. Proceedings of the Royal Society Biological Sciences Series B 273:547-555.

Council, N. R. 2004. Endangered and threatened fishes in the Klamath River Basin: causes of decline and strategies for recovery. National Academies Press.

Courtney, M. B., M. D. Evans, J. F. Strøm, A. H. Rikardsen, and A. C. Seitz. 2019. Behavior and thermal environment of Chinook salmon Oncorhynchus tshawytscha in the North Pacific Ocean, elucidated from pop-up satellite archival tags. Environmental Biology of Fishes 102(8):1039-1055.

Cox, T. M., and coauthors. 2006. Understanding the impacts of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7(3):177-187.

Cranford, T. W., and P. Krysl. 2015. Fin whale sound reception mechanisms: Skull vibration enables low-frequency hearing. PLoS ONE 10(1):e116222.

Crawford, J. D., and X. Huang. 1999. Communication signals and sound production mechanisms of mormyrid electric fish. Journal of Experimental Biology 202:1417-1426.

Croll, D., and coauthors. 1999a. From wind to whales: Foraging ecology of rorquals in the California Current. Thirteen Biennial Conference on the Biology of Marine Mammals, 28 November - 3 December Wailea Maui HI. p.41.

Croll, D. A., A. Acevedo-Gutierrez, B. R. Tershy, and J. Urban-Ramirez. 2001a. The diving behavior of blue and fin whales: is dive duration shorter than expected based on oxygen stores? Comparative Biochemistry and Physiology a-Molecular & Integrative Physiology 129(4):797-809.

Croll, D. A., and coauthors. 2002. Only male fin whales sing loud songs. Nature 417:809.

Croll, D. A., C. W. Clark, J. Calambokidis, W. T. Ellison, and B. R. Tershy. 2001b. Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. Animal Conservation 4(1):13–27.

Croll, D. A., B. R. Tershy, A. Acevedo, and P. Levin. 1999b. Marine vertebrates and low frequency sound. Technical report for LFA EIS, 28 February 1999. Marine Mammal and Seabird Ecology Group, Institute of Marine Sciences, University of California Santa Cruz. 437p.

Crozier, L. G., and coauthors. 2008. Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1(2):252-270.

Crozier, L. G., and coauthors. 2019. Climate vulnerability assessment for Pacific salmon and steelhead in the California Current Large Marine Ecosystem. PLoS ONE 14(7).

Crum, L., and Y. Mao. 1996. Acoustically enhanced bubble growth at low frequencies and its implications for human diver and marine mammal safety. Acoustical Society of America 99(5):2898-2907.

Crum, L. A., and coauthors. 2005. Monitoring bubble growth in supersaturated blood and tissue ex vivo and the relevance to marine mammal bioeffects. Acoustics Research Letters Online 6(3):214-220.

Culik, B. M., S. Koschinski, N. Tregenza, and G. Ellis. 2001. Reactions of harbor porpoises Phocoena phocoena and herring Clupea harengus to acoustic alarms. Marine Ecology Progress Series 211:255-260.

Cummings, W. C., and P. O. Thompson. 1971a. Gray whales, Eschrichtius robustus, avoid the underwater sounds of killer whales, Orcinus orca. Fishery Bulletin 69(3):525-530.

Cummings, W. C., and P. O. Thompson. 1971b. Underwater sounds from the blue whale, *Balaenoptera musculus*. Journal of the Acoustical Society of America 50(4B):1193–1198.

Cummings, W. C., and P. O. Thompson. 1994. Characteristics and seasons of blue and finback whale sounds along the U.S. west coast as recorded at SOSUS stations. Journal of the Acoustical Society of America 95:2853.

Cunningham, K. A., B. L. Southall, and C. Reichmuth. 2014. Auditory sensitivity of seals and sea lions in complex listening scenarios. Journal of the Acoustical Society of America 136(6):3410-3421.

Cure, C., and coauthors. 2012. Pilot whales attracted to killer whale sounds: Acoustically-mediated interspecific interactions in cetaceans. PLoS ONE 7(12):e52201.

Cure, C., and coauthors. 2015. Predator sound playbacks reveal strong avoidance responses in a fight strategist baleen whale. Marine Ecology Progress Series 526:267-282.

D'Agnese, E., and coauthors. 2020. Reemergence of Guadalupe fur seals in the U.S. Pacific Northwest: The epidemiology of stranding events during 2005-2016. Marine Mammal Science:18pp.

D'Vincent, C. G., R. M. Nilson, and R. E. Hanna. 1985. Vocalization and coordinated feeding behavior of the humpback whale in southeastern Alaska. Scientific Reports of the Whales Research Institute 36:41-47.

Dagorn, L., K. N. Holland, V. Restrepo, and G. Moreno. 2013. Is it good or bad to fish with FADs? What are the real impacts of the use of drifting FADs on pelagic marine ecosystems? Fish and Fisheries 14(3):391-415.

Dahl, P. H., and coauthors. 2020. Physical effects of sound exposure from underwater explosions on Pacific sardines (Sardinops sagax). The Journal of the Acoustical Society of America 147(4):2383-2395.

Dahlheim, M. E., J. M. Waite, and P. A. White. 2009. Cetaceans of southeast Alaska: Distribution and seasonal occurrence. U.S. Department of Commerce, Lincoln, Nebraska.

Dähne, M., and coauthors. 2014. Marine mammals and windfarms: effects of alpha ventus on harbour porpoises. Pages 133–149 *in* Ecological Research at the Offshore Windfarm alpha ventus. Springer.

Dalla-Rosa, L., J. K. B. Ford, and A. W. Trites. 2012. Distribution and relative abundance of humpback whales in relation to environmental variables in coastal British Columbia and adjacent waters. Continental Shelf Research 36:89–104.

Daly, E. A., and coauthors. 2014. Juvenile Steelhead Distribution, Migration, Feeding, and Growth in the Columbia River Estuary, Plume, and Coastal Waters. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 6(1):62-80.

Danielsdottir, A. K., E. J. Duke, P. Joyce, and A. Arnason. 1991. Preliminary studies on genetic variation at enzyme loci in fin whales (Balaenoptera physalus) and sei whales (Balaenoptera borealis) form the North Atlantic. Report of the International Whaling Commission Special Issue 13:115-124.

Danil, K., and J. A. St. Leger. 2011. Seabird and dolphin mortality associated with underwater detonation exercises. Marine Technology Society Journal 45(6):89-95.

Danilewicz, D., M. Tavares, I. B. Moreno, P. H. Ott, and C. C. Trigo. 2009. Evidence of feeding by the humpback whale (Megaptera novaeangliae) in mid-latitude waters of the western South Atlantic. JMBA2 - Biodiversity Records-Published Online 3Pgs.

Darling, J. D., C. P. Nicklin, and M. E. Jones. 2006. Humpback whale songs: Do they organize males during the breeding season? Behaviour 143(9):1051-1101.

DART. 2013. http://www.cbr.washington.edu/dart/query/adult_annual_sum. D. A. R. Time), editor.

Daugherty, A. 2016. Rare Fin whale spotted in Puget Sound. K5 News, Seattle, WA.

Davenport, J., and G. H. Balazs. 1991. 'Fiery bodies' - Are pyrosomas an important component of the diet of leatherback turtles? British Herpetological Society Bulletin 37:33-38.

David, L. 2002. Disturbance to Mediterranean cetaceans caused by vessel traffic. Cetaceans of the Mediterranean and Black Seas: State of Knowledge and Conservation Strategies. G. Notarbartolo de Sciara (ed.). Section 11. 21pp. A report to the ACCOBAMS Secretariat, Monaco, February.

Davis, R. W., W. E. Evans, and B. Würsig. 2000. Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations. Volume II: Technical Report. Texas A&M, OCS Study MMS 2000-03, Galveston.

Davis, R. W., and coauthors. 2007. Diving behavior of sperm whales in relation to behavior of a major prey species, the jumbo squid, in the Gulf of California, Mexico. Marine Ecology Progress Series 333:291-302.

Davis, R. W., and coauthors. 2002. Cetacean habitat in the northern oceanic Gulf of Mexico. Deep-Sea Research Part I-Oceanographic Research Papers 49(1):121-142.

De Riu, N., J.-W. Lee, S. S. Huang, G. Moniello, and S. S. J. A. T. Hung. 2014. Effect of dietary selenomethionine on growth performance, tissue burden, and histopathology in green and white sturgeon. 148:65-73.

de Soto, N. A., and coauthors. 2020. Fear of killer whales drives extreme synchrony in deep diving beaked whales. Scientific Reports 10(1):1-9.

de Swart, R. L., P. S. Ross, J. G. Vos, and A. Osterhaus. 1996. Impaired immunity in habour seals (*Phoca vitulina*) exposed to bioaccumulated environmental contaminants: Review of long-term feeding study. Environmental Health Perspectives 104(Supplement 4):823-828.

Deakos, M. H., and M. F. Richlen. 2015. Vessel-Based Marine Mammal Survey on the Navy Range off Kauai in Support of Passive Acoustic Monitoring and Satellite-Tagging Efforts: 1–9 February 2014 (Prepared for Naval Facilities Engineering Command Pacific for Commander, U.S. Pacific Fleet under Contract No. N62470-10-D-3011, CTO KB22). Honolulu, HI: HDR Inc.

Dear, J. J. F., D. A. Carder, C. E. Schlundt, and R. L. 2010. Growth and recovery of temporary threshold shift at 3 kHz in bottlenose dolphins: Experimental data and mathematical models. Journal of the Acoustical Society of America 127(5):3256-3266.

Debich, A. J., and coauthors. 2013. Passive acoustic monitoring for marine mammals in the Gulf of Alaska temporary maritime activities area 2012-2013. University of California San Diego, Scripps Institution of Oceanography, Marine Physical Laboratory, La Jolla, California.

Debich, A. J., and coauthors. 2014. Passive Acoustic Monitoring for Marine Mammals in the Northwest Training Range Complex 2012–2013. Marine Physical Laboratory, Scripps Institution of Oceanography, University of California San Diego, La Jolla, CA.

Deecke, V. B., P. J. B. Slater, and J. K. B. Ford. 2002. Selective habituation shapes acoustic predator recognition in harbour seals. Nature 417(6912):171-173.

Deepwater Horizon Natural Resource Damage Assessment Trustees. 2016. Deepwater Horizon oil spill: Final Programmatic Damage Assessment and Restoration Plan and Final Programmatic Environmental Impact Statement. Chapter 4: Injury to Natural Resources.

DeLacy, A. C., C. R. Hitz, and R. L. Dryfoos. 1964. Maturation, gestation, and birth of rockfish (*Sebastodes*) from Washington and adjacent waters. Washington Department of Fisheries.

Demarchi, M. W., M. Holst, D. Robichaud, M. Waters, and A. O. MacGillivray. 2012. Responses of Steller sea lions (*Eumetopias jubatus*) to in-air blast noise from military explosions. Aquatic Mammals 38(3):279.

Deng, X. 2000. Artificial reproduction and early life stages of the green sturgeon (Acipenser medirostris).

Dennison, S., and coauthors. 2011. Bubbles in live-stranded dolphins. Proceedings of the Royal Society B. Biological Sciences, 10.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: a review. Marine Pollution Bulletin 44(9):842-852.

Deruiter, S., and coauthors. 2013a. Responses of Cuvier's beaked whales to controlled and incidental exposure to mid-frequency active (MFA) sonar sounds. Pages 50-51 *in* Twenty-Seventh Annual Conference of the European Cetacean Society, Setubal, Portugal.

Deruiter, S. L., and coauthors. 2013b. Delphinid whistle production and call matching during playback of simulated military sonar. Marine Mammal Science 29(2):E46-E59.

DeRuiter, S. L., and coauthors. 2017. A multivariate mixed hidden Markov model for blue whale behaviour and responses to sound exposure. The Annals of Applied Statistics 11(1):362-392.

Deruiter, S. L., and coauthors. 2013c. First direct measurements of behavioural responses by Cuvier's beaked whales to mid-frequency active sonar. Biology Letters 9(4):Article 20130223.

DFO. 2017. Identification of Habitat of Special Importance to Fin Whale (*Balaenoptera physalus*) in Canadian Pacific Waters. . DFO Canadian Science Advisory Secretariat Science Advisory Report 2017/039.

Di Lorio, L., and C. W. Clark. 2010. Exposure to seismic survey alters blue whale acoustic communication. Biology Letters 6:51–54.

Dietrich, K. S., V. R. Cornish, K. S. Rivera, and T. A. Conant. 2007. Best practices for the collection of longline data to facilitate research and analysis to reduce bycatch of protected species. NOAA Technical Memorandum NMFS-OPR-35. 101p. Report of a workshop held at the International Fisheries Observer Conference Sydney, Australia, November 8,.

Dohl, T. P. 1983. Return of the humpback whale (Megaptera novaeangliae) to central California. Fifth Biennial Conference on the Biology of Marine Mammals, 27 November-1 December New England Aquarium Boston MA. p.23-24.

Dolphin, W. F. 1987. Ventilation and dive patterns of humpback whales, Megaptera novaeangliae, on their Alaskan feeding grounds. Canadian Journal of Zoology 65(1):83-90.

DoN. 2015. Northwest Training and Testing Activities Final Environmental Impact Statement/Overseas Environmental Impact Statement. N. Naval Facilities Engineering Command, EV21.KK, editor, Silverdale, WA.

Doney, S. C., and coauthors. 2012. Climate change impacts on marine ecosystems. Marine Science 4.

Donnelly, R. F., and R. L. Burr. 1995. Relative abundance and distribution of Puget Sound trawl-caught demersal fishes. Pages 1,038 *in* Puget Sound Research.

Donovan, G. P. 1991. A review of IWC stock boundaries. Report of the International Whaling Commission (Special Issue 13).

Dorn, M. W. 2002. Advice on west coast rockfish harvest rates from Bayesian meta-analysis of stock-recruit relationships. North American Journal of Fisheries Management 22:280-300.

Douglas, A. B., and coauthors. 2008. Incidence of ship strikes of large whales in Washington State. Journal of the Marine Biological Association of the United Kingdom 88(6):1121-1132.

Doukakis, P., and coauthors. 2020. Postrelease survival of green sturgeon (Acipenser medirostris) encountered as bycatch in the trawl fishery that targets California halibut (Paralichthys californicus,), estimated by using pop-up satellite archival tags. Fishery Bulletin 118(1):63-83.

Doyle, L. R., and coauthors. 2008. Applicability of information theory to the quantification of responses to anthropogenic noise by southeast Alaskan humpback whales. Entropy 10(2):33-46.

Drake, J. S., and coauthors. 2010. Status review of five rockfish species in Puget Sound, Washington: bocaccio (Sebastes paucispinis), canary rockfish (S. pinniger), yelloweye rockfish (S. ruberrimus), greenstriped rockfish (S. elongatus), and redstripe rockfish (S. proriger). U.S. Dept. Commer.

Duffy, E. J., D. A. Beauchamp, R. M. Sweeting, R. J. Beamish, and J. S. Brennan. 2010. Ontogenetic diet shifts of juvenile Chinook salmon in nearshore and offshore habitats of Puget Sound. Transactions of the American Fisheries Society 139:803-823.

Dumbauld, B. R., D. L. Holden, and O. P. Langness. 2008. Do sturgeon limit burrowing shrimp populations in Pacific Northwest Estuaries? Environmental Bioology of Fishes 83:283-296.

Dunlop, R., and coauthors. 2013. Multivariate analysis of behavioural response experiments in humpback whales (Megaptera novaeangliae). Journal of Experimental Biology 216(5):759-770.

Dunlop, R. A. 2016. The effect of vessel noise on humpback whale, *Megaptera novaeangliae*, communication behaviour. Animal Behaviour 111:13–21.

Dunlop, R. A. 2019. The effects of vessel noise on the communication network of humpback whales. Royal Society Open Science 6:12.

Dunlop, R. A., D. Cato, and M. J. Noad. 2010. Your attention please: increasing ambient noise levels elictis a change in communication behaviour in humpback whales (Megoptera novaeangliae). Proceedings of the Royal Society B, 277, 2521–2529. .

Dunlop, R. A., D. H. Cato, and M. J. Noad. 2008. Non-song acoustic communication in migrating humpback whales (*Megaptera novaeangliae*). Marine Mammal Science 24(3):613-629.

Dunlop, R. A., D. H. Cato, and M. J. Noad. 2014. Evidence of a Lombard response in migrating humpback whales (Megaptera novaeangliae). Journal of the Acoustical Society of America 136(1):430-437.

Dunlop, R. A., and coauthors. 2015. The behavioural response of humpback whales (*Megaptera novaeangliae*) to a 20 cubic inch air gun. Aquatic Mammals 41(4):412.

Dunlop, R. A., and coauthors. 2017. Determining the behavioural dose-response relationship of marine mammals to air gun noise and source proximity. Journal of Experimental Biology 220(16):2878–2886.

Dutton, P. H. 2006. Geographic variation in foraging strategies of leatherback populations: A hedge against catastrophy? Pages 189 *in* Twenty-Sixth Annual Conference on Sea Turtle Conservation and Biology.

Dutton, P. H., G. H. Balazs, and A. E. Dizon. 1998. Genetic stock identification of sea turtles caught in the Hawaii-based pelagic longline fishery. Pages 45-46 *in* S. P. Epperly, and J. Braun, editors. Seventeenth Annual Sea Turtle Symposium.

Dutton, P. H., B. W. Bowen, D. W. Owens, A. Barragan, and S. K. Davis. 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology 248:397-409.

Dutton, P. H., and coauthors. 2007. Status and genetic structure of nesting populations of leatherback turtles (*Dermochelys coriacea*) in the western Pacific. Chelonian Conservation and Biology 6(1):47-53.

Dwyer, C. M. 2004. How has the risk of predation shaped the behavioural responses of sheep to fear and distress? Animal Welfare 13(3):269-281.

Ebbert, J. C., S. S. Embrey, R. W. Black, A. J. Tesoriero, and A. L. Haggland. 2000. Water Quality in the Puget Sound Basin, Washington and British Columbia, 1996–98, volume 1216. U.S. Geological Survey, Tacoma, Washington.

Echeverria, T. W. 1987. Thirty-four species of California rockfishes: maturity and seasonality of reproduction. Fisheries Bulletin 85(2):229-250.

Eckert, K. L. 1993. The biology and population status of marine turtles in the North Pacific Ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOM-TM-NM FS-S W FSC-186, Honolulu, HI.

Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication.

Eckert, S. A. 2002. Distribution of juvenile leatherback sea turtle *Dermochelys coriacea* sightings. Marine Ecology Progress Series 230:289-293.

Eckert, S. A., K. L. Eckert, and T. H. Richardson. 1989. Proceedings of the Ninth Annual Workshop on Sea Turtle Conservation and Biology. U.S. Department of Commerce, NMFS-SEFC-232.

Eckert, S. A., J. Gearhart, and P. Lewis. 2007. Experiment to evaluate the target catch and bycatch reduction effectiveness of surface and mid-water drift gillnets in Trinidad. National Marine Fisheries Service, NOAA-NMFS-PO DG133F06SE5011.

Eckert, S. A., H.-C. Liew, K. L. Eckert, and E.-H. Chan. 1996. Shallow water diving by leatherback turtles in the South China Sea. Chelonian Conservation and Biology 2(2):237-243.

Eckert, S. A., and L. Sarti. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. Marine Turtle Newsletter 78:2-7.

Edds-Walton, P. L. 1997. Acoustic communication signals of mysticete whales. Bioacoustics-the International Journal of Animal Sound and Its Recording 8:47-60.

Edds-Walton, P. L., and J. J. Finneran. 2006. Evaluation of Evidence for Altered Behavior and Auditory Deficits in Fishes Due to Human-Generated Noise Sources. Pages 50 *in* U.S. Department of the Navy, editor. SPAWAR Systems Center, San Diego, CA.

Edds, P. L. 1988. Characteristics of finback *Balaenoptera physalus* vocalizations in the St. Lawrence estuary. Bioacoustics 1:131-149.

Edds, P. L., and J. A. F. Macfarlane. 1987. Occurrence and general behavior of balaenopterid cetaceans summering in the St. Lawrence Estuary, Canada. Canadian Journal of Zoology 65(6):1363-1376.

Edgell, T. C., and M. W. Demarchi. 2012. California and Steller sea lion use of major winter haulout in the Salish Sea over 45 years. Marine Ecology Progress Series 467:253-262.

Edwards, M. H., and coauthors. 2016. The Hawaii Undersea Military Munitions Assessment. Deep-Sea Research II, 128, 4–13.

Efroymson, R. A., G. W. Suter Ii, W. H. Rose, and S. Nemeth. 2001. Ecological risk assessment framework for low-altitude aircraft overflights: I planning the analysis and estimating exposure. Risk Analysis 21:251-262.

Elfes, C. T., and coauthors. 2010. Geographic variation of persistent organic pollutant levels in humpback whale (Megaptera novaeangliae) feeding areas of the North Pacific and North Atlantic. Environmental Toxicology and Chemistry 29(4):824-834.

Eller, A. I., and R. C. Cavanagh. 2000. Subsonic Aircraft Noise at and Beneath the Ocean Surface: Estimation of Risk for Effects on Marine Mammals. United States Air Force Research Laboratory, McLean, VA.

Elliott, J. E., L. K. Wilson, and B. Wakeford. 2005. Polybrominated diphenyl ether trends in eggs of marine and freshwater birds from British Columbia, Canada, 1979-2002. Environmental Science & Technology 39(15):5584-5591.

Ellison, W. T., B. L. Southall, C. W. Clark, and A. S. Frankel. 2011. A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. Conservation Biology.

Ellison, W. T., B. L. Southall, C. W. Clark, and A. S. Frankel. 2012. A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. Conservation Biology 26(1):21-28.

Emmons, C. K., B. M. Hanson, and M. O. Lammer. 2017. Seasonal occurrence of cetaceans along the Washington Coast from passive acoustic monitoring. The Journal of the Acoustical Society of America.

Emmons, C. K., M. B. Hanson, and M. O. Lammers. 2019. Monitoring the occurrence of Southern resident killer whales, other marine mammals, and anthropogenic sound in the Pacific Northwest. Prepared for: U.S. Navy, U.S. Pacific Fleet, Pearl Harbor, HI. National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center under MIPR N00070-17-MP-4C419.

Engas, A., E. Haugland, and J. Ovredal. 1998. Reactions of Cod (Gadus Morhua L.) in the Pre-Vessel Zone to an Approaching Trawler under Different Light Conditions. Hydrobiologia, 371/372: 199–206. .

Engas, A., O. Misund, A. Soldal, B. Horvei, and A. Solstad. 1995. Reactions of Penned Herring and Cod to Playback of Original, Frequency-Filtered and Time-Smoothed Vessel Sound. Fisheries Research, 22: 243–54. .

Enger, P. S. 1981. Frequency discrimination in teleosts-central or peripheral?, New York, Spring Verlag.

Environmental Sciences Group. 2005. Canadian Forces Maritime Experimental and Test Ranges (CFMETR) Environmental Assessment Update 2005. Environmental Sciences Group, Royal Military College, RMC-CCE-ES-05-21, Kingston, Ontario.

Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Marine Mammal Science 18(2):394-418.

Erbe, C., A. MacGillivray, and R. Williams. 2012. Mapping cumulative noise from shipping to inform marine spatial planning. Journal of the Acoustical Society of America 132(5):EL423-EL428.

Erbe, C., and coauthors. 2019. The effects of ship noise on marine mammals- A review. Frontiers in Marine Science 6.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. 2015. Communication masking in marine mammals: a review and research strategy. Marine Pollution Bulletin, 1–24.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. 2016. Communication masking in marine mammals: A review and research strategy. Marine Pollution Bulletin 103(1–2):15–38.

Erbe, C., R. Williams, D. Sandilands, and E. Ashe. 2014. Identifying modeled ship noise hotspots for marine mammals of Canada's Pacific region. PLoS ONE 9(3):e89820.

Erickson, D. L., and M. A. Webb. 2007. Spawning periodicity, spawning migration, and size at maturity of green sturgeon, Acipenser medirostris, in the Rogue River, Oregon. Environmental Biology of Fishes 79(3-4):255-268.

Eschmeyer, W. N., E. S. Herald, and H. Hammann. 1983. A Field Guide to Pacific Coast Fishes of North America. Houghton Mifflin Company, Boston, Massachusetts.

Esperon-Rodriguez, M., and J. P. Gallo-Reynoso. 2012. The re-colonization of the Archipelago of San Benito, Baja California, by the Guadalupe fur seal. Revista Mexicana de Biodiversidad 83(1):170-176.

Esperon-Rodriguez, M., and J. P. Gallo-Reynoso. 2013. Juvenile and subadult feeding preferences of the Guadalupe fur seal (*Arctocephalus townsendi*) at San Benito Archipelago, Mexico. Aquatic Mammals 39(2):125-131.

Etnier, M. A. 2002. Occurrences of Guadalupe fur seals (*Arctocephalus townsend*) on the Washington coast over the past 500 years. Marine Mammal Science 18(2):551-557.

Evans, K., M. Hindell, and G. Hince. 2004. Concentrations of organochlorines in sperm whales (*Physeter macrocephalus*) from Southern Australian waters. Marine Pollution Bulletin 48:486-503.

Evans, P. G. H., and A. Bjørge. 2013. Impacts of climate change on marine mammals. Marine Climate Change Impacts Parternship: Science Review:134-148.

Evans, P. G. H., P. J. Canwell, and E. Lewis. 1992. An experimental study of the effects of pleasure craft noise upon bottle-nosed dolphins in Cardigan Bay, West Wales. European Research on Cetaceans 6:43-46. Proceedings of the Sixth Annual Conference of the European Cetacean Society, San Remo, Italy, 20-22 February.

Evans, P. G. H., and coauthors. 1994. A study of the reactions of harbour porpoises to various boats in the coastal waters of southeast Shetland. European Research on Cetaceans 8:60-64.

Fahlman, A., S. K. Hooker, A. Szowka, B. L. Bostrom, and D. R. Jones. 2009. Estimating the effect of lung collapse and pulmonary shunt on gas exchange during breath-hold diving: The Scholander and Kooyman legacy. Respiratory Physiology and Neurobiology 165(1):28-39.

Fahlman, A., and coauthors. 2014a. Inflation and deflation pressure-volume loops in anesthetized pinnipeds confirms compliant chest and lungs. Frontiers in Physiology 5.

Fahlman, A., A. Olszowka, B. Bostrom, and D. R. Jones. 2006. Deep diving mammals: Dive behavior and circulatory adjustments contribute to bends avoidance. Respiratory Physiology and Neurobiology 153(1):66-77.

Fahlman, A., P. L. Tyack, P. J. Miller, and P. H. Kvadsheim. 2014b. How man-made interference might cause gas bubble emboli in deep diving whales. Frontiers in Physiology 5:13.

Fair, P. A., and coauthors. 2014. Stress response of wild bottlenose dolphins (Tursiops truncatus) during capture-release health assessment studies. General and Comparative Endocrinology 206:203-212.

Falcone, E. A., and G. S. Schorr. 2014. Distribution and demographics of marine mammals in SOCAL through photo-identification, genetics, and satellite telemetry: A summary of surveys conducted 1 July 2012 – 30 June 2013. Cascadia Research Collective, Olympia, Washington.

Falcone, E. A., and coauthors. 2009. Sighting characteristics and photo-identification of Cuvier's beaked whales (Ziphius cavirostris) near San Clemente Island, California: A key area for beaked whales and the military? Marine Biology 156(12):2631-2640.

Falcone, E. A., and coauthors. 2017. Diving behaviour of Cuvier's beaked whales exposed to two types of military sonar. Royal Society Open Science 4(170629):1–21.

Falke, K. J., and coauthors. 1985. Seal lungs collapse during free diving: Evidence from arterial nitrogen tensions. Science 229(4713):556-558.

Farak, A. M., M. W. Richie, J. A. Rivers, and R. K. Uyeyama. 2011. Cruise Report, Marine Species Monitoring and Lookout Effectiveness Study Koa Kai, November 2010, Hawaii Range Complex. Prepared for Commander, U.S. Pacific Fleet.

Farmer, N. A., D. P. Noren, E. M. Fougères, A. Machernis, and K. Baker. 2018. Resilience of the endangered sperm whale *Physeter macrocephalus* to foraging disturbance in the Gulf of Mexico, USA: A bioenergetic approach. Marine Ecology Progress Series 589:241–261.

Farr, R. A., and J. C. Kern. 2004. Green sturgeon population characteristics in Oregon. Oregon Department of Fish and Wildlife, Clackamas, Oregon.

Farrell, R. E., and S. D. Siciliano. 2004. Environmental Effects of Radio Frequency (RF) Chaff Released during Military Training Exercises: A Review of the Literature.

Fearnbach, H., J. W. Durhan, S. A. Mizroch, S. Barbeaux, and P. R. Wade. 2012. Winter observations of a group of female and immature sperm whales in the high-latitude waters near the Aleutian Islands, Alaska. Marine Biodiversity Records 5: e13.

Felix, F. 2001. Observed changes of behavior in humphack whales during whalewatching encounters off Ecuador. Pages 69 *in* 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Félix, F. 2001. Observed changes of behavior in humpback whales during whalewatching encounters off Ecuador. 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Felix, F., and K. V. Waerebeek. 2005. Whale mortality from ship strikes in Ecuador and West Africa. Latin American Journal of Aquatic Mammals 4(1):55-60.

Felleman, F., J. R. Heimlich-Boran, and R. W. Osborne. 1991. The feeding ecology of killer whales (*Orcinus orca*) in the Pacific Northwest. Pages 113-148 *in* K. Pryor, and K. Norris, editors. Dolphin Societies, discoveries, and puzzles. University of California Press, Berkeley, California.

Ferguson, M. C., J. Barlow, P. Feidler, S. B. Reilly, and T. Gerrodette. 2006. Spatial models of delphinid (family Delphinidae) encouter rate and group size in the eastern Pacific Ocean. Ecological Modelling 193:645–662.

Ferguson, M. C., C. Curtice, and J. Harrison. 2015. Biologically important areas for cetaceans within U.S. waters – Gulf of Alaska region. Aquatic Mammals (Special Issue) 41(1):65–78.

Fernandez, A., and coauthors. 2005a. Gas and fat embolic syndrome involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Veterinary Pathology 42(4):446-457.

Fernandez, A., and coauthors. 2005b. New gas and fat embolic pathology in beaked whales stranded in the Canary Islands. Pages 90 *in* Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Ferrara, G. A., T. Mongillo, and L. M. Barré. 2017. Reducing disturbance from vessels to southern resident killer whales : assessing the effectiveness of the 2011 federal regulations in advancing recovery goals.

Fiedler, P. C., and coauthors. 1998. Blue whale habitat and prey in the California Channel Islands. Deep-Sea Research Part Ii-Topical Studies in Oceanography 45(8-9):1781-1801.

Figureroa-Carranza, A. L. 1994. Early lactation and attendance behavior of the Guadalupe fur seal females (*Arctocephalus townsendi*). University of California, Santa Cruz, California.

Filadelfo, R., and coauthors. 2009a. Correlating military sonar use with beaked whale mass strandings: What do the historical data show? Aquatic Mammals 35(4):435-444.

Filadelfo, R., and coauthors. 2009b. Correlating whale strandings with Navy exercises off southern California. Aquatic Mammals 35(4):445-451.

Findlay, K. P., and P. B. Best. 1995. Summer incidence of humpback whales on the west coast of South Africa. (Megaptera novaeangliae). South African Journal of Marine Science 15:279-282.

Finneran, J., and B. Branstetter. 2013. Effects of Noise on Sound Perception in Marine Mammals Animal Communication and Noise (Vol. 2, pp. 273–308). Springer Berlin Heidelb.

Finneran, J., D. Carder, C. Schlundt, and R. Dear. 2010. Temporary threshold shift in a bottlenose dolphin (Tursiops truncatus) exposed to intermittent tones. The Journal of Acoustical Society of America, 127(5), 3267–3272. .

Finneran, J. J. 2015a. Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. Journal of the Acoustical Society of America 138(3):1702-1726.

Finneran, J. J. 2015b. Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. The Journal of the Acoustical Society of America 138(3):1702-1726.

Finneran, J. J. 2018. Conditioned attenuation of auditory brainstem responses in dolphins warned of an intense noise exposure: Temporal and spectral patterns. J Acoust Soc Am 143(2):795.

Finneran, J. J., D. A. Carder, and S. H. Ridgway. 2001. Temporary threshold shift (TTS) in bottlenose dolphins (Tursiops truncatus) exposed to tonal signals. Journal of the Acoustical Society of America 110(5 Part 2):2749.

Finneran, J. J., D. A. Carder, C. E. Schlundt, and S. H. Ridgway. 2005. Temporary threshold shift in bottlenose dolphins (Tursiops truncatus) exposed to mid-frequency tones. Journal of the Acoustical Society of America 118(4):2696-2705.

Finneran, J. J., R. Dear, D. A. Carder, and S. H. Ridgway. 2003. Auditory and behavioral responses of California sea lions (*Zalophus californianus*) to single underwater impulses from an arc-gap transducer. The Journal of the Acoustical Society of America 114(3):1667–1677.

Finneran, J. J., and C. E. Schlundt. 2004. Effects of intense pure tones on the behavior of trained odontocetes. SPAWAR Systems Center, San Diego.

Finneran, J. J., and C. E. Schlundt. 2010. Frequency-dependent and longitudinal changes in noise-induced hearing loss in a bottlenose dolphin (Tursiops truncatus). Journal of the Acoustical Society of America 128(2):567-570.

Finneran, J. J., and C. E. Schlundt. 2013. Effects of fatiguing tone frequency on temporary threshold shift in bottlenose dolphins (Tursiops truncatus). Journal of the Acoustical Society of America 133(3):1819-1826.

Finneran, J. J., C. E. Schlundt, B. Branstetter, and R. L. Dear. 2007. Assessing temporary threshold shift in a bottlenose dolphin (Tursiops truncatus) using multiple simultaneous auditory evoked potentials. Journal of the Acoustical Society of America 122(2):1249-1264.

Finneran, J. J., and coauthors. 2015. Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. Journal of the Acoustical Society of America 137(4):1634-1646.

Finneran, J. J., C. E. Schlundt, R. Dear, D. A. Carder, and S. H. Ridgway. 2002. Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic watergun. Journal of the Acoustical Society of America 111(6):2929-2940.

Fiori, L., E. Martinez, M. B. Orams, and B. Bollard. 2019. Effects of whale-based tourism in Vava'u, Kingdom of Tonga: Behavioural responses of humpback whales to vessel and swimming tourism activities. PLoS ONE 14(7).

Fish, U., and W. Service. 1995. Working paper on restoration needs: habitat restoration actions to double natural production of anadromous fish in the Central Valley of California. US Fish and Wildlife Service.

Fisher, F. W. 1994. Past and present status of Central Valley chinook salmon. Conservation Biology:870-873.

Fisher, J. P., and W. G. Pearcy. 1995. Distribution, migration, and growth of juvenile Chinook salmon, Oncorhynchus tshawytscha, off Oregon and Washington. Fisheries Bulletin 93:274-289.

Fisher, J. P., and coauthors. 2014. Early ocean dispersal patterns of Columbia River Chinook and coho salmon. Transactions of the American Fisheries Society 143(1):252-272.
Fisheries and Oceans Canada. 2019. Necropsy results: Southern Resident Killer Whale J34.
Fitch, J. E., and P. H. Young. 1948. Use and Effect of Explosives in California Coastal Waters. California Division Fish and Game, Sacramento, CA.
Fleischer, L. A. 1978. The distribution, abundance, and population characteristics of the Guadalupe fur seal, *Arctocephalus townsendi* (Merriam 1897). University of Washington, Seattle, Washington.
Fleming, A., and J. Jackson. 2011. Global review of humpback whales (Megaptera novaeangliae).
Flinn, R. D., A. W. Trites, E. J. Gregr, and R. I. Perry. 2002. Diets of fin, sei, and sperm whales in British Columbia: an analysis of commercial whaling records, 1963–1967. Marine Mammal Science 18(3):663-679.
Flower, J. E., and coauthors. 2015. Baseline plasma corticosterone, haematological and biochemical results in nesting and rehabilitating loggerhead sea turtles (*Caretta caretta*). Conservation Physiology 3(1).
Fonnesbeck, C. J., L. P. Garrison, L. I. Ward-Geiger, and R. D. Baumstark. 2008. Bayesian hierarchichal model for evaluating the risk of vessel strikes on North Atlantic right whales in the SE United States. Endangered Species Research 6(1):87-94.
Foote, A. D., R. W. Osborne, and A. R. Hoelzel. 2004. Whale-call response to masking boat noise. Nature 428:910.
Foott, J. S., R. Harmon, and R. Stone. 2003. Ceratomyxosis resistance in juvenile Chinook salmon and steelhead trout from the Klamath River, 2002 Investigational Report. U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [768 Kb] at KRIS Web site.
Force, U. S. D. o. t. A. 1997. Environmental effects of self-protection chaff and flares. Langley Air Force Base, VA: U.S. Department of the Air Force.
Ford, J., and G. Ellis. 2006. Selective foraging by fish-eating killer whales Orcinus orca in British Columbia. Marine Ecology-progress Series - MAR ECOL-PROGR SER 316:185-199.
Ford, J. K., and coauthors. 1998. Dietary specialization in two sympatric populations of killer whales (Orcinus orca) in coastal British Columbia and adjacent waters. Canadian Journal of Zoology 76(8):1456-1471.
Ford, J. K. B. 1989. Acoustic behaviour of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Canadian Journal of Zoology 67:727-745.
Ford, J. K. B. 1991. Vocal traditions among resident killer whales (*Orcinus orca*) in coastal waters of British Columbia. Canadian Journal of Zoology 69(6):1454-1483.
Ford, J. K. B., A.L. Rambeau, R.M. Abernathy, M.D. Boogaards, L.M. Nichol, and L.D. Spaven 2009. As Assessment of the Potential for Recovery of Humpback Whales off the Pacific Coast of Canada. Pages 33 *in*. DFO Canadian Science Advisory Secretariat Research Document 2009/015.
Ford, J. K. B., and coauthors. 2010a. Distribution and Relative Abundance of Cetaceans in Western Canadian Waters From Ship Surveys, 2002-2008. Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, Canada.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer whales: The natural history and genealogy of *Orcinus orca* in British Columbia and Washington State, 2nd Edition. UBC Press, Vancouver, British Columbia.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 1994. Killer whales: The natural history and genealogy of Orcinus orca in British Columbia and Washington State. UBC Press, University of British Columbia, Vancouver, Canada. 102pp. ISBN 0-7748-0469-6.

Ford, J. K. B., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2009. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biology Letters 6(1):139–142.

Ford, J. K. B., and R. R. Reeves. 2008. Fight or flight: Antipredator strategies of baleen whales. Mammal Review 38(1):50-86.

Ford, J. K. B., B. M. Wright, G. M. Ellis, and J. R. Candy. 2010b. Chinook salmon predation by resident killer whales: seasonal and regional selectivity, stock identity of prey, and consumption rates. Canadian Science Advisory Secretariat= Secrétariat canadien de consultation ….

Ford, M. J., (editor). 2011a. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. Pages 281 p *in* U.S. Department of Commerce, editor.

Ford, M. J., and coauthors. 2011. Inferred Paternity and Male Reproductive Success in a Killer Whale (*Orcinus orca*) Population. Journal of Heredity 102(5):537-553.

Ford, M. J., and coauthors. 2016. Estimation of a killer whale (Orcinus orca) population's diet using sequencing analysis of DNA from feces. PLoS ONE 11(1).

Ford, M. J., and coauthors. 2018. Inbreeding in an endangered killer whale population. Animal Conservation 21:423-432.

Ford, M. J. e. 2011b. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest, volume NMFS-NWFSC-113. U.S. Dept. Commer., NOAA Tech. Memo.

Forney, K. A. 2007. Preliminary estimates of cetacean abundance along the U.S. west coast and within four national marine sanctuarnes during 2005. NOAA Technical Memorandum NMFS-SWFSC-406. 33p.

Forney, K. A., and J. Barlow. 1998. Seasonal patterns in the abundance and distribution of California cetaceans, 1991–1992. Marine Mammal Science 14(3):460–489.

Forney, K. A., and R. L. Brownell Jr. 1996. Preliminary report of the 1994 Aleutian Island marine mammal survey. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Paper SC/48/011, La Jolla, California.

Forney, K. A., and coauthors. 2012. Habitat-based spatial models of cetacean density in the eastern Pacific Ocean. Endangered Species Research 16(2):113-133.

Forney, K. A., and coauthors. 2017. Nowhere to go: noise impact assessments for marine mammal populations with high site fidelity. Endangered Species Research 32:391-413.

Fox, J. W. 2007. Testing the mechanisms by which source-sink dynamics alter competitive outcomes in a model system. The American Naturalist 170(3):396-408.

Frankel, A. S., and C. W. Clark. 2000. Behavioral responses of humpback whales (Megaptera novaeangliae) to full-scale ATOC signals. Journal of the Acoustical Society of America 108(4):1930-1937.

Franks, D. 2020. Going Viral on the Social Network: Disease spread in killer whale social networks. The Third International TRANSIT Workshop on Cross-disciplinary Research: Dynamics on and of Networks, Ron Cooke Hub, University of York, UK.

Frantzis, A., and P. Alexiadou. 2008. Male sperm whale (Physeter macrocephalus) coda production and coda-type usage depend on the presence of conspecifics and the behavioural context. Canadian Journal of Zoology 86(1):62-75.

Fraser, G. S. 2014. Impacts of offshore oil and gas development on marine wildlife resources. Pages 191-217 *in* J. E. Gates, D. L. Trauger, and B. Czech, editors. Peak Oil, Economic Growth, and Wildlife Conservation. Springer Publishers, New York.

Fraser, M. D., L. H. McWhinnie, R. R. Canessa, and C. T. Darimont. 2020. Compliance of small vessels to minimum distance regulations for humpback and killer whales in the Salish Sea. Marine Policy:104171.

Frazer, L. N., and E. Mercado Iii. 2000. A sonar model for humpback whale song. IEEE Journal of Oceanic Engineering 25(1):160-182.

French, R., H. Bilton, M. Osako, and A. Hartt. 1976a. Distribution and origin of sockeye salmon (*Oncorhynchus nerka*) in offshore waters of the North Pacific Ocean. International North Pacific Fisheries Commission.

French, R., H. Bilton, M. Osako, and A. C. Hartt. 1976b. Distribution and origin of sockeye salmon (*Oncorhynchus nerka*) in offshore waters of the North Pacific Ocean. International North Pacific Fisheries Commission, Vancouver, Canada.

Fresh, K. L., E. Casillas, L. Johnson, and D. L. Bottom. 2005. Role of the estuary in the recovery of Columbia River basin salmon and steelhead: an evaluation of limiting factors. NOAA Technical Memorandum NMFS-NWFSC 69:105.

Frey, A., P. H. Dutton, and G. H. Balazs. 2013. Insights on the demography of cryptic nesting by green turtles (Chelonia mydas) in the main Hawaiian Islands from genetic relatedness analysis. Journal of Experimental Marine Biology and Ecology 442:80-87.

Frid, A. 2003. Dall's sheep responses to overflights by helicopter and fixed-wing aircraft. Biological Conservation 110(3):387-399.

Frid, A., and L. Dill. 2002. Human-caused disturbance stimuli as a form of predation risk. Conservation Ecology 6(1):11.

Friedlaender, A. S., and coauthors. 2016a. Prey-mediated behavioral responses of feeding blue whales in controlled sound exposure experiments. Ecological Applications 26(4):1075-1085.

Friedlaender, A. S., and coauthors. 2016b. Prey-mediated behavioral responses of feeding blue whales in controlled sound exposure experiments. Ecological Applications.

Frinodig, A., and coauthors. 2009. Review of the California trawl fishery for Pacific ocean shrimp, *Pandalus jordani*, from 1992 to 2007. Marine Fisheries Review 71(2):1-13.

Fristrup, K. M., L. T. Hatch, and C. W. Clark. 2003. Variation in humpback whale (Megaptera novaeangliae) song length in relation to low-frequency sound broadcasts. Journal of the Acoustical Society of America 113(6):3411-3424.

Fritz, L., K. Sweeney, R. Towell, and T. Gelatt. 2016. Aerial and Ship-Based Surveys of Steller Sea Lions (*Eumetopias jubatus*) Conducted in Alaska in June-July 2013 through 2015, and an Update on the Status and Trend of the Western Distinct Population Segment in Alaska. A. F. S. Center, editor, Seattle, Washington.

Fromentin, J.-M., and B. Planque. 1996. *Calanus* and environment in the eastern North Atlantic. II. Influence of the North Atlantic Oscillation on *C. finmarchicus* and *C. helgolandicus*. Marine Ecology Progress Series 134:111-118.

Fromm, D. M. 2009. Reconstruction of acoustic exposure on orcas in Haro Strait.

Fujihara, J., T. Kunito, R. Kubota, and S. Tanabe. 2003. Arsenic accumulation in livers of pinnipeds, seabirds and sea turtles: Subcellular distribution and interaction between arsenobetaine and glycine betaine. Comparative Biochemistry and Physiology C Toxicology and Pharmacology 136(4):287-296.

Furin, C. G., F. von Hippel, B. Hagedorn, and T. O'Hara. 2013. Perchlorate trophic transfer increases tissue concentrations above ambient water exposure alone in a predatory fish. Journal of Toxicology and Environmental Health. Part A, 76(18), 1072–1084. .

Futuymda, D. J. 1986. Evolutionary biology, Second ed. edition. Sinauer

Associates, Inc., Sunderland, Massachussetts.

Gabriele, C. M., and A. S. Frankel. 2002. Surprising humpback whale songs in Glacier Bay National Park. Alaska Park Science: Connections to Natural and Cultural Resource Studies in Alaska's National Parks. p.17-21.

Gailey, G., and coauthors. 2016. Behavioural responses of western gray whales to a 4-D seismic survey off northeastern Sakhalin Island, Russia. Endangered Species Research 30:53–71.

Gailey, G., B. Wursig, and T. L. McDonald. 2007. Abundance, behavior, and movement patterns of western gray whales in relation to a 3-D seismic survey, Northeast Sakhalin Island, Russia. Environmental Monitoring and Assessment 134:75–91.

Gallagher, S. P., D. W. Wright, B. W. Collins, and P. B. Adams. 2010. A regional approach for monitoring salmonid status and trends: results from a pilot study in coastal Mendocino County, California. North American Journal of Fisheries Management 30(5):1075-1085.

Galli, L., and coauthors. 2003. Boat Source-Level Noise in Haro Strait: Relevance to Orca Whales. Orca Vocalization and Localization. Colorado College.

Gallo-Reynoso, J. P. 1994. Factors affecting the population status of Guadalupe fur seals, *Arctocephalus townsendi* (Merriam 1897), at Isla de Guadalupe, Baja California, Mexico. University of California, Santa Cruz.

Gallo-Reynoso, J. P., B. J. L. Boeuf, and A. L. Figueroa. 1995. Track, location, duration and diving behavior during foraging trips of Guadalupe fur seal females. Pages 41 *in* Eleventh Biennial Conference on the Biology of Marine Mammals, Orlando, Florida.

Gambaiani, D. D., P. Mayol, S. J. Isaac, and M. P. Simmonds. 2009. Potential impacts of climate change and greenhouse gas emissions on Mediterranean marine ecosystems and cetaceans. Journal of the Marine Biological Association of the United Kingdom 89(1):179-201.

Gambell, R. 1985a. Fin Whale *Balaenoptera physalus* (Linnaeus, 1758). Pages 171-192 *in* Handbook of Marine Mammals. Vol. 3: The Sirenians and Baleen Whales. Academic Press, London, U.K.

Gambell, R. 1985b. Sei whale, *Balaenoptera borealis* Lesson, 1828. Pages 155-170 *in* S. H. Ridway, and S. R. Harrison, editors. Handbook of Marine Mammals, volume 3: The Sirenians and Baleen Whales. Academic Press, London.

García-Aguilar, M. C., F. R. Elorriaga-Verplancken, H. Rosales-Nanduca, and Y. Schramm. 2018. Population status of the Guadalupe fur seal (Arctocephalus townsendi). Journal of Mammalogy 99(6):1522-1528.

García-Capitanachi, B., Y. Schramm, and G. Heckel. 2017. Population Fluctuations of Guadalupe Fur Seals (Arctocephalus philippii townsendi) Between the San Benito Islands and Guadalupe Island, Mexico, During 2009 and 2010. Aquatic Mammals 43(5).

García-Fernández, A. J., and coauthors. 2009. Heavy metals in tissues from loggerhead turtles (*Caretta caretta*) from the southwestern Mediterranean (Spain). Ecotoxicology and Environmental Safety 72(2):557-563.

Garcia Parraga, D., M. Moore, and A. Fahlman. 2018. Pulmonary ventilation–perfusion mismatch: a novel hypothesis for how diving vertebrates may avoid the bends.¿Proceedings of the Royal Society B: Biological Sciences,¿285(1877), 20180482. http://doi.org/10.1098/rspb.2018.0482.

Gardner, S. C., and S. Chavez-Rosales. 2000. Changes in the relative abundance and distribution of gray whales (*Eschrichtius robustus*) in Magdalena Bay, Mexico during an El Nino event. Marine Mammal Science 16(4):728-738.

Gaskin, D. E. 1973. Sperm whales in the western South Pacific. (Physeter catodon). New Zealand Journal of Marine and Freshwater Research 7-Jan(2-Jan):1-20.

Gaspar, P., and M. Lalire. 2017. A model for simulating the active dispersal of juvenile sea turtles with a case study on western Pacific leatherback turtles. PLoS ONE 12(7):e0181595.

Gaspin, J. B. 1975a. Experimental Investigations of the Effects of Underwater Explosions on Swimbladder Fish, I: 1973 Chesapeake Bay Tests. Naval Surface Weapons Center, White Oak Laboratory, Silver Spring, MD.

Gaspin, J. B. 1975b. Experimental Investigations of the Effects of Underwater Explosions on Swimbladder Fish, I: 1973 Chesapeake Bay Tests. Naval Surface Weapons Center, White Oak Laboratory, Silver Spring, MD.

Gaspin, J. B., G. B. Peters, and M. L. Wisely. 1976. Experimental investigations of the effects of underwater explosions on swimbladder fish. Naval Ordnance Lab, Silver Spring, MD.

Gauthier, J. M., C. D. Metcalfe, and R. Sears. 1997. Chlorinated organic contaminants in blubber biopsies from Northwestern Atlantic Balaenopterid whales summering in the Gulf of St Lawrence. Marine Environmental Research 44(2):201-223.

Gaydos, J. K., K. C. Balcomb III, R. W. Osborne, and L. Dierauf. 2004. Evaluating potential infectious disease threats for southern killer whales, *Orcinus orca*: a model for endangered species. Biological Conservation 117:253-262.

Gende, S. M., and coauthors. 2011. A Bayesian approach for understanding the role of ship speed in whale-ship encounters. Ecological Applications 21(6):2232-2240.

General. 1991. Conventional Warfare Ballistic, Blast, and Burn Injuries. Office of Surgeon General. In R. Zajitchuk, Col. (Ed.), U.S.A. Textbook of Military Medicine. Washington, DC: Office of the Surgeon General.

Geraci, J. R. 1990. Physiologic and toxic effects on cetaceans. Pages 167-197 *in* J. R. Geraci, and D. J. S. Aubin, editors. Sea Mammals and Oil: Confronting the Risks. Academic Press, San Diego.

Geraci, J. R., and coauthors. 1989. Humpback whales (Megaptera novaeangliae) fatally poisoned by dinoflagellate toxin. Canadian Journal of Fisheries and Aquatic Sciences 46(11):1895-1898.

Geraci, J. R., and D. J. St. Aubin. 1990. Sea mammals and oil: confronting the risks. Academic Press, New York.

Gero, S., D. Engelhaupt, L. Rendell, and H. Whitehead. 2009. Who Cares? Between-group variation in alloparental caregiving in sperm whales. Behavioral Ecology 20(4):838-843.

Giles, D. A., and R. Cendak. 2010. An Assessment of Vessel Effects on the Cohesion State of Southern Resident Killer Whale groups, and Measuring Vessel Compliance with Boating Guidelines.

Gill, J. A., K. Norris, and W. J. Sutherland. 2001. Why behavioural responses may not reflect the population consequences of human disturbance. Biological Conservation 97:265-268.

Gilman, E. 2008. Pacific leatherback conservation and research activities, financing and priorities. The World Conservation Union and Western Pacific Fishery Management Council and IUCN, Honolulu, Hawaii.

Gilpatrick, J. W., Jr., W. L. Perryman, J. R. L. Brownell, M. S. Lynn, and M. L. Deangelis. 1997. Geographical variation in North Pacific and Southern Hemisphere blue whales (Balaenoptera musculus). Unpublished paper to the IWC Scientific Committee. 33 pp. Bournemouth, September (SC/49/O9).

Gjertz, I., and A. Børset. 1992. Pupping in the most northerly harbor seal (*Phoca vitulina*). Marine Mammal Science 8(2):103–109.

Gockel, C., and T. Mongillo. 2013. Potential Effects of PBDEs on Puget Sound and Southern Resident Killer Whales: A Report on the Technical Workgroups and Policy Forum. EPA and NOAA.

Godley, B. J., D. R. Thompsonà, and R. W. Furness. 1999. Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? Marine Pollution Bulletin 38(6):497-502.

Goertner, J. F. 1978. Dynamical Model for Explosion Injury to Fish. U.S. Department of the Navy, Naval Surface Weapons Center, Dalgren, VA.

Goertner, J. F. 1982. Prediction of Underwater Explosion Safe Ranges for Sea Mammals. Naval Surface Weapons Center, NSWC TR 82-188, Dahlgren, VA.

Goertner, J. F., M. L. Wiley, G. A. Young, and W. W. McDonald. 1994. Effects of Underwater Explosions on Fish Without Swimbladders. Naval Surface Warfare Center, NSWC TR 88-114, Silver Spring, MD.

Goff, G. P., and J. Lien. 1988. Atlantic leatherback turtles, Dermochelys coriacea, in cold water off Newfoundland and Labrador. The Canadian Field-Naturalist 102:1-5.

Goldbogen, J. a., and coauthors. 2013. Blue whales respond to simulated mid-frequency military sonar. Proceedings of the Royal Society B: Biological Sciences 280(1765):20130657.

Gomez, C., and coauthors. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: the disparity between science and policy. Canadian Journal of Zoology.

Gong, Z., and coauthors. 2014. Ecosystem scale acoustic sensing reveals humpback whale behavior synchronous with herring spawning processes and re-evaluation finds no effect of sonar on humpback song occurrence in the Gulf of Maine in fall 2006. PLoS ONE 9(10):e10473.

Good, T. P., R. S. Waples, and P. Adams. 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Department of Commerce, NMFS-NWFSC-66, Seattle, Washington.

Goodwin, L., and P. A. Cotton. 2004. Effects of boat traffic on the behaviour of bottlenose dolphins (*Tursiops truncatus*). Aquatic Mammals 30(2):279-283.

Goodyear, C. P. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use. Canadian Special Publication of Fisheries and Aquatic Sciences:67-82.

Goold, J. C. 1999. Behavioural and acoustic observations of sperm whales in Scapa Flow, Orkney Islands. Journal of the Marine Biological Association of the United Kingdom 79(3):541-550.

Goold, J. C., and S. E. Jones. 1995. Time and frequency domain characteristics of sperm whale clicks. Journal of the Acoustical Society of America 98(3):1279-1291.

Goold, J. C., H. Whitehead, and R. J. Reid. 2002. North Atlantic sperm whale, Physeter macrocephalus, strandings on the coastlines of the British Isles and Eastern Canada. The Canadian Field-Naturalist 116:371–388.

Gordon, A. N., A. R. Pople, and J. Ng. 1998. Trace metal concentrations in livers and kidneys of sea turtles from south-eastern Queensland, Australia. Marine and Freshwater Research 49(5):409-414.

Gordon, J., and coauthors. 2003. A review of the effects of seismic surveys on marine mammals. Marine Technology Society Journal, 37(4), 16–34.

Gordon, J. C. D. 1987. The behaviour and ecology of sperm whales off Sri Lanka. (Physeter macrocephalus). University of Cambridge, Cambridge. 347 pp.

Götz, T., and V. M. Janik. 2010. Aversiveness of sounds in phocid seals: Psycho-physiological factors, learning processes and motivation. The Journal of Experimental Biology 213:1536–1548.

Götz, T., and V. M. Janik. 2011. Repeated elicitation of the acoustic startle reflex leads to sensation in subsequent avoidance behaviour and induces fear conditioning. BMC Neuroscience 12(30):13.

Götz, T., A. F. Pacini, P. Nachtigall, and V. M. Janik. 2020. The startle reflex in echolocating odontocetes: basic physiology and practical implications. Journal of Experiential Biology 223(5):12.

Graham, I. M., and coauthors. 2017. Responses of bottlenose dolphins and harbor porpoises to impact and vibration piling noise during harbor construction. Ecosphere 8(5):1–16.

Grant, S. C. H., and P. S. Ross. 2002. Southern Resident Killer Whales at Risk: Toxic Chemicals in the British Columbia and Washington Environment.

Green, G. A., and coauthors. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990. Oregon and Washington Marine Mammal and Seabird Surveys. Minerals Management Service Contract Report 14-12-0001-30426.

Greene, C., and A. Godersky. 2012. Larval rockfish in Puget Sound surface waters, Northwest Fisheries Science Center.

Greene, C. R., and S. E. Moore. 1995. Man-made Noise. Pp in Marine Mammals and Noise. Pages 101-158 *in* W. J. Richardson, C. R. Greene, C. I. Malme, and D. H. Thomson, editors. Marine Mammals and Noise. Academic Press, Inc., San Diego, California.

Greer, A. E., J. D. Lazell Jr., and R. M. Wright. 1973. Anatomical evidence for counter-current heat exchanger in the leatherback turtle (*Dermochelys coriacea*). Nature 244:181.

Gregory, L. F., and J. R. Schmid. 2001. Stress response and sexing of wild Kemp's ridley sea turtles (*Lepidochelys kempii*) in the Northeastern Gulf of Mexico. General and Comparative Endocrinology 124:66–74.

Gregr, E. J., L. Nichol, J. K. B. Ford, G. Ellis, and A. W. Trites. 2000. MIGRATION AND POPULATION STRUCTURE OF NORTHEASTERN PACIFIC WHALES OFF

COASTAL BRITISH COLUMBIA: AN ANALYSIS OF COMMERCIAL WHALING RECORDS FROM 1908-1967. Marine Mammal Science 16(4):699-727.

Gregr, E. J., and A. W. Trites. 2001. Predictions of critical habitat for five whale species in the waters of coastal British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 58(7):1265-1285.

Griffin, R. B. 1999. Sperm whale distributions and community ecology associated with a warm-core ring off Georges Bank. Marine Mammal Science 15(1):33-51.

Group, E. S. 2005. Canadian Forces Maritime Experimental and Test Ranges (CFMETR) Environmental Assessment Update 2005 (pp. 652). Kingston, Ontario: Environmental Sciences Group, Royal Military College.

Guazzo, R. A., and coauthors. 2020. The Lombard effect in singing humpback whales: Source levels increase as ambient ocean noise levels increase. The Journal of the Acoustical Society of America 148(2):542-555.

Guerra, M., S. M. Dawson, T. E. Brough, and W. J. Rayment. 2014. Effects of boats on the surface and acoustic behaviour of an endangered population of bottlenose dolphins. Endangered Species Research 24(3):221-236.

Guillen, G. 2003. Klamath River fish die-off, September 2002: causative factors of mortality. Report number AFWO-F-02-03. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office. Arcata, California 128 p.

Gulland, F. M. D., and A. J. Hall. 2007. Is marine mammal health deteriorating? Trends in the global reporting of marine mammal disease. Ecohealth 4(2):135-150.

Gustafson, R. G., editor. 2016. Status Review Update of Eulachon (*Thaleichthys pacificus*) Listed under the Endangered Species Act: Southern Distinct Population Segement.

Gustafson, R. G., and coauthors. 2012. Conservation status of eulachon in the California Current. Fish and Fisheries 13(2):121-138.

Gustafson, R. G., M. J. Ford, D. Teel, and J. S. Drake. 2010. Status review of eulachon (*Thaleichthys pacificus*) in Washington, Oregon,and California. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Haelters, J., V. Dulière, L. Vigin, and S. Degraer. 2014. Towards a numerical model to simulate the observed displacement of harbour porpoises *Phocoena phocoena* due to pile driving in Belgian waters. Hydrobiologia 756(1):105–116.

Hain, J. H. W., G. R. Carter, S. D. Kraus, C. A. Mayo, and H. E. Winn. 1982. Feeding behavior of the humpback whale, Megaptera novaeangliae, in the western North Atlantic. Fishery Bulletin 80(2):259-268.

Hain, J. H. W., and coauthors. 1995. Apparent bottom feeding by humpback whales on Stellwagen Bank. Marine Mammal Science 11(4):464-479.

Hain, J. H. W., M. A. M. Hyman, R. D. Kenney, and H. E. Winn. 1985. The role of cetaceans in the shelf-edge region of the Northeastern United States. Marine Fisheries Review 47(1):13-17.

Hain, J. H. W., M. J. Ratnaswamy, R. D. Kenney, and H. E. Winn. 1992. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Reports of the International Whaling Commission 42:653-669.

Hall, A. J., and coauthors. 2018. Predicting the effects of polychlorinated biphenyls on cetacean populations through impacts on immunity and calf survival. Environmental Pollution 233:407-418.

Hall, J. D. 1982. Prince William Sound, Alaska: Humpback whale population and vessel traffic study. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, Juneau Management Office, Contract No. 81-ABG-00265., Juneau, Alaska.

Haller, L. Y., and coauthors. 2015. Effect of nutritional status on the osmoregulation of green sturgeon (Acipenser medirostris). 88(1):22-42.

Hallock, R. J., W.F. Van Woert, and L. Shapovalov. 1961. An evaluation of stocking hatchery reared steelhead rainbow trout (Salmo gairdnerii gairdnerii) in the Sacramento River system.

Halstead, B. W., P. S. Auerbach, and D. R. Campbell. 1990. A colour atlas of dangerous marine animals. Wolfe Medical Publications Ltd, W.S. Cowell Ltd, Ipswich, England.

Halvorsen, M. B., W. T. Ellison, D. R. Choicoine, and A. N. Popper. 2012. Effects of mid-frequency active sonar on hearing in fish. Journal of the Acoustical Society of America 131(1):599-607.

Hamilton, P. K., G. S. Stone, and S. M. Martin. 1997. Note on a deep humpback whale (Megaptera novaeangliae) dive near Bermuda. Bulletin of Marine Science 61(2):491-494.

Hamilton, T. A., and coauthors. 2009. Atlas of cetacean sightings from Southwest Fisheries Science Center cetacean ecosystem surveys: 1986-2005. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Hance, A. J., and coauthors. 1982. Hormonal changes and enforced diving in the harbor seal Phoca vitulina. II. Plasma catecholamines. American Journal of Physiology - Regulatory Integrative and Comparative Physiology 242(5):R528-R532.

Hanlon, R. T., and J. B. Messenger. 1996. Cephalopod Behaviour. Cambridge University Press, Cambridge, New York.

Hannah, R. W., and S. A. Jones. 2007. Effectiveness of bycath reduction devices (BRDs) in the ocean shrimp (*Pandalus jordani*) trawl fishery. Fisheries Research 85(1-2):217-225.

Hanson, M. B., and coauthors. 2010. Species and stock identification of prey consumed by endangered southern resident killer whales in their summer range. Endangered Species Research 11(1):69-82.

Hanson, M. B., C.K. Emmons, M.J. Ford, K. Parsons, J. Hempelmann, D.M.V. Doornik, G.S. Schorr, J. Jacobsen, M. Sears, J.G. Sneva, R.W. Baird and L. Barre. In prep. Seasonal diet of Southern Resident Killer Whales.

Hanson, M. B., E. J. Ward, C. K. Emmons, and M. M. Holt. 2018. Modeling the occurrence of endangered killer whales near a U.S. Navy Training Range in Washington State using satellite-tag locations to improve acoustic detection data. Northwest Fisheries Science Center, Seattle, WA.

Hanson, M. B., E. J. Ward, C. K. Emmons, M. M. Holt, and D. M. Holzer. 2017a. Assessing the Movements and Occurrence of Southern Resident Killer Whales Relative to the U.S. Navy's Northwest Training Range Complex in the Pacific Northwest. Northwest Fisheries Science Center, MIPR N00070-15-MP-4C363, Seattle, WA.

Hanson, M. B., E. J. Ward, C. K. Emmons, M. M. Holt, and D. M. Holzer. 2017b. Assessing the movements and occurrence of Southern Resident Killer Whales relative to the U.S. Navy's Northwest Training Range Complex in the Pacific Northwest.

Harcourt, R., V. Pirotta, G. Heller, V. Peddemors, and D. Slip. 2014. A whale alarm fails to deter migrating humpback whales: An empirical test. Endangered Species Research 25(1):35-42.

Hare, S. R., and N. J. Mantua. 2001. An historical narrative on the Pacific Decadal Oscillation, interdecadal climate variability and ecosystem impacts. University of Washington.

Hare, S. R., N. J. Mantua, and R. C. Francis. 1999. Inverse production regimes: Alaska and West Coast Pacific salmon. Fisheries 24(1):6-14.

Hargrove, S. T., T. Work, S. Brunson, A. M. Foley, and G. Balazs. 2016. Proceedings of the 2015 international summit on fibropapillomatosis: global status, trends, and population impacts. U.S. Department of Commerce, NOAA Technical Memo.

Harrington, F. H., and A. M. Veitch. 1992. Calving success of woodland caribou exposed to low-level jet fighter overflights. Arctic 45(3):213-218.

Harris, C. M., and coauthors. 2019. Changes in the Spatial Distribution of Acoustically Derived Minke Whale (Balaenoptera acutorostrata) Tracks in Response to Navy Training. Aquatic Mammals 45(6):661-674.

Harris, C. M., and coauthors. 2015. Dose response severity functions for acoustic disturbance in cetaceans using recurrent event survival analysis. Ecosphere 6(11):Article 236.

Harris, C. M., and L. Thomas. 2015. Status and future of research on the behavioral responses of marine mammals to US. Navy sonar. University of St. Andrews, Centre for Research into Ecological & Environmental Modelling (CREEM).

Harris, C. M., and coauthors. 2017a. Marine mammals and sonar: dose-response studies, the risk-disturbance hypothesis and the role of exposure context. Journal of Applied Ecology:1-9.

Harris, C. M., L. J. Wilson, C. G. Booth, and J. Harwood. 2017b. Population consequences of disturbance: A decision framework to identify priority populations for PCoD modelling. 22nd Biennial Conference on the Biology of Marine Mammals, Halifax, Nova Scotia, Canada.

Harris, H. S., and coauthors. 2011. Comparative health assessment of western Pacific leatherback turtles (Dermochelys coriacea) foraging off the coast of California, 2005–2007. Journal of Wildlife Diseases 47(2):321-337.

Hart, G. A. 2012. Environmental Monitoring Report. U.S. Pacific Fleet, Pearl Harbor, HI.

Hart, J. L. 1973. Pacific fishes of Canada. Bulletin of the Fisheries Research Board of Canada 180.

Hartt, A. C., and M. B. Dell. 1986. Early oceanic migrations and growth of juvenile Pacific salmon and steelhead trout. International North Pacific Fisheries Commission, Vancouver, Canada.

Harvey, C. J., N. Tolimieri, and P. S. Levin. 2006. Changes in body size, abundance, and energy allocation in rockfish assemblages of the Northeast Pacific. Ecological Applications 16(4):1502-1515.

Harwood, J., and coauthors. 2014. An interim framework for assessing the population consequences of acoustic disturbance. Pages 33 *in* Fifth International Meeting on the Effects of Sounds in the Ocean on Marine Mammals (ESOMM - 2014), Amsterdam, The Netherlands.

Harwood, L. A., and I. Stirling. 1987. Patterns of aggregation in ringed seals, bearded seals and bowhead whales in the Beaufort Sea during late summer. Seventh Biennial Conference on the Biology of Marine Mammals, 5-9 December Miami Florida. p.29.

Harwood, M. B., and D. Hembree. 1987. Incidental catch of small cetaceans in the offshore gillnet fishery in northern Australian waters: 1981-1985. Report of the International Whaling Commission 37:363-367.

Hassler, T. J. 1987. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest). Coho Salmon. DTIC Document.

Hassrick, J. L., and coauthors. 2016. Early ocean distribution of juvenile Chinook salmon in an upwelling ecosystem. Fisheries Oceanography 25(2):133-146.

Hastings, K. K., and coauthors. 2020. Demographic consequences and characteristics of recent population mixing and colonization in Steller sea lions, *Eumetopias jubatus*. Journal of Mammalogy 101(1):107-120.

Hastings, M., A. Popper, J. Finneran, and P. Lanford. 1996. Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish Astronotus ocellatus. Journal of the Acoustical Society of America 99(3):1759-1766.

Hastings, M. C. 1990. Effects of underwater sound on fish. AT&T Bell Laboratories.

Hastings, M. C. 1995. Physical effects of noise on fishes. The 1995 International Congress on Noise Control Engineering.

Hatch, L., and A. J. Wright. 2007a. A brief review of anthropogenic sound in the oceans. International Journal of Comparative Psychology 20:12.

Hatch, L. T., C. W. Clark, S. M. V. Parijs, A. S. Frankel, and D. W. Ponirakis. 2012. Quantifying loss of acoustic communication space for right whales in and around a US. National Marine Sanctuary. Conservation Biology 26(6):983-994.

Hatch, L. T., and A. J. Wright. 2007b. A brief review of anthropogenic souond in the oceans. International Journal of Comparative Psychology 201(2-3):121-133.

Hatchery Scientific Review Group. 2015. Annual Report to Congress on the Science of Hatcheries, 2015: A report on the application of up-to-date science in the management of salmon and steelhead hatcheries in the Pacific Northwest.

Hawkins, A. D., C. Johnson, and A. N. Popper. 2020. How to set sound exposure criteria for fishes. The Journal of the Acoustical Society of America 147(3):1762-1777.

Hawkins, A. D., and A. D. F. Johnstone. 1978. The hearing of the Atlantic salmon, *Salmo salar*. Journal of Fish Biology 13:655–673.

Hay, D. E., and P. B. McCarter. 2000. Status of the eulachon *Thaleichthys pacificus* in Canada. Department of Fisheries and Oceans Canada. Canadian Stock Assessment Secretariat, Ottawa, Ontario.

Hayes, S. A., M. H. Bond, C. V. Hanson, and R. B. MacFarlane. 2004. Interactions between endangered wild and hatchery salmonids: can the pitfalls of artificial propagation be avoided in small coastal streams? Journal of Fish Biology 65:101 - 121.

Hayes, S. A., and coauthors. 2018. US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2017. Northeast Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NE-245, Woods Hole, Massachusetts.

Hayhoe, K., and coauthors. 2018. In *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* (Reidmiller, D.R., et al. [eds.]). U.S. Global Change Research Program, Washington, DC, USA.

Hayman, R. A., E. M. Beamer, and R. E. McClure. 1996. FY 1995 Skaig River chinook restoration research. Report by Skagit System Cooperative, La Conner, Washington:54p + Appendices.

Hays, G. C. 2000. The implications of variable remigration intervals for the assessment of population size in marine turtles. J Theor Biol 206(2):221-7.

Hazel, J., I. R. Lawler, H. Marsh, and S. Robson. 2007. Vessel speed increases collision risk for the green turtle *Chelonia mydas*. Endangered Species Research 3:105-113.

Hazen, E. L., and coauthors. 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change 3(3):234-238.

Hazen, E. L., and coauthors. 2016. WhaleWatch: A dynamic management tool for predicting blue whale density in the California Current. Journal of Applied Ecology 54(5):1415-1428.

HCCC. 2005. Hood Canal and Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan.

HDR. 2011. Jacksonville (JAX) Southeast Anti-Submarine Warfare Integration Training Initiative (SEASWITI) Marine Species Monitoring Aerial Monitoring Surveys Trip Report, 3-5 December 2010. Prepared by HDR Environmental Operations & Construction, Inc. (HDR EOC) under Contract # N62470-10-D-3011.

Healey, M., C. Groot, and L. Margolis. 1991. Life history of chinook salmon (Oncorhynchus tshawytscha). Pacific salmon life histories:313-393.

Healey, M. C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). Pages 311-394 *in* C. Groot, and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, Canada.

Heimlich-Boran, J. R. 1988. Behavioral ecology of killer whales (*Orcinus orca*) in the Pacific Northwest. Canadian Journal of Zoology 66:565-578.

Heimlich-Boran, S. L. 1986. Cohesive relationships among Puget Sound killer whales. Pages 251-284 *in* B. C. Kirkevold, and J. S. Lockard, editors. Behavioral Biology of Killer Whales. Alan R. Liss, Inc. , New York.

Helfman, G., B. B. Collette, D. E. Facey, and B. W. Bowen. 2009. The Diversity of Fishes: Biology, Evolution, and Ecology. Wiley.

Helker, V. T., and coauthors. 2017. Human-Caused Mortality and Injury of NMFS-Managed Alaska Marine Mammal Stocks, 2011–2015. Alaska Fisheries Science Center, Seattle, WA.

Helweg, D. A., A. S. Frankel, J. Joseph R. Mobley, and L. M. Herman. 1992. Humpback whale song: Our current understanding. Pages 459-483 *in* J. A. Thomas, R. A. Kastelein, and A. Y. Supin, editors. Marine Mammal Sensory Systems. Plenum Press, New York.

Henderson, D. 1990. Gray whales and whalers on the China coast in 1869. (Eschrichtius robustus). Whalewatcher 24(4):14-16.

Henderson, E. E., J. Aschettino, M. Deakos, G. Alongi, and T. Leota. 2019. Quantifying the Behavior of Humpback Whales (Megaptera novaeangliae) and Potential Responses to Sonar. Aquatic Mammals 45(6):612-631.

Henderson, E. E., R. Manzano-Roth, C. Martin, and B. Matsuyama. 2015. Impacts of U.S. Navy training events on beaked whale foraging dives in Hawaiian waters: Update. San Diego, CA: SPAWAR Systems Center Pacific.

Henderson, E. E., S. W. Martin, S. W. Martin, and B. Matsuyama. 2016. Occurrence and habitat use of foraging Blainville's beaked whales (Mesoplodon densirostris) on a U.S. Navy range in Hawai'i. Aquatic Mammals, 42(4). .

Henderson, E. E., and coauthors. 2014. Delphinid behavioral responses to incidental mid-frequency active sonar. Journal of the Acoustical Society of America 136(4-Jan):2003-2014.

Henry, J., and P. B. Best. 1983. Organochlorine residues in whales landed at Durban, South Africa. Marine Pollution Bulletin 14(6):223-227.

Heppell, S. S. 2007. Elasticity analysis of green sturgeon life history. Environmental Biology of Fishes 79(3-4):357-368.

Herman, D. P., and coauthors. 2005a. Feeding ecology of eastern North Pacific killer whales Orcinus orca from fatty acid, stable isotope, and organochlorine analyses of blubber biopsies. Marine Ecology-progress Series - MAR ECOL-PROGR SER 302:275-291.

Herman, D. P., and coauthors. 2005b. Feeding ecology of eastern North Pacific killer whales *Orcinus orca* from fatty acid, stable isotope, and organochlorine analyses of blubber biopsies. Marine Ecology Progress Series 302:275-291.

Heublein, J. C., and coauthors. 2017. Life history and current monitoring inventory of San Francisco Estuary sturgeon. NOAA Tech. Memo. NMFS-SWFSC-589.

Heublein, J. C., J. T. Kelly, C. E. Crocker, A. P. Klimley, and S. T. Lindley. 2009. Migration of green sturgeon, Acipenser medirostris, in the Sacramento River. Environmental Biology of Fishes 84(3):245-258.

Hewitt, R. P. 1985. Reaction of dolphins to a survey vessel: Effects on census data. Fishery Bulletin 83(2):187-194.

Heyning, J. E., and T. D. Lewis. 1990. Entanglements of baleen whales in fishing gear off southern California. (*Eschrichtius robustus, Balaenoptera acutorostrata, Megaptera novaeangliae*). Report of the International Whaling Commission 40:427-431.- Sc/41/Ps14).

Hilborn, R., and coauthors. 2012. The Effects of Salmon Fisheries on Southern Resident Killer Whales: Final Report of the Independent Science Panel. Prepared with the assistance of D.R. Marmorek and A.W. Hall, ESSA Technologies Ltd., Vancouver, B.C. for National Marine Fisheries Service (Seattle. WA) and Fisheries and Oceans Canada (Vancouver. BC). xv + 61 pp. + Appendices.

Hildebrand, J. 2004a. Impacts of anthropogenic sound on cetaceans. Unpublished paper submitted to the International Whaling Commission Scientific Committee SC/56 E 13.

Hildebrand, J. 2004b. Sources of anthropogenic sound in the marine environment. University of California, San Diego, Scripps Institution of Oceanography.

Hildebrand, J. A. 2005. Impacts of anthropogenic sound. Pages 101-124 *in* J. E. Reynolds, editor. Marine Mammal Research: Conservation Beyond Crisis. The John Hopkins University Press.

Hildebrand, J. A. 2009. Metrics for characterizing the sources of ocean anthropogenic noise. Journal of the Acoustical Society of America 125(4):2517.

Hildebrand, J. A., and coauthors. 2011. Passive Acoustic Monitoring for Marine Mammals in the SOCAL Naval Training Area 2010-2011. Marine Physical Laboratory, Scripps Institution of Oceanography, University of California San Diego, MPL Technical Memorandum #531, La Jolla, California.

Hildebrand, J. A., and coauthors. 2012. Passive Acoustic Monitoring for Marine Mammals in the SOCAL Naval Training Area 2011-2012. Marine Physical Laboratory, Scripps Institution of Oceanography, University of California San Diego, MPL Technical Memorandum #537, La Jolla, California.

Hill, P. S., and D. P. DeMaster. 1998. Alaska Marine Mammal Stock Assessments, 1998. U.S. Department of Commerce, NMFS-AFSC-97.

Hill, P. S., J. L. Laake, and E. Mitchell. 1999. Results of a pilot program to document interactions between sperm whales and longline vessels in Alaska waters. NOAA Technical Memorandum NMFS-AFSC-108. 51p.

Hinrichsen, R. A., and C. M. Paulsen. 2020. Low carrying capacity a risk for threatened Chinook Salmon. Ecological Modelling 432:109223.

Hirth, H., J. Kasu, and T. Mala. 1993. Observations on a leatherback turtle *Dermochelys coriacea* nesting population near Piguwa, Papua New Guinea. Biological Conservation 65:77-82.

Hochachka, P. W., and coauthors. 1995. Hormonal regulatory adjustments during voluntary diving in Weddell seals. Comparative Biochemistry and Physiology B Biochemistry and Molecular Biology 112(2):361-375.

Hodge, R. P., and B. L. Wing. 2000. Occurrences of marine turtles in Alaska Waters: 1960-1998. Herpetological Review 31(3):148-151.

Holsman, K., and coauthors. 2019. Climate change impacts, vulnerabilities, and adaptations: North Pacific and Pacific Arctic marine fisheries. Impacts of climate change on fisheries and aquaculture.

Holsman, K. K., M. D. Scheuerell, E. Buhle, and R. Emmett. 2012. Interacting effects of translocation, artificial propagation, and environmental conditions on the marine survival of Chinook salmon from the Columbia River, Washington, USA. Conservation Biology 26(5):912-922.

Holst, M., and coauthors. 2011. Responses of pinnipeds to Navy missile launches at San Nicolas Island, California. Aquatic Mammals 37(2):139-150.

Holt, M., V. Veirs, and S. Veirs. 2008. Investigating noise effects on the call amplitude of endangered Southern Resident killer whales (*Orcinus orca*). Journal of the Acoustical Society of America 123(5 Part 2):2985.

Holt, M. M. 2008. Sound exposure and Southern Resident killer whales (Orcinus orca): A review of current knowledge and data gaps. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Holt, M. M., M. B. Hanson, D. A. Giles, C. K. Emmons, and J. T. Hogan. 2017. Noise levels received by endangered killer whales *Orcinus orca* before and after implementation of vessel regulations. Endangered Species Research 34:15-26.

Holt, M. M., D. P. Noren, R. C. Dunkin, and T. M. Williams. 2015. Vocal performance affects metabolic rate in dolphins: implications for animals communicating in noisy environments. Journal of Experimental Biology 218(11):1647-1654.

Holt, M. M., D. P. Noren, and C. K. Emmons. 2011. Effects of noise levels and call types on the source levels of killer whale calls. Journal of the Acoustical Society of America 130(5):3100-3106.

Holt, M. M., D. P. Noren, V. Veirs, C. K. Emmons, and S. Veirs. 2009. Speaking up: Killer whales (Orcinus orca) increase their call amplitude in response to vessel noise. Journal of the Acoustical Society of America 125(1):EL27-EL32.

Hooker, S. K., R. W. Baird, and A. Fahlman. 2009. Could beaked whales get the bends? Effect of diving behaviour and physiology on modelled gas exchange for three species: Ziphius cavirostris, Mesoplodon densirostris and Hyperoodon ampullatus. Respiratory Physiology and Neurobiology 167(3):235-246.

Hooker, S. K., and coauthors. 2012. Deadly diving? Physiological and behavioural management of decompression stress in diving mammals. Proceedings of the Royal Society of London Series B Biological Sciences 279(1731):1041-1050.

Hoopes, L. A., A. M. Landry Jr., and E. K. Stabenau. 2000. Physiological effects of capturing Kemp's ridley sea turtles, *Lepidochelys kempii*, in entanglement nets. Canadian Journal of Zoology 78(11):1941–1947.

Hoover, A. A. 1988. Harbor Seal (*Phoca vitulina)*. Pages 125–157 *in* J. W. Lentfer, editor. Selected Marine Mammals of Alaska: Species Accounts with Research and Management Recommendations. Marine Mammal Commission, Washington, DC.

Hore, P. 2012. Are biochemical reactions affected by weak magnetic fields? Proceedings of the National Academy of Sciences, 109(5), 1357–1358. .

Horwood, J. W. 1987. The sei whale: Population biology, ecology, and management. Croom Helm Ltd., Kent, England.

Hotchkin, C., and S. Parks. 2013. The Lombard effect and other noise-induced vocal modifications: Insight from mammalian communication systems. Biological Reviews 88(4):809-824.

Houser, D., R. Howard, and S. Ridgway. 2001. Can diving behavior increase the chance of acoustically driven bubble growth in marine mammals? Pages 103 *in* Fourteenth Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Houser, D., S. W. Martin, and J. Finneran. 2013a. Exposure amplitude and repetition affect bottlenose dolphin behavioral responses t esponses to simulated mid-fr o simulated mid-frequency sonar signals. Journal of Experimental Marine Biology and Ecology 443:123-133.

Houser, D. S., L. A. Dankiewicz-Talmadge, T. K. Stockard, and P. J. Ponganis. 2009. Investigation of the potential for vascular bubble formation in a repetitively diving dolphin. Journal of Experimental Biology 213(1):52-62.

Houser, D. S., S. Martin, D. E. Crocker, and J. J. Finneran. 2020. Endocrine response to simulated US Navy mid-frequency sonar exposures in the bottlenose dolphin (*Tursiops truncatus*). The Journal of the Acoustical Society of America 147(3):1681-1687.

Houser, D. S., S. W. Martin, and J. J. Finneran. 2013b. Behavioral responses of California sea lions to mid-frequency (3250-3450 Hz) sonar signals. Marine Environmental Research 92:268–278.

Houser, D. S., S. W. Martin, and J. J. Finneran. 2013c. Exposure amplitude and repetition affect bottlenose dolphin behavioral responses to simulated mid-frequency sonar signals. Journal of Experimental Marine Biology and Ecology 443:123-133.

Houser, D. S., L. C. Yeates, and D. E. Crocker. 2011. Cold stress induces an adrenocortical response in bottlenose dolphins (Tursiops truncatus). Journal of Zoo and Wildlife Medicine 42(4):565-571.

Houston, J. J. 1988. Status of green sturgeon, Acipenser medirostris, in Canada. Canadian field-naturalist 102:286-290.

Howell, P., and coauthors. 1985. Stock assessment of Columbia River anadromous salmonids. Volume II: Steelhead stock summaries stock transfer guidelines-information needs. Final

report to Bonneville Power Administration. Bonneville Power Administration, DE-A179-84BP12737, Project 83-335, Portland, Oregon.

Hoyt, E. 2001. Whale Watching 2001: Worldwide Tourism Numbers, Expenditures, and Expanding Socioeconomic Benefits. International Fund for Animal Welfare,, Yarmouth Port, MA, USA.

Hubbs, C., and A. Rechnitzer. 1952a. Report on experiments designed to determine effects of underwater explosions on fish life. California Fish and Game 38:333–366.

Hubbs, C. L. 1956. Back from oblivion. Guadalupe fur seal: Still a living species. Pacific Discovery 9(6):14-21.

Hubbs, C. L., and A. B. Rechnitzer. 1952b. Report on experiments designed to determine effects of underwater explosions on fish life. Pages 333–366 *in* California Fish and Game Bulletin. California Department of Fish and Game, San Diego, CA.

Huff, D. D., S. T. Lindley, P. S. Rankin, and E. A. Mora. 2011. Green sturgeon physical habitat use in the coastal Pacific Ocean. PLoS ONE 6(9).

Huff, D. D., S. T. Lindley, B. K. Wells, and F. Chai. 2012. Green sturgeon distribution in the Pacific Ocean estimated from modeled oceanographic features and migration behavior. PLoS ONE 7(9):e45852.

Huijser, L. A. E., and coauthors. 2018. Population structure of North Atlantic and North Pacific sei whales (Balaenoptera borealis) inferred from mitochondrial control region DNA sequences and microsatellite genotypes. Conservation Genetics.

Hullar, T. L., and coauthors. 1999. Environmental effects of RF chaff: A select panel report to the Undersecretary of Defense for Environmental Security. Naval Research Laboratory, Washington, DC.

Hume, M. 2013. "Miracle" whale leaves scientists elated – and relieved. The Globe and Mail, The Globe and Mail, http://www.theglobeandmail.com/news/british-columbia/miraclewhale-leaves-scientists-elated-and-relieved/article12735936/.

Hurford, W. E., and coauthors. 1996. Splenic contraction, catecholamine release, and blood volume redistribution during diving in the Weddell seal. Journal of Applied Physiology 80(1):298-306.

Hurrell, J. W. 1995. Decadal trends in the North Atlantic Oscillation: Regional temperatures and precipitation. Science 269:676-679.

Ichihara, T. 1966. The pygmy blue whale, Balaenoptera musculus brevicauda, a new subspecies from the Antarctic. Whales, Dolphins and Porpoises. K. S. Norris (ed.). University of California Press, Berkeley, CA. p.79-113.

ICTRT. 2003. Independent populations of chinook, steelhead, and sockeye for listed evolutionarily significant units within the Interior Columbia River Domain. NMFS, Northwest Fisheries Science Center, Seattle, Washington.

ICTRT. 2008a. Entiat spring Chinook population. NMFS, Northwest Fisheries Science Center, Seattle, Washington.

ICTRT. 2008b. Methow spring Chiniook salmon. NMFS, Northwest Fisheries Science Center, Seattle, Washington.

ICTRT. 2008c. Wenatchee River spring Chinook population. NMFS, Northwest Fisheries Science Center, Seattle, Washington.

Ilyashenko, V. Y. 2009. How isolated is the 'western' gray whale population? International Whaling Commission Scientific Committee, Madeira, Portugal.

Incardona, J. P., and coauthors. 2015. Very low embryonic crude oil exposures cause lasting cardiac defects in salmon and herring. Scientific Reports 5(1):13499.

Indeck, K. L., E. Girola, M. Torterotot, M. J. Noad, and R. A. Dunlop. 2020. Adult female-calf acoustic communication signals in migrating east Australian humpback whales. Bioacoustics.

Independent Science Advisory Board. 2007. Climate change impacts on Columbia River Basin fish and wildlife. Northwest Power and Conservation Council, Portland, Oregon.

IPCC. 2014. Climate change 2014: Impacts, adaptation, and vulnerability. IPCC Working Group II contribution to AR5. Intergovernmental Panel on Climate Change.

IPCC. 2018. Summary for Policymakers. In: *Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [Masson-Delmotte, V., P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J.B.R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, Maycock, M. Tignor, and T. Waterfield (eds.)]. World Meteorological Organization, Geneva, Switzerland:32pp.

Ishida, Y., A. Yano, M. Ban, and M. Ogura. 1997. Vertical Movement of Chum Salmon, Oncorhynchus keta, in the western North Pacific Ocean as Determined by a Depth-recording Archival Tag. National Research Institute of Far Seas Fisheries, Japan.

Isojunno, S., and coauthors. 2016. Sperm whales reduce foraging effort during exposure to 1-2 kHz sonar and killer whale sounds. Ecological Applications 26(1):77-93.

Isojunno, S., and coauthors. 2020. When the noise goes on: received sound energy predicts sperm whale responses to both intermittent and continuous navy sonar. Journal of Experiential Biology 223:15pp.

Israel, J. A., K. J. Bando, E. C. Anderson, and B. May. 2009. Polyploid microsatellite data reveal stock complexity among estuarine North American green sturgeon (Acipenser medirostris). Canadian Journal of Fisheries and Aquatic Sciences 66(9):1491-1504.

Israel, J. A., J. F. Cordes, M. A. Blumberg, and B. May. 2004. Geographic patterns of genetic differentiation among collections of green sturgeon. North American Journal of Fisheries Management 24(3):922-931.

IUCN. 2010. IUCN Red List of Threatened Species. Version 2010.4.

IUCN. 2012. The IUCN red list of threatened species. Version 2012.2. International Union for Conservation of Nature and Natural Resources.

Ivashin, M. V., and A. A. Rovnin. 1967. Some results of the Soviet whale marking in the waters of the North Pacific. Norsk Hvalfangst-Tidende 56(6):123-135.

IWC. 2003. Report of the workshop on the western gray whale: Research and monitoring needs. International Whaling Commission.

IWC. 2005. Annex K: Report of the standing working group on environmental concerns. International Whaling Commission.

IWC. 2007. Whale population estimates. International Whaling Commission.

IWC. 2012. Extracts from the IWC64 Scientific Committee report relevant to the WGWAP. International Whaling Commission.

IWC. 2016. Report of the Scientific Committee. Journal of Cetacean Research and Management (Supplement) 17.

Jacobsen, J. K., L. Massey, and F. Gulland. 2010. Fatal ingestion of floating net debris by two sperm whales (Physeter macrocephalus). Marine Pollution Bulletin 60(5):765-767.

Jahoda, M., and coauthors. 2003. Mediterranean Fin Whale's (Balaenoptera physalus) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. Marine Mammal Science 19(1):96-110.

Jambeck, J. R., and coauthors. 2015. Plastic waste inputs from land into the ocean. Science 347(6223):768-771.

James, M. C., R. A. Myers, and C. A. Ottensmeyer. 2005. Behaviour of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. Proceedings of the Royal Society Biological Sciences Series B 272(1572):1547-1555.

James, M. C., S. A. Sherrill-Mix, and R. A. Myers. 2007. Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series 337:245-254.

Jansen, J. K., P. L. Boveng, S. P. Dahle, and J. L. Bengtson. 2010. Reaction of harbor seals to cruise ships. Journal of Wildlife Management 74(6):1186–1194.

Jaquet, N. 1996. How spatial and temporal scales influence understanding of sperm whale distribution: A review. (Physeter macrocephalus). Mammal Review 26(1):51-65.

Jaquet, N., and D. Gendron. 2009. The social organization of sperm whales in the Gulf of California and comparisons with other populations. Journal of the Marine Biological Association of the United Kingdom 89(5):975-983.

Jaquet, N., and H. Whitehead. 1996. Scale-dependent correlation of sperm whales distribution with environmental features and productivity in the South Pacific. Marine Ecology Progress Series 135:1-9.

Jarvela-Rosenberger, A. L., M. MacDuffee, A. G. J. Rosenberger, and P. S. Ross. 2017. Oil Spills and Marine Mammals in British Columbia, Canada: Development and Application of a Risk-Based Conceptual Framework. Archives of Environmental Contamination and Toxicology 73:131-153.

Jasny, M., J. Reynolds, C. Horowitz, and A. Wetzler. 2005. Sounding the depths II: The rising toll of sonar, shipping and industrial ocean noise on marine life. Natural Resources Defense Council, New York, New York.

Jauniaux, T., and coauthors. 2000. Pathological findings in two fin whales (Balaenoptera physalus) with evidence of morbillivirus infection. Journal of comparative pathology 123(2-3):198-201.

Jay, A., and coauthors. 2018. In: *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA:33-71.

Jefferson, T. A., and B. E. Curry. 1996. Acoustic methods of reducing or eliminating marine mammal-fishery interactions: Do they work? Ocean and Coastal Management 31(1):41-70.

Jefferson, T. A., and coauthors. 2008. *Delphinapterus leucas*. IUCN Red List of Threatened Species. Version 2010.4.

Jefferson, T. A., P. J. Stacey, and R. W. Baird. 1991. A review of killer whale interactions with other marine mammals: Predation to co-existence. (Orcinus orca). Mammal Review 21(4):151-180.

Jeffries, S. 2014. Aerial Surveys of Pinniped Haulout Sites in Pacific Northwest Inland Waters. Final Report (Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, Hawaii). Naval Facilities Engineering Command (NAVFAC) Northwest, Silverdale, Washington.

Jeffries, S. J., P. J. Gearin, H. R. Huber, D. L. Saul, and D. A. Pruett. 2000. Atlas of Seal and Sea Lion Haulout Sites in Washington. Washington Department of Fish and Wildlife, Wildlife Science Division, Olympia, Washington.

Jemison, L. A., and coauthors. 2013. Inter-population movements of Steller sea lions in Alaska with implications for population separation. PLoS ONE 8(8):e70167.

Jensen, A. S., and G. K. Silber. 2003. Large whale ship strike database. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-OPR.

Jensen, A. S., and G. K. Silber. 2004. Large Whale Ship Strike Database. U.S. Department of Commerce, NMFS-OPR-25.

Jepson, P. D., and coauthors. 2003. Gas-bubble lesions in stranded cetaceans: Was sonar responsible for a spate of whale deaths after an Atlantic military exercise? Nature 425(6958):575-576.

Jessop, T. S. 2001. Modulation of the adrenocortical stress response in marine turtles (Cheloniidae): evidence for a hormonal tactic maximizing maternal reproductive investment Journal of Zoology 254:57-65.

Jessop, T. S., M. Hamann, M. A. Read, and C. J. Limpus. 2000. Evidence for a hormonal tactic maximizing green turtle reproduction in response to a pervasive ecological stressor. General and Comparative Endocrinology 118:407-417.

Jessop, T. S., J. Sumner, V. Lance, and C. Limpus. 2004. Reproduction in shark-attacked sea turtles is supported by stress-reduction mechanisms. Proceedings of the Royal Society Biological Sciences Series B 271:S91-S94.

Jessop, T. S., A. D. Tucker, C. J. Limpus, and J. M. Whittier. 2003. Interactions between ecology, demography, capture stress, and profiles of corticosterone and glucose in a free-living population of Australian freshwater crocodiles. General and Comparative Endocrinology 132(1):161–170.

Jochens, A. E., and coauthors. 2006. Sperm whale seismic study in the Gulf of Mexico; Summary report 2002-2004. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, OCS Study MMS 2006-034, New Orleans, Louisiana.

Johnson, M. A., T. A. Friesen, D. J. Teel, and D. M. V. Doornik. 2013. Genetic stock identification and relative natural production of Willamette River steelhead.

Johnson, O., and coauthors. 1997. Status review of chum salmon from Washington, Oregon, and California, Seattle, WA.

Jones, E. L., and coauthors. 2017. Seals and shipping: Quantifying population risk and individual exposure to vessel noise. Journal of Applied Ecology 54(6):1930–1940.

Jones, K., E. Ariel, G. Burgess, and M. Read. 2015. A review of fibropapillomatosis in green turtles (*Chelonia mydas*). The Veterinary Journal 212:48-57.

Jones, T. T., and coauthors. 2018. Review of draft response to PRD's request for information to support ESA section 7 consultation on the effects of Hawaii-based longline fisheries on ESA listed species. NMFS Pacific Islands Fisheries Science Center, Honolulu, HI.

Jørgensen, R., N. O. Handegard, H. Gjøsæter, and A. Slotte. 2004. Possible vessel avoidance behaviour of capelin in a feeding area and on a spawning ground. Fisheries Research 69(2):251–261.

Jorgensen, R., K. Olsen, I. Petersen, and P. Kanapthipplai. 2005. Investigations of potential effects of low frequency sonar signals on survival, development and behaviour of fish larvae and juveniles. University of Tromso, The Norwegian College of Fishery Science.

Joy, R., and coauthors. 2019a. Potential benefits of vessel slowdowns on endangered Southern Resident killer whales. Frontiers in Marine Science 6:20.

Joy, R., and coauthors. 2019b. Potential Benefits of Vessel Slowdowns on Endangered Southern Resident Killer Whales. Frontiers in Marine Science 6(344).

Joyce, T. W., and coauthors. 2019. Behavioral responses of satellite tracked Blainville's beaked whales (Mesoplodon densirostris) to mid-frequency active sonar. Marine Mammal Science.

Jurasz, C. M., and V. Jurasz. 1979. Feeding modes of the humpback whale, Megaptera novaeangliae, in southeast Alaska. Scientific Reports of the Whales Research Institute, Tokyo 31:69-83.

Kahnle, A., and coauthors. 1998. Stock status of Atlantic sturgeon of Atlantic Coast estuaries. Report for the Atlantic States Marine Fisheries Commission. Draft III.

Kakuschke, A., and A. Prange. 2007. The influence of metal pollution on the immune system a potential stressor for marine mammals in the North Sea. International Journal of Comparative Psychology 201(2-3):179-193.

Kalmijn, A. J. 2000. Detection and processing of electromagnetic and near-field acoustic signals in elasmobranch fishes. Philosophical Transactions of the Royal Society of London Series B-Biological Sciences 355(1401):1135-1141.

Kanda, N., M. Goto, V. Y. Ilyashenko, and L. A. Pastene. 2010. Preliminary mtDNA analysis of gray whales from Japan and Russia. International Whaling Commission.

Kanda, N., M. Goto, K. Matsuoka, H. Yoshida, and L. A. Pastene. 2011. Stock identity of sei whales in the central North Pacific based on microsatellite analysis of biopsy samples obtained from IWC/Japan joint cetacean sighting survey in 2010. IWC Scientific Committee, Tromso, Norway.

Kanda, N., M. Goto, and L. A. Pastene. 2006. Genetic characteristics of western North Pacific sei whales, Balaenoptera borealis, as revealed by microsatellites. Marine Biotechnology 8(1):86-93.

Kanda, N., K. Matsuoka, M. Goto, and L. A. Pastene. 2015. Genetic study on JARPNII and IWC-POWER samples of sei whales collected widely from the North Pacific at the same time of the year. IWC Scientific Committee, San Diego, California.

Kanda, N., K. Matsuoka, H. Yoshida, and L. A. Pastene. 2013. Microsatellite DNA analysis of sei whales obtained from the 2010-2012 IWC-POWER. IWC Scientific Committee, Jeju, Korea.

Kane, A. S., and coauthors. 2010. Exposure of fish to high-intensity sonar does not induce acute pathology. Journal of Fish Biology 76(7):1825-1840.

Karpovich, S. A., J. P. Skinner, J. E. Mondragon, and G. M. Blundell. 2015. Combined physiological and behavioral observations to assess the influence of vessel encounters on harbor seals in glacial fjords of southeast Alaska. Journal of Experimental Marine Biology and Ecology 473:110–120.

Kastak, D., and coauthors. 2007. Onset, growth, and recovery of in-air temporary threshold shift in a California sea lion (Zalophus californianus). Journal of the Acoustical Society of America 122(5):2916-2924.

Kastak, D., B. L. Southall, R. J. Schusterman, and C. R. Kastak. 2005. Underwater temporary threshold shift in pinnipeds: Effects of noise level and duration. Journal of the Acoustical Society of America 118(5):3154-3163.

Kastelein, R. A., I. V. D. Belt, R. Gransier, and T. Johansson. 2015a. Behavioral responses of a harbor porpoise (Phocoena phocoena) to 255- to 245-kHz sonar down-sweeps with and without side bands. Aquatic Mammals 41(4):400-411.

Kastelein, R. A., I. V. D. Belt, L. Helder-Hoek, R. Gransier, and T. Johansson. 2015b. Behavioral responses of a harbor porpoise (Phocoena phocoena) to 25-kHz FM sonar signals. Aquatic Mammals 41(3):311-326.

Kastelein, R. A., S. A. Cornelisse, L. A. E. Huijser, and L. Helder-Hoek. 2020. Temporary hearing threshold shift in harbor porpoises (*Phocoena phocoena*) due to one-sixth octave noise band at 63 kHz. Aquatic Mammals 46(2):167-182.

Kastelein, R. A., R. Gransier, and L. Hoek. 2013a. Comparative temporary threshold shifts in a harbor porpoise and harbor seal, and severe shift in a seal. Journal of the Acoustical Society of America 134(1):13-16.

Kastelein, R. A., R. Gransier, L. Hoek, A. Macleod, and J. M. Terhune. 2012a. Hearing threshold shifts and recovery in harbor seals (Phoca vitulina) after octave-band noise exposure at 4 kHz. Journal of the Acoustical Society of America 132(4):2745-2761.

Kastelein, R. A., R. Gransier, L. Hoek, and J. Olthuis. 2012b. Temporary threshold shifts and recovery in a harbor porpoise (Phocoena phocoena) after octave-band noise at 4 kHz. Journal of the Acoustical Society of America 132:3525-3537.

Kastelein, R. A., R. Gransier, J. Schop, and L. Hoek. 2015c. Effects of exposure to intermittent and continuous 6-7 kHz sonar sweeps on harbor porpoise (Phocoena phocoena) hearing. Journal of the Acoustical Society of America 137(4):1623-1633.

Kastelein, R. A., D. D. Haan, N. Vaughan, C. Staal, and N. M. Schooneman. 2001. The influence of three acoustic alarms on the behaviour of harbour porpoises (Phocoena phocoena) in a floating pen. Marine Environmental Research 52(4):351-371.

Kastelein, R. A., L. Helder-Hoek, S. Cornelisse, L. A. E. Huijser, and R. Gransier. 2019a. Temporary hearing threshold shift in harbor porpoises (*Phocoena phocoena*) due to one-sixth octave noise band at 32 kHz. Aquatic Mammals 45(5):549-562.

Kastelein, R. A., L. Helder-Hoek, and R. Gransier. 2019b. Frequency of greatest temporary hearing threshold shift in harbor seals (Phoca vitulina) depends on fatiguing sound level. The Journal of the Acoustical Society of America 145(3):1353.

Kastelein, R. A., and coauthors. 2015d. Hearing thresholds of harbor seals (*Phoca vitulina*) for playbacks of seal scarer signals, and effects of the signals on behavior. Hydrobiologia 756(1):75–88.

Kastelein, R. A., and coauthors. 2014a. Hearing thresholds of harbor seals (Phoca vitulina) for playbacks of seal scarer signals, and effects of the signals on behavior. Hydrobiologia.

Kastelein, R. A., L. Helder-Hoek, G. Janssens, R. Gransier, and T. Johansson. 2015e. Behavioral responses of harbor seals (*Phoca vitulina*) to sonar signals in the 25-kHz range. Aquatic Mammals 41(4):388–399.

Kastelein, R. A., L. Helder-Hoek, R. van Kester, R. Huisman, and R. Gransier. 2019c. Temporary hearing threshold shift in harbor porpoises (*Phocoena phocoena*) due to one-sixth octave noise band at 16 kHz. Aquatic Mammals 45(3):280-292.

Kastelein, R. A., L. Hoek, R. Gransier, M. Rambags, and N. Claeys. 2014b. Effect of level, duration, and inter-pulse interval of 1-2kHz sonar signal exposures on harbor porpoise hearing. Journal of the Acoustical Society of America 136(1):412-422.

Kastelein, R. A., J. Huybrechts, J. Covi, and L. Helder-Hoek. 2017. Behavioral Responses of a Harbor Porpoise (Phocoena phocoena) to Sounds from an Acoustic Porpoise Deterrent. Aquatic Mammals 43(3):233-244.

Kastelein, R. A., N. Jennings, W. C. Verboom, D. D. Haan, and N. M. Schooneman. 2006. Differences in the response of a striped dolphin (Stenella coeruleoalba) and a harbour porpoise (Phocoena phocoena) to an acoustic alarm. Marine Environmental Research 61(3):363-378.

Kastelein, R. A., J. Schop, R. Gransier, and L. Hoek. 2014c. Frequency of greatest temporary hearing threshold shift in harbor porpoises (Phocoena phocoena) depends on the noise level. Journal of the Acoustical Society of America 136(3):1410-1418.

Kastelein, R. A., J. Schop, R. Gransier, N. Steen, and N. Jennings. 2014d. Effect of series of 1 to 2 kHz and 6 to 7 kHz up-sweeps and down-sweeps on the behavior of a harbor porpoise (Phocoena phocoena). Aquatic Mammals 40(3):232-242.

Kastelein, R. A., D. van Heerden, R. Gransier, and L. Hoek. 2013b. Behavioral responses of a harbor porpoise (*Phoceoena phocoena*) to playbakcs of broadband pile driving sounds. Marine Environmental Research 92:206–214.

Kastelein, R. A., W. C. Verboom, M. Muijsers, N. V. Jennings, and S. Van Der Heul. 2005. The influence of acoustic emissions for underwater data transmission on the behavior of harbour poroises (Phocoena phocoena) in a floating pen. Marine Enviornmental Research 59:287-307.

Kasuya, T., and T. Miyashita. 1988. Distribution of sperm whale stocks in the North Pacific. Scientific Reports of the Whales Research Institute, Tokyo 39:31-75.

Kato, H., and T. Kasuya. 2002. Some analyses on the modern whaling catch history of the western North Pacific stock of gray whales (Eschrichtius robustus), with special reference to the Ulsan whaling ground. Journal of Cetacean Research and Management 4(3):277-282.

Kato, H., T. Miyashita, and H. Shimada. 1995. Segregation of the two sub-species of the blue whale in the Southern Hemisphere. (Balanenoptera musculus). Report of the International Whaling Commission 45:273-283.-Sc/46/Sh10).

Kawamura, A. 1974. Food and feeding ecology of the southern sei whale. Scientific Reports of the Whales Research Institute, Tokyo 26:25-144.

Keen, E. M., J. Wray, J. F. Pilkington, K. I. Thompson, and C. R. Picard. 2018. Distinct habitat use strategies of sympatric rorqual whales within a fjord system. Marine Environmental Research 140(1):180–189.

Keevin, T. M., and G. L. Hempen. 1997. The Environmental Effects of Underwater Explosions with Methods to Mitigate Impacts. U.S. Army Corps of Engineers, St. Louis, MO.

Kelley, C., G. Carton, M. Tomlinson, and A. Gleason. 2016. Analysis of towed camera images to determine the effects of disposed mustard-filled bombs on the deep water benthic community off south Oahu. Deep Sea Research Part II: Topical Studies in Oceanography. .

Kenney, R. D., and H. E. Winn. 1987. Cetacean biomass densities near submarine canyons compared to adjacent shelf/slope areas. Continental Shelf Research 7(2):107-114.

Kerosky, S. M., and coauthors. 2013. Passive Acoustic Monitoring for Marine Mammals in the Northwest Training Range Complex 2011–2012. Marine Physical Laboratory Scripps Institution of Oceanography, University of California San Diego, La Jolla, CA.

Ketten, D. R. 1992. The cetacean ear: Form, frequency, and evolution. Pages 53-75 *in* J. A. Supin, editor. Marine Mammal Sensory Systems. Plenum Press, New York.

Ketten, D. R. 1995. Estimates of blast injury and acoustic trauma zones for marine mammals from underwater explosions. Pages 391-407 *in* R. A. Kastelein, J. A. Thomas, and P. E. Nachtigall, editors. Sensory Systems of Aquatic Mammals. De Spil Publishers, Woerden.

Ketten, D. R. 1997. Structure and function in whale ears. Bioacoustics-the International Journal of Animal Sound and Its Recording 8:103-135.

Ketten, D. R. 1998. Marine Mammal Auditory Systems: A Summary of Audiometroc and Anatomical Data and its Implications for Underwater Acoustic Impacts. Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-256, La Jolla, California.

Ketten, D. R. 2000. Cetacean ears. Pages 43-108 *in* W. W. L. A. A. N. P. R. R. Fay, editor. Hearing by Whales and Dolphins. Springer-Verlag, New York.

Ketten, D. R., and S. M. Bartol. 2006. Functional measures of sea turtle hearing. Office of Naval Research, Arlington, VA.

Ketten, D. R., J. Lien, and S. Todd. 1993. Blast injury in humpback whale ears: Evidence and implications. Journal of the Acoustical Society of America 94(3 Part 2):1849-1850.

Ketten, D. R., and D. C. Mountain. 2014. Inner ear frequency maps: First stage audiograms of low to infrasonic hearing in mysticetes. Pages 41 *in* Fifth International Meeting on the Effects of Sounds in the Ocean on Marine Mammals (ESOMM - 2014), Amsterdam, The Netherlands.

Kier Associates. 1991. Long range plan for the Klamath River Basin Conservatin Area Fishery Restoration Program. U.S. Fish and Wildlife Service Klamath River Fishery Resource Office. Yreka, California. [8.5 MB] at KRIS Web site:403 p.

Kiessling, I. 2003. Finding solutions: derelict fishing gear and other marine debris in northern Australia. National Oceans Office.

Kindsvater, H. K., M. Mangel, J. D. Reynolds, and N. K. Dulvy. 2016. Ten principles from evolutionary ecology essential for effective marine conservation. Ecology and Evolution 6(7):2125-2138.

Kindt-Larsen, L., C. W. Berg, S. Northridge, and F. Larsen. 2018. Harbor porpoise (Phocoena phocoena) reactions to pingers. Marine Mammal Science 35(2):552-573.

King, S. L., and coauthors. 2015. An interim framework for assessing the population consequences of disturbance. Methods in Ecology and Evolution 6(10):1150–1158.

Kintisch, E. 2006. As the seas warm: Researchers have a long way to go before they can pinpoint climate-change effects on oceangoing species. Science 313:776-779.

Kipple, B., and C. Gabriele. 2007. Underwater noise from skiffs to ships. Pages 172-175 *in* Fourth Glacier Bay Science Symposium.

Kirkwood, G. P. 1992. Background to the development of revised management procedures, Annex I. Report of the International Whaling Commission 42:236-239.

Kirschvink, J. L. 1990. Geomagnetic sensitivity in cetaceans: An update with live stranding records in the Unites States. Pages 639-649 *in* R. A. J. A. K. Thomas, editor. Sensory Abilities of Cetaceans: Laboratory and Field Evidence.

Kirschvink, J. L., A. E. Dizon, and J. A. Westphal. 1986. Evidence from strandings for geomagnetic sensitivity in cetaceans. Journal of Experimental Biology, 120, 1–24. .

Kittinger, J. N., K. S. V. Houtan, L. E. McClenachan, and A. L. Lawrence. 2013. Using historical data to assess the biogeography of population recovery. Ecography 36(8):868-872.

Kjeld, J. M. 1982. Hormones, electrolytes and other blood constituents in large whales. Unpublished paper to the IWC Scientific Committee. 4 pp. Cambridge, June (SC/34/O12).

Kjelson, M. A., P. F. Raquel, and F. W. Fisher. 1982. Life History of Fall-run Juvenile Chinook Salmon, Oncorhynchus Tshawytscha, in the Sacaramento-San Joaquin Estuary, California.

Klinck, H., and coauthors. 2015. Cetacean Studies on the Quinault Range Site in June 2012: Passive Acoustic Monitoring of Marine Mammals Using Gliders - Results from an Engineering Test. Final Report. HDR Inc., Honolulu, HI.

Klinowska, M. 1985. Cetacean live stranding dates relate to geomagnetic disturbances. Aquatic Mammals 11(3):109-119.

Klinowska, M. 1990. Geomagnetic orientation in cetaceans: Behavioural evidence. Pages 651-663 *in* J. A. T. R. A. Kastelein, editor. Sensory Abilities of Cetaceans: Laboratory and Field Evidence. Plenum Press, New York.

Knowlton, A. R., P. K. Hamilton, M. Marx, H. M. Pettis, and S. D. Kraus. 2012. Monitoring North Atlantic right whale Eubalaena glacialis entanglement rates: A 30 yr retrospective. Marine Ecology Progress Series 466:293-302.

Knowlton, A. R., F. T. Korsmeyer, J. E. Kerwin, H. Wu, and B. Hynes. 1995. The hydrodynamic effects of large vessels on right whales. Pages 62 *in* Eleventh Biennial Conference on the Biology of Marine Mammals, Orlando, Florida.

Knowlton, A. R., and S. D. Kraus. 2001. Mortality and serious injury of northern right whales (Eubalaena glacialis) in the western North Atlantic Ocean. Journal of Cetacean Research and Management Special Issue 2:193-208.

Koide, S., J. Silva, V. Dupra, and M. Edwards. 2016. Bioaccumulation of chemical warfare agents, energetic materials, and metals in deep-sea shrimp from discarded military munitions sites off Pearl Harbor. Deep Sea Research Part II: Topical Studies in Oceanography, 128, 53–62. .

Koot, B. 2015. Winter behaviour and population structure of fin whales (Balaenoptera physalus) in British Columbia inferred from passive acoustic data. University of British Columbia.

Kooyman, G. L., D. H. Kerem, W. B. Campbell, and J. J. Wright. 1973. Pulmonary gas exchange in freely diving weddell seals Leptonychotes weddelli. Respiration Physiology 17(3):283-290.

Kooyman, G. L., and coauthors. 1972. Blood nitrogen tensions of seals during simulated deep dives. American Journal of Physiology 223(5):1016-1020.

Kooyman, G. L., and E. E. Sinnett. 1982. Pulmonary shunts in harbor seals and sea lions during simulated dives to depth. Physiological Zoology 55(1):105-111.

Koski, K. 2009. The fate of coho salmon nomads: the story of an estuarine-rearing strategy promoting resilience. Ecology and Society 14(1).

Koski, W. R., and coauthors. 2015. Evaluation of UAS for photographic re-identification of bowhead whales, Balaena mysticetus. Journal of Unmanned Vehicle Systems 3(1):22-29.

Koski, W. R., J. W. Lawson, D. H. Thomson, and W. J. Richardson. 1998. Point Mugu Sea Range marine mammal technical report. Naval Air Warfare Center, Weapons Division and Southwest Division, Naval Facilities Engineering Command.

Krahn, M. M., and coauthors. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54(12):1903-1911.

Krahn, M. M., and coauthors. 2002. Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. U.S. Deptartment of Commerce, NMFS-NWFSC-54.

Kraus, S. D., and coauthors. 2016. Recent Scientific Publications Cast Doubt on North Atlantic Right Whale Future. Frontiers in Marine Science.

Krieger, K., and B. L. Wing. 1984. Hydroacoustic surveys and identifications of humpback whale forage in Glacier Bay, Stephens Passage, and Frederick Sound, southeastern Alaska, Summer 1983. U.S. Department of Commerce, NMFS/NWC-66.

Kriete, B. 2002. Bioenergetic changes from 1986 to 2001 in the Southern Resident killer whale populations, *Orcinus orca*. Orca Relief Citizens' Alliance, Friday Harbor, Washington.

Kriete, B. 2007. Orca Relief citizens' Alliance Recommendations: Protective Regulations for Killer Whales in the Northwest Region under the Endangered Species Act and Marine Mammal Protection Act.

Kruse, S. 1991. The interactions between killer whales and boats in Johnstone Strait, B.C. K. Pryor, and K. Norris, editors. Dolphin Societies: Discoveries and Puzzles. University of California Press.

Kryter, K. D., W. D. Ward, J. D. Miller, and D. H. Eldredge. 1965. Hazardous exposure to intermittent and steady-state noise. Journal of the Acoustical Society of America 39(3):451-464.

Kufeld, W. G. B., and R. M. 2005. UH-60A airloads catalog. National Aeronautics and Space Administration.

Kujawa, S. G., and M. C. Liberman. 2009. Adding insult to injury: Cochlear nerve degeneration after temporary noise-induced hearing loss. Journal of Neuroscience 29(14-Feb):14077-85.

Kuningas, S., P. H. Kvadsheim, F. P. A. Lam, and P. J. O. Miller. 2013. Killer whale presence in relation to naval sonar activity and prey abundance in northern Norway. ICES Journal of Marine Science 70(7):1287-1293.

Kvadsheim, P. 2012. Estimated tissue and blood N2 levels and risk of decompression sickness in deep-, intermediate-, and shallow-diving toothed whales during exposure to naval sonar. Frontiers in Physiology 3.

Kvadsheim, P. H., and coauthors. 2017. Avoidance responses of minke whales to 1-4kHz naval sonar. Mar Pollut Bull.

Kvadsheim, P. H., and E. M. Sevaldsen. 2005. The potential impact of 1-8 kHz active sonar on stocks of juvenile fish during sonar exercises. Forsvarets Forskningsinstitutt.

Kvadsheim, P. H., E. M. Sevaldsen, L. P. Folkow, and A. S. Blix. 2010a. Behavioural and physiological responses of hooded seals (Cystophora cristata) to 1 to 7 kHz sonar signals. Aquatic Mammals 36(3):239-247.

Kvadsheim, P. H., E. M. Sevaldsen, D. Scheie, L. P. Folkow, and A. S. Blix. 2010b. Effects of Naval Sonar on Seals. Norwegian Defense Research Establishment, Kjeller, Norway.

Kyhn, L. A., and coauthors. 2015. Pingers cause temporary habitat displacement in the harbour porpoise Phocoena phocoena. Marine Ecology Progress Series 526:253-265.

Lachmuth, C. L., L. G. Barrett-Lennard, D. Q. Steyn, and W. K. Milsom. 2011. Estimation of southern resident killer whale exposure to exhaust emissions from whale-watching vessels and potential adverse health effects and toxicity thresholds. Marine Pollution Bulletin 62:792-805.

Lacy, R. C. 1997. Importance of Genetic Variation to the Viability of Mammalian Populations. Journal of Mammalogy 78(2):320-335.

Lafortuna, C. L., M. Jahoda, A. Azzellino, F. Saibene, and A. Colombini. 2003. Locomotor behaviours and respiratory pattern of the Mediterranean fin whale (Balaenoptera physalus). European Journal of Applied Physiology 303(3-4):387-395.

Lafortuna, C. L., and coauthors. 1999. Locomotor behaviour and respiratory patterns in Mediterranean fin whales (Balaenoptera physalus) tracked in their summer feeding ground. Pages 156-160 *in* P. G. H. Evan, and E. C. M. Parsons, editors. Proceedings of the Twelfth Annual Conference of the European Cetacean Society, Monaco.

Lagerquist, B. A., K. M. Stafford, and B. R. Mate. 2000. Dive characteristics of satellite-monitored blue whales (*Balaenoptera musculus*) off the Central California coast. Marine Mammal Science 16(2):375-391.

Laggner, D. 2009. Blue whale (*Baleanoptera musculus*) ship strike threat assessment in the Santa Barbara Channel, California. Master's. Evergreen State College.

Laidig, T. E., J. R. Chess, and D. F. Howard. 2007. Relationship between abundance of juvenile rockfishes (Sebastes spp.) and environmental variables documented off northern California and potential mechanisms for the covariation. Fishery Bulletin 105(1):39-48.

Laist, D. W. 1997. Impacts of marine debris: Entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. Pages 99-140 *in* D. B. J. M. R. Coe, editor. Marine Debris: Sources, Impacts, and Solutions. Springer-Verlag, New York, New York.

Laist, D. W., J. M. Coe, and K. J. O'Hara. 1999. Marine debris pollution. Pages 342-366 *in* J. R. Twiss Jr., and R. R. Reeves, editors. Conservation and Management of Marine Mammals. Smithsonian Institution Press, Washington, D.C.

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. 2001. Collisions between ships and whales. Marine Mammal Science 17(1):35-75.

Lalas, C., and H. McConnell. 2016. Effects of seismic surveys on New Zealand fur seals during daylight hours: Do fur seals respond to obstacles rather than airgun noise? Marine Mammal Science 32(2):643–663.

Lambertsen, R. 1992a. Crassicaudosis: a parasitic disease threatening the health and population recovery of large baleen whales. REVUE SCIENTIFIQUE ET TECHNIQUE-OFFICE INTERNATIONAL DES EPIZOOTIES 11:1131-1131.

Lambertsen, R. H. 1983. Crassicaudiasis of the North Atlantic fin whale (Balaenoptera physalus): Prevalence, pathogenesis, transmission, and life cycle. Fifth Biennial Conference on the Biology of Marine Mammals, 27 November-1 December New England Aquarium Boston MA. p.59.

Lambertsen, R. H. 1986. Disease of the common fin whale (*Balaenopters physalus*): Crassicaudiosis of the urinary system. Journal of Mammalogy 67(2):353-366.

Lambertsen, R. H. 1992b. Crassicaudosis: A parasitic disease threatening the health and population recovery of large baleen whales. (*Balaenoptera musculus, Balaenoptera*

physalus, Megaptera novaeangliae). Revue Scientifique Et Technique Office International Des Epizooties 11(4):1131-1141.

Lambertsen, R. H., B. A. Kohn, J. P. Sundberg, and C. D. Buergelt. 1987. Genital papillomatosis in sperm whale bulls. Journal of Wildlife Diseases 23(3):361-367.

Lambourn, D. M., and coauthors. 2012. 2007-2009 Pacific Northwest Guadalupe fur seal (*Arctocephalus townsendi*) Unusual Mortality Event (UME) Summary Report, NOAA UME Committee, May 2012.

Lammers, A., A. Pack, and L. Davis. 2003. Historical evidence of whale/vessel collisions in Hawaiian waters (1975-present). Ocean Science Institute.

Lammers, M. O., and coauthors. 2017. Acoustic monitoring of coastal dolphins and their response to naval mine neutralization exercises. Royal Society Open Science 4(12):e170558.

Lance, V. A., R. M. Elsey, G. Butterstein, and P. L. Trosclair Iii. 2004. Rapid suppression of testosterone secretion after capture in male American alligators (*Alligator mississippiensis*). General and Comparative Endocrinology 135(2):217–222.

Lande, R. 1991. Applications of genetics to management and conservation of cetaceans. Report of the International Whaling Commission Special Issue 13:301-311.

Lande, R., S. Engen, and B. E. Saether. 2003. Estimating density dependence in time-series of age-structured populations. Pages 55-65 *in* R. M. Sibley, J. Hone, and T. H. Clutton-Brock, editors. Wildlife Population Growth Rates. Cambridge University Press, Cambridge, United Kingdom.

Landry, M. R., and B. M. Hickey. 1989. Coastal Oceanography of Washington and Oregon. Elsevier Science Publishing Company Inc., New York, NY.

Laney, H., and R. C. Cavanagh. 2000. Supersonic aircraft noise at and beneath the ocean surface: Estimation of risk for effects on marine mammals. United States Air Force Research Laboratory.

Lang, A. R., and coauthors. 2011. Genetic analysis of stock structure and movements of gray whales in the eastern and western North Pacific. International Whaling Commission.

Lang, A. R., and coauthors. 2004. Genetic differentiation between western and eastern gray whale populations using microsatellite markers. International Whaling Commission Scientific Committee, Sorrento, Italy.

Lang, A. R., D. W. Weller, R. G. Leduc, A. M. Burdin, and J. R. L. Brownell. 2005. Genetic assessment of the western gray whale population. Current research and future directions. Unpublished paper to the IWC Scientific Committee. 13 pp. Ulsan, Korea, June (SC/57/BRG14).

Lang, A. R., D. W. Weller, R. G. Leduc, A. M. Burdin, and J. Robert L. Brownell. 2010. Genetic differentiation between western and eastern (Eschrichtius robustus) gray whale populations using microsatellite markers. Unpublished paper to the IWC Scientific Committee, Agadir, Morocco.

Laplanche, C., O. Adam, M. Lopatka, and J. F. Motsch. 2005. Sperm whales click focussing: Towards an understanding of single sperm whale foraging strategies. Pages 56 *in* Nineteenth Annual Conference of the European Cetacean Society, La Rochelle, France.

Larson, Z. S., and M. R. Belchik. 2000. A preliminary status review of eulachon and Pacific lamprey in the Klamath River Basin. Yurok Tribal Fisheries Program, Klamath, California.

Latishev, V. M. 2007. Scientific report from factory ships "Vladivostok" and "Dalniy Vostok" in 1967. Pages 16-17 *in* Y. V. Ivashchenko, P. J. Clapham, and R. L. Brownell Jr., editors. Scientific reports of Soviet whaling expeditions in the North Pacific, 1955-1978. , volume NOAA Technical Memorandum NMFS-AFSC-175. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, Seattle, Washington.

Law, R. J., and coauthors. 2010. Levels and trends of brominated diphenyl ethers in blubber of harbor porpoises (Phocoena phocoena) from the U.K., 1992-2008. Environmental Science & Technology 44(12):4447-4451.

Law, R. J., R. L. Stringer, C. R. Allchin, and B. R. Jones. 1996. Metals and organochlorines in sperm whales (*Physetes macrocephalus*) stranded arounf the North Sea during the 1994/1995 winter. Marine Pollution Bulletin 32(1):72-77.

Lazell Jr., J. D. 1980. New England Waters: Critical Habitat for Marine Turtles. Copeia 1980(2):290-295.

LCFRB. 2010. Washington Lower Columbia Salmon Recovery and Fish & Wildlife Subbasin Plan. Lower Columbia Fish Recovery Board, Washington. May 28, 2010.

Learmonth, J. A., and coauthors. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology: an Annual Review 44:431-464.

Leatherwood, S., F. T. Awbrey, and J. A. Thomas. 1982a. Minke whale response to a transiting survey vessel. Report of the International Whaling Commission 32:795-802.

Leatherwood, S., D. K. Caldwell, and H. E. Winn. 1976. Whales, dolphins, and porpoises of the western North Atlantic: A guide to their identification. NOAA Technical Report NMFS CIRCULAR No. 396. 176p.

Leatherwood, S., R. R. Reeves, W. F. Perrin, and W. E. Evans. 1982b. Whales, dolphins and porpoises of the eastern north Pacific and adjacent Arctic waters. A guide to their Identification. National Marine Fisheries Service.

Leduc, R. G., and coauthors. 2012. Genetic analysis of right whales in the eastern North Pacific confirms severe extirpation risk. Endangered Species Research 18(2):163-167.

Leduc, R. G., and coauthors. 2002. Genetic differences between western and eastern gray whales (Eschrichtius robustus). Journal of Cetacean Research and Management 4(1):1-5.

Lemon, M., T. P. Lynch, D. H. Cato, and R. G. Harcourt. 2006. Response of travelling bottlenose dolphins (Tursiops aduncus) to experimental approaches by a powerboat in Jervis Bay, New South Wales, Australia. Biological Conservation 127(4):363-372.

Lenhardt, M. L. 1994. Seismic and very low frequency sound induced behaviors in captive loggerhead marine turtles (*Caretta caretta*). Pages 238-241 *in* K. A. Bjorndal, A. B. Bolten, D. A. Johnson, and P. J. Eliazar, editors. Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Lenhardt, M. L., S. Bellmund, R. A. Byles, S. W. Harkins, and J. A. Musick. 1983. Marine turtle reception of bone conducted sound. The Journal of auditory research 23:119-125.

Lenhardt, M. L., S. E. Moein, J. A. Musick, and D. E. Barnard. 1994. Evaluation of the Response of Loggerhead Sea Turtles (Caretta caretta) to a Fixed Sound Source. Draft Final Report Submitted to the U.S. Army Corps of Engineers, Waterways Experiment Station:13.

Lesage, V., C. Barrette, M. C. S. Kingsley, and B. Sjare. 1999. The effect of vessel noise on the vocal behavior of Belugas in the St. Lawrence River estuary, Canada. Marine Mammal Science 15(1):65-84.

Lewison, R. L., S. A. Freeman, and L. B. Crowder. 2004. Quantifying the effects of fisheries on threatened species: The impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecology Letters 7:221-231.

Lien, J. 1994. Entrapments of large cetaceans in passive inshore fishing gear in Newfoundland and Labrador (1979-1990). Report of the International Whaling Commission (Special Issue 15):149–157.

Light, J. T., C. K. Harris, and R. L. Burgner. 1989. Ocean distribution and migration of steelhead (*Oncorhynchus mykiss*, formerly *Salmo gairdneri*). International North Pacific Fisheries Commission, Fisheries Research Institute.

Lima, S. L. 1998. Stress and decision making under the risk of predation. Advances in the Study of Behavior 27:215-290.

Lin, H. W., A. C. Furman, S. G. Kujawa, and M. C. Liberman. 2011. Primary neural degeneration in the guinea pig cochlea after reversible noise-induced threshold shift. Journal of the Association for Research in Otalaryngology 12(605-616).

Lindley, S., and coauthors. 2004. Population structure of threatened and endangered Chinook salmon ESUs in California's Central Valley basin. NOAA Technical Memorandum NMFS-SWFSC 360.

Lindley, S. T., and coauthors. 2011. Electronic tagging of green sturgeon reveals population structure and movement among estuaries. Transactions of the American Fisheries Society 140(1):108-122.

Lindley, S. T., and coauthors. 2008. Marine migration of North American green sturgeon. Transactions of the American Fisheries Society 137(1):182-194.

Lindley, S. T., and coauthors. 2006. Historical population structure of Central Valley steelhead and its alterations by dams. San Francisco Estuary & Watershed Science 4(1):1-19.

Lindley, S. T., and coauthors. 2007. Framework for assessing viability of threatened and endangered Chinook salmon and steelhead in the Sacramento–San Joaquin Basin. San Francisco Estuary and Watershed Science 5(1).

Linnæus, C. 1758. Systema Naturæ per Regna Tria Naturæ, Secundum Classes, Ordines, Genera, Species, Cum Characteribus, Differentiis, Synonymis, Locis, volume Tomus I. Salvius.

Lloyd, B. D. 2003. Potential effects of mussel farming on New Zealand's marine mammals and seabirds: A discussion paper. Department of Conservation.

Lombarte, A., H. Y. Yan, A. N. Popper, J. C. Chang, and C. Platt. 1993. Damage and regeneration of hair cell ciliary bundles in a fish ear following treatment with gentamicin. Hearing Research 66:166-174.

Lopez, P., and J. Martin. 2001. Chemosensory predator recognition induces specific defensive behaviours in a fossorial amphisbaenian. Animal Behaviour 62:259-264.

Lotufo, G. R., A. B. Gibson, and J. L. Yoo. 2010. Toxicity and bioconcentration evaluation of RDX and HMX using sheepshead minnows in water exposures. Ecotoxicology and Environmental Safety 73(7):1653-1657.

Love, M. S., M. H. Carr, and L. J. Haldorson. 1991. The ecology of substrate-associated juveniles of the genus *Sebastes*. Environmental Biology of Fishes 30:225-243.

Love, M. S., J. E. Caselle, and K. Herbinson. 1998a. Declines in nearshore rockfish recruitment and populations in the southern California Bight as measured by impingement rates in coastal electrical power generating stations. Fishery Bulletin 96(3):492-501.

Love, M. S., J. E. Caselle, and W. Van Buskirk. 1998b. A severe decline in the commercial passenger fishing vessel rockfish (*Sebastes* spp.) catch in the Southern California Bight, 1980-1996. California Cooperative Oceanic Fisheries Investigations Reports 39:180-195.

Love, M. S., M. M. Yoklavich, and L. Thorsteinson. 2002a. The rockfishes of the Northeast Pacific, Berkeley, California.

Love, M. S., D. M. Schroeder, and W. Lenarz. 2005. Distribution of bocaccio (Sebastes paucispinis) and cow cod (Sebastes levis) around oil platforms and natural outcrops off California with implications for larval production. Bulletin of Marine Science 77:397-408.

Love, M. S., M. M. Yoklavich, and L. Thorsteinson. 2002b. The Rockfishes of the Northeast Pacific. University of California Press, Berkeley, California.

Luksenburg, J. A., and E. C. Parsons. 2009. The effects of aircraft on cetaceans: Implications for aerial whalewatching.

Lundin, J. I., and coauthors. 2016. Persistent Organic Pollutant Determination in Killer Whale Scat Samples: Optimization of a Gas Chromatography/Mass Spectrometry Method and Application to Field Samples. Archives of Environmental Contamination and Toxicology 70(1):9-19.

Lundquist, D., N. J. Gemmell, and B. Wursig. 2012. Behavioural responses of dusky dolphin groups (*Lagenorhynchus obscurus*) to tour vessels off Kaikoura, New Zealand. PLoS ONE 7(7):e41969.

Lusseau, D. 2003. Effects of tour boats on the behavior of bottlenose dolphins: Using Markov chains to model anthropogenic impacts. Conservation Biology 17(6):1785-1793.

Lusseau, D. 2004. The hidden cost of tourism: Detecting long-term effects of tourism using behavioral information. Ecology and Society 9(1):2.

Lusseau, D. 2006. The short-term behavioral reactions of bottlenose dolphins to interactions with boats in Doubtful Sound, New Zealand. Marine Mammal Science 22(4):802-818.

Lusseau, D., D. E. Bain, R. Williams, and J. C. Smith. 2009. Vessel traffic disrupts the foraging behavior of southern resident killer whales *Orcinus orca*. Endangered Species Research 6:211-221.

Lutcavage, M. E., P. Plotkin, B. E. Witherington, and P. L. Lutz. 1997. Human impacts on sea turtle survival. Pages 387-409 *in* P. L. Lutz, and J. A. Musick, editors. The biology of sea turtles. CRC Press, Boca Raton, Florida.

Lyamin, O. I., S. M. Korneva, V. V. Rozhnov, and L. M. Mukhametov. 2011. Cardiorespiratory changes in beluga in response to acoustic noise. Doklady Biological Sciences 440(1):275-278.

Lyrholm, T., and U. Gyllensten. 1998. Global matrilineal population structure in sperm whales as indicated by mitochondrial DNA sequences. Proceedings of the Royal Society B-Biological Sciences 265(1406):1679-1684.

MacCall, A. D. 1996. Patterns of low frequency variability in fish populations of the California Current. California Cooperative Oceanic Fisheries Investigation Report.

MacCall, A. D. 2002. Use of known-biomass production models to determine productivity of west coast groundfish stocks. North American Journal of Fisheries Management 22:272-279.

MacCall, A. D. 2003. Status of bocaccio off California in 2003. Pacific Fishery Management Council, Portland, Oregon.

MacCall, A. D., and X. He. 2002. Status review of the southern stock of bocaccio (*Sebastes paucispinis*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz Laboratory, Santa Cruz, California.

MacFadyen, A., B. M. Hickey, and W. P. Cochlan. 2008. Influences of the Juan de Fuca Eddy on circulation, nutrients, and phytoplankton production in the northern California Current System. Journal of Geophysical Research 113(C08008):1–19.

Macfadyen, G., T. Huntington, and R. Cappell. 2009. Abandoned, lost or otherwise discarded fishing gear. Food and Agriculture Organization of the United Nations (FAO).

MacFarlane, R. B., and E. C. Norton. 2002. Physiological ecology of juvenile chinook salmon (Oncorhynchus tshawytscha) at the southern end of their distribution, the San Francisco Estuary and Gulf of the Farallones, California. Fishery Bulletin 100(2):244-257.

MacGillivray, A. O., Z. Li, D. E. Hannay, K. B. Trounce, and O. M. Robinson. 2019. Slowing deep-sea commerical vessels reduces underwater radiated noise. The Journal of the Acoustical Society of America 146(1):340-351.

Mackintosh, N. A. 1965. Blue and fin whales. Pages 174-182 *in* The Stocks of Whales. Fishing News.

MacLeod, C. D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. Endangered Species Research 7(2):125-136.

MacLeod, C. D., and coauthors. 2005. Climate change and the cetacean community of north-west Scotland. Biological Conservation 124(4):477-483.

Madsen, P. T., and coauthors. 2003. Sound production in neonate sperm whales. Journal of the Acoustical Society of America 113(6):2988–2991.

Madsen, P. T., and coauthors. 2006. Quantitative measures of air-gun pulses recorded on sperm whales (*Physeter macrocephalus*) using acoustic tags during controlled exposure experiments. The Journal of Acoustical Society of America 120(4):2366–2379.

Madson, P. L., B. K. van der Leeuw, K. M. Gibbons, and T. H. Van Hevelingen. 2017. Evaluation of Pinniped Predation on Adult Salmonids and Other Fish in the Bonneville Dam Tailrace, 2016. U.S. Army Corps of Engineers, Cascade Locks, Oregon.

Magalhaes, S., and coauthors. 2002. Short-term reactions of sperm whales (*Physeter macrocephalus*) to whale-watching vessels in the Azores. Aquatic Mammals 28(3):267-274.

Malme, C. I., B. Würsig, J. E. Bird, and P. Tyack. 1986. Behavioral responses of gray whales to industrial noise: Feeding observations and predictive modelling. Bolt Beranek, & Newman, Inc, Anchorage, AK.

Malme, C. I., B. Würsig, J. E. Bird, and P. Tyack. 1988. Observations of feeding gray whale responses to controlled industrial noise exposure. Pages 55–73 *in* W. M. Sackinger, M. O. Jeffries, J. L. Imm, and S. D. Tracey, editors. Port and Ocean Engineering Under Artic Conditions, volume 2. Geophysical Institute, University of Alaska, Fairbanks, AK.

Maloni, M., J. A. Paul, and D. M. Gligor. 2013. Slow steaming impacts on ocean carriers and shippers. Maritime Economics & Logistics 15(2):151–171.

Manci, K. M., D. N. Gladwin, R. Villella, and M. G. Cavendish. 1988. Effects of aircraft noise and sonic booms on domestic animals and wildlife: A literature synthesis. U.S. Fish and Wildlife Service, National Ecology Research Center, Ft. Collins, Colorado.

Mantua, N. J., and S. R. Hare. 2002. The Pacific decadal oscillation. Journal of Oceanography 58(1):35-44.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78(6):1069-1079.

Manzano-Roth, R., E. E. Henderson, S. W. Martin, C. Martin, and B. Matsuyama. 2016. Impacts of U.S. Navy Training Events on Blainville's Beaked Whale (Mesoplodon densirostris) Foraging Dives in Hawaiian Waters. Aquatic Mammals, 42(4), 507–518. .

Marcoux, M., H. Whitehead, and L. Rendell. 2006. Coda vocalizations recorded in breeding areas are almost entirely produced by mature female sperm whales (Physeter macrocephalus). Canadian Journal of Zoology 84(4):609-614.

Márquez, M. R. 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date.

Marsili, L., and S. Focardi. 1996. Organochlorine levels in subcutaneous blubber biopsies of fin whales (*Balaenoptera physalus*) and striped dolphins (*Stenella coeruleoalba*) from the Mediterranean Sea. Environmental Pollution 91(1):1-9.

Martin, S. L., and coauthors. 2020. Assessing the population level impacts of North Pacific Loggerhead and western Pacific Leatherback interactions in the Hawaii-based shallow set longline fishery.

Martin, S. W., C. R. Martin, B. M. Matsuyama, and E. E. Henderson. 2015. Minke whales (Balaenoptera acutorostrata) respond to navy training. Journal of the Acoustical Society of America 137(5):2533-2541.

Martineau, D. 2007. Potential synergism between stress and contaminants in free-ranging cetaceans. International Journal of Comparative Psychology (20):194-216.

Masaki, Y. 1976. Biological studies on the North Pacific sei whale. Bulletin of Far Seas Fishery Research 14:1-104.

Masaki, Y. 1977. The separation of the stock units of sei whales in the North Pacific. Report of the International Whaling Commission (Special Issue 1):71-79.

Maser, C., B. R. Mate, J. F. Franklin, and C. T. Dyrness. 1981. Natural history of Oregon coast mammals. U.S. Department of Agriculture, Forest Service, PNW-133, Portland, OR.

Matarese, A. C., D. M. Blood, and W. C. Rugen. 2011. A taxonomic guide and atlas for the early life history stages of northeast pacific fishes. U.S. Department of Commerce, NOAA.

Matarese, A. C., A. W. Kendall, D. M. Blood, and B. M. Vinter. 1989. Laboratory guide to early life history stages of northeast Pacific fishes. NOAA.

Mate, B., A. Bradford, G. Tsidulko, V. Vertyankin, and V. Ilyashenko. 2011. Late-feeding season movements of a western North Pacific gray whale off Sakhalin Island, Russia and subsequent migration into the eastern North Pacific. International Whaling Commission-Scientific Committee, Tromso, Norway.

Mate, B., and coauthors. 2015. Final Report Baleen (Blue & Fin) Whale Tagging in Southern California in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas (SOCAL, NWTRC, GOA) Final Report. Prepared for Commander, U.S. Pacific Fleet. Submitted to Naval Facilities Engineering Command Pacific, Pearl Harbor, Hawaii under Contract No. N62470-10-D-3011, Task Orders JP03 and KB27 issued to HDR, Inc., San Diego, California. 31 July 2015.

Mate, B. R., B. A. Lagerquist, and J. Calambokidis. 1998. The movements of North Pacific blue whales off southern California and their southern fall migration. The World Marine

Mammal Science Conference, 20-24 January Monaco. p.87-88. (=Twelth Biennial Conference on the Biology of Marine Mammals).

Mate, B. R., and coauthors. 2017. Baleen Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas Covering the Years 2014, 2015, and 2016. Final Report. Naval Facilities Engineering Command, Pacific, Pearl Harbor, Hawaii.

Mate, B. R., and coauthors. 2019. Humpback Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean: Preliminary Summary of Field Tagging Effort in Hawaii in March 2019. Prepared for Commander, U.S. Pacific Fleet, and Commander, Naval Sea Systems Command. Submitted to Naval Facilities Engineering Command, Southwest, under Cooperative Ecosystem Studies Unit, Department of the Navy Cooperative Agreement No. N62473-19-2-0002. Oregon State University, Newport, Oregon.

Mate, B. R., and coauthors. 2020. Humpback Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean: Preliminary Summary of Field Tagging Effort in Washington in September-October 2019. Prepared for Commander, U.S. Pacific Fleet. Submitted to Naval Facilities Engineering Command, Southwest, under Cooperative Ecosystem Studies Unit, Department of the Navy Cooperative Agreement No. N62473-19-2-0002. Oregon State University, Newport, Oregon.

Mateo, J. M. 2007. Ecological and hormonal correlates of antipredator behavior in adult Belding's ground squirrels (*Spermophilus beldingi*). Behavioral Ecology and Sociobiology 62(1):37-49.

Mather, J. 2004. Cephalopod skin displays: From concealment to communication. D. K. O. U. Griebel, editor. The Evolution of Communication Systems: A Comparitive Approach. The Vienna Series in Theoretical Biology and the Massachusettes Institute of Technology, Cambridge, Massachusetts.

Matkin, C. O., M. J. Moore, and F. M. D. Gulland. 2017. Review of Recent Research on Southern Resident Killer Whales (SRKW) to Detect Evidence of Poor Body Condition in the Population.

Matkin, C. O., E. L. Saulitis, G. M. Ellis, P. Olesiuk, and S. D. Rice. 2008. Ongoing population-level impacts on killer whales *Orcinus orca* following the 'Exxon Valdez' oil spill in Prince William Sound, Alaska. Marine Ecology Progress Series 356:269-281.

Matsuoka, K., T. Hakamada, and T. Miyashita. 2013. Research plan for a cetacean sighting surveys in the western North Pacific in 2013. IWC Scientific Committee, Jeju, Korea.

Matthews, G., and R. Waples. 1991. Status review for Snake River spring and summer Chinook salmon. Department of Commerce, National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center, Seattle, Wash. NOAA Fisheries Tech. Memo. No. NMFS-NWFSC-200.

Mauger, G. S., and coauthors. 2015. State of Knowledge: Climate Change in Puget Sound. Report prepared for the Puget Sound Partnership and the National Oceanic and Atmospheric Administration. Climate Impacts Group, University of Washington, Seattle.

May-Collado, L. J., and S. G. Quinones-Lebron. 2014. Dolphin changes in whistle structure with watercraft activity depends on their behavioral state. Journal of the Acoustical Society of America 135(4):EL193-EL198.

May-Collado, L. J., and D. Wartzok. 2008. A comparison of bottlenose dolphin whistles in the Atlantic Ocean: Factors promoting whistle variation. Journal of Mammalogy 89(5):1229-1240.

Maybaum, H. L. 1990. Effects of a 3.3 kHz sonar system on humpback whales, *Megaptera novaeangliae*, in Hawaiian waters. EOS 71:92.

Mccarthy, E., and coauthors. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (Mesoplodon densirostris) during multiship exercises with mid-frequency sonar. Marine Mammal Science 27(3):E206-E226.

Mccauley, R. D., and coauthors. 2000a. Marine seismic surveys - a study of environmental implications. Appea Journal 40:692-708.

McCauley, R. D., and coauthors. 2000b. Marine seismic surveys: Analysis and propagation of air-gun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid. Curtin University of Technology, Western Australia.

McCauley, R. D., M. N. Jenner, C. Jenner, K. A. McCabe, and J. Murdoch. 1998. The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: Preliminary results of observations about a working seismic vessel and experimental exposures. Australian Petroleum Production and Exploration Association Journal:692–706.

McClure, M., T. Cooney, and Interior Columbia Technical Recovery Team. 2005. Updated population delineation in the interior Columbia Basin. Memorandum to NMFS NW Regional Office, co-managers and other interested parties.

McCue, L. M., C. C. Fahy, J. Greenman, and K. Wilkinson. 2020. Status Review of the Guadalupe Fur Seal (*Arctocephalus townsendi*). National Marine Fisheries Service, Protected Resources Division, West Coast Region, Long Beach, California.

McCullough, D. A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. Columbia River Inter-Tribal Fish Commission, EPA 910-R-99-010, Portland, Oregon.

Mcdonald, B. I., and P. J. Ponganis. 2012. Lung collapse in the diving sea lion: Hold the nitrogen and save the oxygen. Biology Letters 8(6):1047-1049.

McDonald, M. A., J. Calambokidis, A. M. Teranishi, and J. A. Hildebrand. 2001. The acoustic calls of blue whales off California with gender data. Journal of the Acoustical Society of America 109(4):1728–1735.

McDonald, M. A., J. A. Hildebrand, and S. Mesnick. 2009. Worldwide decline in tonal frequencies of blue whale songs. Endangered Species Research 9(1):13–21.

McDonald, M. A., J. A. Hildebrand, and S. C. Webb. 1995a. Blue and fin whales observed on a seafloor array in the northeast Pacific. Journal of the Acoustical Society of America 98(2 Part 1):712–721.

McDonald, M. A., J. A. Hildebrand, and S. C. Webb. 1995b. Blue and fin whales observed on a seafloor array in the Northeast Pacific. The Journal of Acoustical Society of America 98(2):712–721.

McDonald, M. A., and coauthors. 2005. Sei whale sounds recorded in the Antarctic. Journal of the Acoustical Society of America 118(6):3941-3945.

McDonald, M. A., S. L. Mesnick, and J. A. Hildebrand. 2006. Biogeographic characterisation of blue whale song worldwide: Using song to identify populations. Journal of Cetacean Research and Management 8(1):55–65.

McElhany, P., and coauthors. 2003. Interim report on viability criteria for Willamette and Lower Columbia basin Pacific salmonids. National Marine Fisheries Service, Seattle, WA.
McElhany, P., M. Chilcote, J. Myers, and R. Beamesderfer. 2007. Viability status of Oregon salmon and steelhead populations in the Willamette and lower Columbia basins. NMFS and Oregon Department of Fish and Wildlife, Draft, Seattle, Washington.
McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. Pages 156 p *in* U.S. Department of Commerce, editor.
McEwan, D. R. 2001. Central Valley steelhead. California Department of Fish and Game, Sacramento, California.
Mckenna, M. F. 2011. Blue whale response to underwater noise from commercial ships. University of California, San Diego.
McKenna, M. F., J. Calambokidis, E. M. Oleson, D. W. Laist, and J. A. Goldbogen. 2015. Simultaneous tracking of blue whales and large ships demonstrates limited behavioral responses for avoiding collision. Endangered Species Research 27(3):219-232.
Mckenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2012. Underwater radiated noise from modern commercial ships. Journal of the Acoustical Society of America 131(2):92-103.
McKenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2013. Relationship between container ship underwater noise levels and ship design, operational and oceanographic conditions. Scientific Reports 3.
Mclennan, M. W. 1997. A simple model for water impact peak pressure and width: A technical memorandum.
McMahon, C. R., and G. C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12(7):1330-1338.
McMichael, G., and coauthors. 2008. Lower Monumental Reservoir juvenile fall Chinook salmon behavior studies, 2007. Report prepared for USACE, Walla Walla District, Walla Walla, Washington.
McSweeney, D. J., K. C. Chu, W. F. Dolphin, and L. N. Guinee. 1989. North Pacific humpback whale songs - a comparison of southeast Alaskan feeding ground songs with Hawaiian wintering ground songs. Marine Mammal Science 5(2):139-148.
Meier, S. K., and coauthors. 2007. Distribution and abundance of western gray whales off northeastern Sakhalin Island, Russia, 2001-2003. Environmental Monitoring and Assessment 134(1-3):107-136.
Meissner, A. M., and coauthors. 2015. Behavioural effects of tourism on oceanic common dolphins, Delphinus sp, in New Zealand: The effects of Markov analysis variations and current tour operator compliance with regulations. PLoS ONE 10(1):e116962.
Melbourne, B. A., and A. Hastings. 2008. Extinction risk depends strongly on factors contributing to stochasticity. Nature (454):100-103.
Melcon, M. L., and coauthors. 2012. Blue whales respond to anthropogenic noise. PLoS ONE 7(2):e32681.
Mellinger, D. K., and C. W. Clark. 2003. Blue whale (*Balaenoptera musculus*) sounds from the North Atlantic. Journal of the Acoustical Society of America 114(2):1108–1119.

Mellinger, D. K., K. M. Stafford, and C. G. Fox. 2004a. Seasonal occurrence of sperm whale (*Physeter macrocephalus*) sounds in the Gulf of Alaska, 1999-2001. Marine Mammal Science 20(1):48-62.

Mellinger, D. K., K. M. Stafford, S. E. Moore, L. Munger, and C. G. Fox. 2004b. Detection of North Pacific right whale (*Eubalaena japonica*) calls In the Gulf of Alaska. Marine Mammal Science 20(4):872-879.

Mercado III, E. 2018. The sonar model for humpback whale song revised. Frontiers in Psychology 9:20.

Mesnick, S. L., and coauthors. 2011. Sperm whale population structure in the eastern and central North Pacific inferred by the use of single-nucleotide polymorphisms, microsatellites and mitochondrial DNA. Mol Ecol Resour 11 Suppl 1:278-98.

Metcalfe, C., B. Koenig, T. Metcalfe, G. Paterson, and R. Sears. 2004. Intra- and inter-species differences in persistent organic contaminants in the blubber of blue whales and humpback whales from the Gulf of St. Lawrence, Canada. Marine Environmental Research 57:245–260.

Metro, O. 2015. 2014 Urban Growth Report: Investing in Our Communities 2015-2035.

Miksis, J. L., and coauthors. 2001. Cardiac responses to acoustic playback experiments in the captive bottlenose dolphin (Tursiops truncatus). Journal of Comparative Psychology 115(3):227-232.

Miller, P., and coauthors. 2011. The 3S experiments: Studying the behavioural effects of naval sonar on killer whales (Orcinus orca), sperm whales (Physeter macrocephalus), and long-finned pilot whales (Globicephala melas) in Norwegian waters. Scottish Oceans Institute.

Miller, P. J. O. 2006. Diversity in sound pressure levels and estimated active space of resident killer whale vocalizations. Journal of Comparative Physiology A (192):449-459.

Miller, P. J. O., and coauthors. 2014. Dose-response relationships for the onset of avoidance of sonar by free-ranging killer whales. Journal of the Acoustical Society of America 135(2):975-993.

Miller, P. J. O., N. Biassoni, A. Samuels, and P. L. Tyack. 2000. Whale songs lengthen in response to sonar. Nature 405(6789):903.

Miller, P. J. O., and coauthors. 2009. Using at-sea experiments to study the effects of airguns on the foraging behavior of sperm whales in the Gulf of Mexico. Deep Sea Research I 56(7):1168–1181.

Miller, P. J. O., M. P. Johnson, and P. L. Tyack. 2004. Sperm whale behaviour indicates the use of echolocation click buzzes 'creaks' in prey capture. Proceedings of the Royal Society of London Series B Biological Sciences 271(1554):2239-2247.

Miller, P. J. O., and coauthors. 2012. The severity of behavioral changes observed during experimental exposures of killer (Orcinus orca), long-finned pilot (Globicephala melas), and sperm (Physeter macrocephalus) whales to naval sonar. Aquatic Mammals 38(4):362-401.

Miller, P. J. O., and coauthors. 2015. First indications that northern bottlenose whales are sensitive to behavioural disturbance from anthropogenic noise. Royal Society Open Science 2:140484.

Miller, R., and E. Brannon. 1982. The origin and development of life history patterns in Pacific salmonids. Pages 296-309 *in* Proceedings of the Salmon and Trout Migratory Behavior Symposium. Edited by EL Brannon and EO Salo. School of Fisheries, University of Washington, Seattle, WA.

Mintz, J., and R. Filadelfo. 2011a. Exposure of marine mammals to broadband radiated noise. CNA Analysis & Solutions.

Mintz, J. D. 2012. Vessel Traffic in the Hawaii-Southern California and Atlantic Fleet Testing and Training Study Areas. Center for Naval Analyses, Alexandria, VA.

Mintz, J. D. 2016. Characterization of Vessel Traffic in the Vicinities of HRC, SOCAL, and the Navy Operating Areas off the U.S. East Coast. Center for Naval Analyses, Alexandria, VA.

Mintz, J. D., and R. J. Filadelfo. 2011b. Exposure of Marine Mammals to Broadband Radiated Noise, CRM D0024311.A2/Final.

Misund, O. A. 1997. Underwater acoustics in marine fisheries and fisheries research. Reviews in Fish Biology and Fisheries 7:1–34.

Mitson, R. B., and H. P. Knudsen. 2003. Causes and effects of underwater noise on fish abundance estimation. Aquatic Living Resources 16(3):255-263.

Mizroch, S. A., and D. W. Rice. 2013. Ocean nomads: Distribution and movements of sperm whales in the North Pacific shown by whaling data and Discovery marks. Marine Mammal Science 29(2):E136-E165.

Mizroch, S. A., D. W. Rice, and J. M. Breiwick. 1984. The blue whale, *Balaenoptera musculus*. Marine Fisheries Review 46(4):15-19.

Moberg, G. P. 2000. Biological response to stress: Implications for animal welfare. Pages 21-Jan *in* J. A. G. P. M. Moberg, editor. The Biology of Animal Stress. Oxford University Press, Oxford, United Kingdom.

Mobley, J. R. 2011. Aerial Survey Monitoring for Marine Mammals and Sea Turtles in the Hawaii Range Complex in Conjunction with Two Navy Training Events. SCC and USWEX February 16–March 5, 2011. Final Field Report. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, HI, 96860-3134, under Contract # N6247010D3011, CTO KB07. Submitted by HDR Inc., San Diego.

Mobley, J. R., and M. H. Deakos. 2015. Aerial Shoreline Surveys for Marine Mammals and Sea Turtles in the Hawaii Range Complex, Conducted after Navy Training Events. Koa Kai Surveys: 31 January and 5 February 2014. RIMPAC Surveys: 1 and 4–6 July 2014 (Prepared for Naval Facilities Engineering Command Pacific for Commander, U.S. Pacific Fleet under Contract No. N62470-10-D-3011, CTO KB26, issued to HDR, Inc.). Pearl Harbor, HI: HDR Inc.

Mobley, J. R., and A. Pacini. 2012. Aerial Survey Monitoring for Marine Mammals and Sea Turtles in the Hawaii Range Complex in Conjunction with a Navy Training Event, SCC February 15–25, 2010, Final Field Report. Prepared for Commander, Pacific Fleet Environmental. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, HI, 96860-3134, under Contract No. N62470-10-D-3011. Submitted by HDR Inc, Honolulu, HI, July 25, 2012.

Mobley, J. R., M. A. Smultea, C. E. Bacon, and A. Frankel. 2012. Preliminary Report: Aerial Survey Monitoring for Marine Mammals and Sea Turtles in the Hawaii Range Complex--Summary of Focal Follow Analysis for 2008–2012 SCC Events: Preliminary Report. Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, Hawaii. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, Hawaii 96860-3134, under Contract # N62470-10-D-3011, 11 June 2013, issued to HDR Inc., San Diego, California 92123. 11 June 2013.

Moein, S. E., and coauthors. 1994. Evaluation of seismic sources for repelling sea turtles from hopper dredges. U.S. Army Corps of Engineers, Waterways Experiment Station. Virginia Institute of Marine Science (VIMS), College of William and Mary, Gloucester Point, Virginia.

Mohl, B., M. Wahlberg, P. T. Madsen, A. Heerfordt, and A. Lund. 2003. The monopulsed nature of sperm whale clicks. Journal of the Acoustical Society of America 114(2):1143-1154.

Molinos, J. G., and coauthors. 2016. Climate velocity and the future global redistribution of marine biodiversity. Nature Climate Change 6:83-88.

Mongillo, T., G. Ferrara, and L. Barre. 2018. Did the rules work? An assessment on the effectiveness of federal vessel regulations for Southern Resident killer whales. Salish Sea Ecosystem Conference, Seattle, Washington.

Mongillo, T. M., and coauthors. 2016. Exposure to a mixture of toxic chemicals: Implications for the health of endangered Southern Resident killer whales.

Monnahan, C. C., T. A. Branch, and A. E. Punt. 2014a. Do ship strikes threaten the recovery of endangered eastern North Pacific blue whales? Marine Mammal Science.

Monnahan, C. C., T. A. Branch, K. M. Stafford, Y. V. Ivashchenko, and E. M. Oleson. 2014b. Estimating historical eastern North Pacific blue whale catches using spatial calling patterns. PLoS ONE 9(6):e98974.

Montgomery, S. A. 2020. Eulachon (*Thaleichthys pacificus*) marine ecology: Applying ocean ecosystem indicators from salmon to develop a multi-year model of freshwater abundance. University of Washington.

Mooney, T. A., P. E. Nachtigall, M. Breese, S. Vlachos, and W. W. L. Au. 2009a. Predicting temporary threshold shifts in a bottlenose dolphin (Tursiops truncatus): The effects of noise level and duration. Journal of the Acoustical Society of America 125(3):1816-1826.

Mooney, T. A., P. E. Nachtigall, and S. Vlachos. 2009b. Sonar-induced temporary hearing loss in dolphins. Biology Letters 5(4):565-567.

Moore, M. 2020. Green Sturgeon detection in the Southern Salish Sea 2012-2019. Pages 5 *in*. NOAA/Northwest Fisheries Science Center, Manchester Research Laboratory.

Moore, M. E., B. A. Berejikian, and E. P. Tezak. 2010. Early marine survival and behavior of steelhead smolts through Hood Canal and the Strait of Juan de Fuca. Transactions of the American Fisheries Society 139:49-61.

Moore, M. J., and coauthors. 2009. Gas bubbles in seals, dolphins, and porpoises entangled and drowned at depth in gillnets. Veterinary Pathology 46(3):536-547.

Moore, M. J., and G. A. Early. 2004. Cumulative sperm whale bone damage and the bends. Science 306(5705):2215.

Moore, M. M. 1980. Factors influencing survival of juvenile steelhead rainbow trout (*Salmo gairdnerii gairdnerii*) in the Ventura River. Master of Science. Humboldt State University, Arcata, CA.

Moore, S. E. 2000. Variability of cetacean distribution and habitat selection in the Alaskan Arctic, autumn 1982-91. Arctic 53(4):448-460.

Moore, S. E., K. M. Stafford, D. K. Mellinger, and J. A. Hildebrand. 2006. Listening for large whales in the offshore waters of Alaska. Bioscience 56(1):49-55.

Mora, E. A., and coauthors. 2018. Estimating the Annual Spawning Run Size and Population Size of the Southern Distinct Population Segment of Green Sturgeon. Transactions of the American Fisheries Society 147(1):195-203.

Moran, J. R., S. D. Rice, and S. F. Teerlink. 2009. Humpback whale predation on Pacific herring in southern Lynn Canal: Testing a top-down hypothesis. Ted Stevens Marine Research Institute, Alaska Fisheries Science Center, National Oceanic and Atmospheric Administration/National Marine Fisheries Service, Juneau, AK.

Moreland, E. E., M. F. Cameron, R. P. Angliss, and P. L. Boveng. 2015. Evaluation of a ship–based unoccupied aircraft system (UAS) for surveys of spotted and ribbon seals in the Bering Sea pack ice. Journal of Unmanned Vehicle Systems 3(3):114–122.

Moretti, D., and coauthors. 2009. An opportunistic study of the effect of sonar on marine mammals, marine mammal monitoring on navy ranges (M3R). 2009 ONR Marine Mammal Program Review, Alexandria, Virginia.

Moretti, D., and coauthors. 2014. A risk function for behavioral disturbance of Blainville's beaked whales derived via passive acoustic monitoring. Pages 53 *in* Fifth International Meeting on the Effects of Sounds in the Ocean on Marine Mammals (ESOMM - 2014), Amsterdam, The Netherlands.

Moser, H. G. 1996. The early life stages of fishes in the California current region. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, La Jolla, California.

Moser, H. G., and coauthors. 2000. Abundance and distribution of rockfish (*Sebastes*) larvae in the Southern California Bight in relation to environmental conditions and fishery exploitation. California Cooperative Oceanic Fisheries Investigations Reports 41:132-147.

Moser, M. L., and S. T. Lindley. 2007. Use of Washington estuaries by subadult and adult green sturgeon. Environmental Biology of Fishes 79(3-4):243.

Moyle, P., P. Foley, and R. Yoshiyama. 1992a. Status of green sturgeon, Acipenser medirostris. California. Final Report submitted to National Marine Fisheries Service, Terminal Island, CA.

Moyle, P., R. Lusardi, P. Samuel, and J. Katz. 2017. State of the Salmonids: Status of California's Emblematic Fishes 2017.

Moyle, P. B. 2002. Inland fishes of California. Univ of California Press.

Moyle, P. B., P. J. Foley, and R. M. Yoshiyama. 1992b. Status of green sturgeon, *Acipenser medirostris*, in California. University of California, Davis, California.

Moyle, P. B., J. D. Kiernan, P. K. Crain, and R. M. Quiñones. 2013. Climate Change Vulnerability of Native and Alien Freshwater Fishes of California: A Systematic Assessment Approach. PLoS ONE 8(5):e63882.

MPI. 2013. Overview of ecological effects of aquaculture., Wellington, New Zealand.

Mrosovsky, N. 1972. Spectrographs of the sounds of leatherback turtles. Herpetologica 28(3): 256-258.

Mrosovsky, N. 1994. Sex ratios of sea turtles. The Journal of Experimental Zoology 270:16-27.

Mundy, P. R. 2005. The Gulf of Alaska: Biology and Oceanography. Alaska Sea Grant College Program, University of Alaska, Fairbanks.

Mundy, P. R., and R. T. Cooney. 2005. Physical and biological background. Pages 15-23 *in* P. R. Mundy, editor. The Gulf of Alaska: Biology and oceanography. Alaska Sea Grant College Program, University of Alaska, Fairbanks, Alaska.

Munger, L. M., M. O. Lammers, and W. Au. 2014. Passive Acoustic Monitoring for Cetaceans within the Marianas Islands Range Complex (MIRC). Preliminary Report. Prepared for U.S. Pacific Fleet. Submitted to Naval Facilities Engineering Command (NAVFAC)

Pacific, Honolulu, Hawaii under Contract No. N62470-10-D-3011, Task Orders KB14 and KB 17, issued to HDR Inc., Honolulu, Hawaii. Prepared by Oceanwide Science Institute, Honolulu, Hawaii and Hawaii Institute of Marine Biology, Kaneohe, Hawaii. 10 February 2014.

Munger, L. M., M. O. Lammers, J. N. Oswald, T. M. Yack, and W. Au. 2015. Passive Acoustic Monitoring of Cetaceans within the Mariana Islands Range Complex (MIRC) Using Ecological Acoustic Recorders (EARs). Final Report. Prepared for Commander, U.S. Pacific Fleet, Environmental Readiness Division, Pearl Harbor, HI. Submitted to Naval Facilities Engineering Command Pacific, EV2 Environmental Planning, Pearl Harbor, HI under Contract No. N62470-10-D-3011 Task Orders KB14 and KB22 issued to HDR Inc., Honolulu, HI. 29 September.

Munk, K. 2001. Maximum ages of groundfishes in waters off Alaska and British Columbia and consideration of age determination. Alaska Fisheries Research Bulletin 8:12-21.

Muñoz, N. J., A. P. Farrell, J. W. Heath, and B. D. Neff. 2015. Adaptive potential of a Pacific salmon challenged by climate change. Nature Climate Change 5(2):163-166.

Musick, J. A., and C. J. Limpus. 1997. Habitat Utilization and Migration in Juvenile Sea Turtles. Pages 137-163 *in* P. L. Lutz, and J. A. Musick, editors. The biology of sea turtles. CRC Press, Inc., Washington, D.C.

Muto, M. M., and coauthors. 2018a. Alaska Marine Mammal Stock Assessments, 2017. Alaska Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, NMFS-AFSC-378, Seattle, Washington.

Muto, M. M., and coauthors. 2017. Alaska Marine Mammal Stock Assessments, 2016. National Marine Mammal Laboratory, NMFS-AFSC-323, Seattle, WA.

Muto, M. M., and coauthors. 2018b. Alaska Marine Mammal Stock Assessments, 2018. Draft. National Marine Fisheries Service, Alaska Fisheries Science Center, Seattle, WA.

Muto, M. M., and coauthors. 2020. Alaska marine mammal stock assessments, 2019. U.S. Department of Commerce.

Muto, M. M., V. T. Helker, R. P. Angliss, B. A. Allen, P. L. Boveng, J. M. Breiwick, M. F. Cameron, P. J. Clapham, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, R. C. Hobbs, Y. V. Ivaschenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, R. G. Towell, P. R. Wade, J. M. Waite, and A. N. Zerbini. 2016. Alaska marine mammal stock assessments, 2015. NOAA-TM-AFSC-323.

Muto, M. M., Van T. Helker, Blair J. Delean, Robyn P. Angliss, Peter L. Boveng,, and coauthors. 2019. Draft NMFS Alaska Marine Mammal Stock Assessments 2019. Pages 215 *in* U. S. D. o. Commerce, editor, Seattle, Washington.

Myers, J., and coauthors. 2006. Historical population structure of Pacific salmonids in the Willamette River and Columbia River basins. U.S. Department of Commerce, NMFS-NWFSC-79, Seattle, Washington.

Myers, J. M., and coauthors. 1998a. Status review of chinook salmon from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC 35:443.

Myers, J. M., and coauthors. 1998b. Status review of chinook salmon from Washington, Idaho, Oregon, and California. U.S. Department of Commerce, NMFS-NWFSC-35, Seattle, Washington.

Myers, K., N. Klovach, O. Gritsenko, S. Urawa, and T. Royer. 2007. Stock-specific distributions of Asian and North American salmon in the open ocean, interannual changes, and oceanographic conditions. North Pacific Anadromous Fish Commission Bulletin 4:159-177.

Nachtigall, P. E., and A. Y. Supin. 2013. False killer whales (Pseudorca crassidens) reduce their hearing sensitivity if a loud sound is preceded by a warning. Pages 153-154 *in* Twentieth Biennial Conference on the Biology of Marine Mammals, Dunedin, New Zealand.

Nachtigall, P. E., A. Y. Supin, J. L. Pawloski, and W. W. L. Au. 2004. Temporary threshold shifts after noise exposure in the bottlenose dolphin (Tursiops truncatus) measured using evoked auditory potentials. Marine Mammal Science 20(4):672-687.

Nachtigall, P. E., A. Y. Supin, A. B. Smith, and A. F. Pacini. 2016. Expectancy and conditioned hearing levels in the bottlenose dolphin (Tursiops truncatus). Journal of Experimental Biology 219(Pt 6):844-50.

Nadeem, K., J. E. Moore, Y. Zhang, and H. Chipman. 2016. Integrating population dynamics models and distance sampling data: A spatial hierarchical state-space approach. Ecology 97(7):1735-1745.

Nakamoto, R. J., T. T. Kisanuki, and G. H. Goldsmith. 1995. Age and growth of Klamath River green sturgeon (Acipenser medirostris).

Narazaki, T., K. Sato, K. J. Abernathy, G. J. Marshall, and N. Miyazaki. 2013. Loggerhead turtles (*Caretta caretta*) use vision to forage on gelatinous prey in mid-water. PLoS ONE 8(6):e66043.

NAS. 2017. Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals. National Academies of Sciences, Engineering, and Medicine. The National Academies Press, Washington, District of Columbia.

Nasu, K. 1974. Movements of baleen whales in relation to hydrographic conditions in the northern part of the North Pacific Ocean, Bering Sea. Pages 345-361 *in* D. W. Hood, and E. J. Kelley, editors. Oceanography of the Bering Sea. University of Alaska, Fairbanks, Alaska.

National Marine Fisheries Service. 2011. Southwest Region Stranding Database Excel file containing stranding from Southwest Region. Provided to Navy, Manuscript on file, Provided to Navy, Manuscript on file.

National Marine Fisheries Service. 2016. Southern Resident Killer Whales (*Orcinus orca*) 5-Year Review: Summary and Evaluation. National Oceanic and Atmospheric Administration, West Coast Region, Seattle, WA.

National Marine Fisheries Service. 2019. Consultation on the Issuance of Eighteen ESA Section 10(a)(1)(A) Scientific Research Permits in Oregon, Washington, and Idaho affecting Salmon, Steelhead, Eulachon, Green Sturgeon, and Rockfish in the West Coast Region. National Marine Fisheries Service, Silver Spring, Maryland.

National Oceanic and Atmospheric Administration. 2001. Office of Coast Survey.

National Oceanic and Atmospheric Administration. 2010. National Marine Fisheries Service's Final Biological Opinion for the Proposed Issuance of a United States Coast Guard Permit to the St. George Reef Lighthouse Preservation Society to Maintain the St. George Reef Lighthouse as a Private Aid to Navigation and Its Effect on the Federally Threatened Eastern Distinct Population Segment of Steller Sea Lion and Designated Critical Habitat. Pages 106 *in*. National Marine Fisheries Service, Silver Spring, MD.

National Oceanic and Atmospheric Administration. 2017. 2016 West Coast Entanglement Summary. National Marine Fisheries Service, Seattle, WA.

Navy. 2003. Report on the Results of the Inquiry into Allegations of Marine Mammal Impacts Surrounding the Use of Active Sonar by USS SHOUP (DDG 86) in the Haro Strait on or about 5 May 2003.

Navy. 2010a. Annual Range Complex Exercise Report 2 August 2009 to 1 August 2010 U.S. Navy Southern California (SOCAL) Range Complex and Hawaii Range Complex (HRC)

Navy. 2010b. Northwest Training Range Complex Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS). U. S. D. o. t. Navy, editor. Naval Facilities Engineering Command, Northwest, Silverdale, WA.

Navy. 2011a. Marine Species Monitoring for the U.S. Navy's Hawaii Range Complex and the Southern California Range Complex, 2011 Annual Report. U.S. Navy Pacific Fleet.

Navy. 2011b. Marine Species Monitoring, Information on Sightings Recorded by U.S. Navy MMOs on Vessels during Sonar Test Events in the Naval Surface Warfare Center Panama City Division (NSWC PCD). Submitted to National Marine Fisheries Service, Office of Protected Resources. Department of the Navy, United States Fleet Forces Command, Norfolk, VA.

Navy. 2012. Marine Species Monitoring for the U.S. Navy's Southern California Range Complex- Annual Report 2012. U.S. Pacific Fleet, Environmental Readiness Division, U.S. Department of the Navy, Pearl Harbor, HI.

Navy. 2013a. Comprehensive Exercise and Marine Species Monitoring Report For the U.S. Navy's Atlantic Fleet Active Sonar Training (AFAST) and Virginia Capes, Cherry Point, Jacksonville, and Gulf of Mexico Range Complexes 2009–2012. Department of the Navy, United States Fleet Forces Command, Norfolk, Virginia.

Navy. 2013b. Hawaii-Southern California Training and Testing Activities Final Environmental Impact Statement/Overseas Environmental Impact Statement.

Navy. 2013c. Water Range Sustainability Environmental Program Assessment, Potomac River Test Range Complex. Dahlgren, VA.

Navy. 2014a. Marine Species Monitoring Report for the U.S. Navy's Atlantic Fleet Active Sonar Training (AFAST) and Virginia Capes, Cherry Point, Jacksonville, and Gulf of Mexico Range Complexes - Annual Report 2013. Department of the Navy, United States Fleet Forces Command, Norfolk, VA.

Navy. 2014b. Unclassified Annual Range Complex Exercise Report, 2 August 2012 to 25 November 2013, for the U.S. Navy's Atlantic Fleet Active Sonar Training (AFAST) Study Area. Prepared for and submitted to National Marine Fisheries Service, Office of Protected Resources. Prepared by the U.S. Department of the Navy in accordance with the Letter of Authorization under the MMPA and ITS authorization under the ESA dated 1 February 2012.

Navy. 2015. Unclassified 2014 Annual Atlantic Fleet Training and Testing (AFTT) Exercise and Testing Report 14 November 2013 to 13 November 2014. Prepared for and submitted to National Marine Fisheries Service, Office of Protected Resources. Prepared by the U.S. Department of the Navy in accordance with the Letter of Authorization under the MMPA and ITS authorization under the ESA dated 14 November 2013.

Navy. 2017a. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III). SSC Pacific.

Navy. 2017b. Draft Environmental Impact Statement/Overseas Environmental Impact Statement Hawaii-Southern California Training and Testing. U.S. Department of the Navy.
Navy. 2018a. Final Environmental Impact Statement/Overseas Environmental Impact Statement for Atlantic Fleet Training and Testing.
Navy. 2018b. Hawaii-Southern California Training and Testing Biological Assessment to Support Endangered Species Act Section 7 Consultation with the National Marine Fisheries Service. United States Department of the Navy; Commander, United States Pacific Fleet; Commander, Naval Sea Systems Command.
Navy. 2018c. Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing. Technical report prepared by Space and Naval Warfare Systems Center Pacific, San Diego and Naval Undersea Warfare Center, Newport.
Navy. 2018d. U.S. Navy Marine Species Density Database Phase III for the Mariana Islands Training and Testing Study Area, Pearl Harbor, HI.
Navy. 2019a. Draft Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement for Northwest Training and Testing. N. Naval Facilities Engineering Command, editor, Oak Harbor, WA.
Navy. 2019b. Final Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement for Mariana Islands Training and Testing. United States Department of the Navy.
Navy. 2019c. Mariana Islands Training and Testing Biological Assessment to Support Endangered Species Act Section 7 Consultation with the National Marine Fisheries Service: Final. United States Department of the Navy; Commander, United States Pacific Fleet; Commander, Naval Sea Systems Command.
Navy. 2019d. Programmatic Environmental Assessment for West Coast Civilian Port Defense Training.
Navy. 2019e. Request for Regulations and Letters of Authorization for the Incidental Taking of Marine Mammals Resulting from U.S. Navy Training and Testing Activities in the Northwest Training and Testing Study Area. Submitted by: Commander, United States Pacific Fleet,

Pearl Harbor-Hickam, HI and Navy System Commands, Washington Navy Yard, DC.
Navy. 2019f. U.S. Navy Marine Species Density Database Phase III for the Northwest Training and Testing Study Area: Technical Report. NAVFAC Pacific Technical Report., Pearl Harbor, HI.
Navy. 2020a. Email sent on August 4, 2020 from Rose Johnson (Navy) to Ron Salz (NMFS) with subject: RE NWTT NMFS Seasonal Mitigation Consideration
Navy. 2020b. Email sent on September 4, 2020 from Andrea Balla-Holden (Navy) to Ron Salz (NMFS) with subject RE: followup question on draft biop comment -Crescent Harbor EOD and critical habitat.
Navy. 2020c. Navy's Response to NMFS Questions Regarding Mine Countermeasure and Neutralization Testing Received on 4 August 2020 (response dated September 10, 2020).
Navy. 2020d. Navy's Response to NMFS Seasonal Mitigation Proposal for Mine Countermeasure and Neutralization Testing Received on 8 June 2020.

Navy. 2020e. Northwest Testing and Training Biological Assessment to Support ESA Section 7 Consultation with the National Marine Fisheries Service. Navy Systems Commands (Naval Sea Systems Command and Naval Air Systems Command), as Represented by Commander, Naval Sea Systems Command.

Navy. 2020f. Northwest Training and Testing Supplemental Environmental Impact Statement September 2020 Final. U. S. D. o. t. Navy, editor, Oak Harbor, Washington.

Navy. 2020g. Unprocessed Navy acoustic exposure estimates in 6dB bins. Data provided to NMFS from Andrea Balla-Holden (Navy) on February 27, 2020

Neave, F., T. Yonemori, and R. G. Bakkala. 1976. Distribution and origin of chum salmon in offshore waters of the North Pacific Ocean. International North Pacific Fisheries Commission.

Nelson, M., M. Garron, R. L. Merrick, R. M. Pace III, and T. V. N. Cole. 2007. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2001-2005. U.S. Department of Commerce, NOAA, Northeast Fisheries Science Center.

Nemoto, T. 1964. School of baleen whales in the feeding areas. Scientific Reports of the Whales Research Institute Tokyo 18:89-110.

Nemoto, T., and A. Kawamura. 1977. Characteristics of food habits and distribution of baleen whales with special reference to the abundance of North Pacific sei and Bryde's whales. Report of the International Whaling Commission (Special Issue 1):80-87.

New, L. F., and coauthors. 2014. Using short-term measures of behaviour to estimate long-term fitness of southern elephant seals. Marine Ecology Progress Series 496:99-108.

New, L. F., and coauthors. 2013a. Modelling the biological significance of behavioural change in coastal bottlenose dolphins in response to disturbance. Functional ecology 27(2):314-322.

New, L. F., D. J. Moretti, S. K. Hooker, D. P. Costa, and S. E. Simmons. 2013b. Using energetic models to investigate the survival and reproduction of beaked whales (family Ziphiidae). PLoS ONE 8(7):e68725.

Nichol, L. M., and coauthors. 2018. Distribution, movements and habitat fidelity patterns of Fin Whales (*Balaenoptera physalus*) in Canadian Pacific Waters. Canadian Science Advisory Secretariat, Fisheries and Oceans Canada, Ottawa, Canada.

Nichol, L. M., and D. M. Shackleton. 1996. Seasonal movements and foraging behaviour of northern resident killer whales (*Orcinus orca*) in relation to the inshore distribution of salmon (*Oncorhynchus* spp) in British Columbia. Canadian Journal of Zoology 74(6):9.

Nichol, L. M. a. J. K. B. F. 2011. Information relevant to the assessment of critical habitat for blue, fin, sei, and North Pacific right whales in British Columbia. DFO Canadian Science Advisory Secretariat Research Document 2011/137.

Nichols, T., T. Anderson, and A. Sirovic. 2015. Intermittent noise induces physiological stress in a coastal marine fish. PLoS ONE, 10(9), e0139157. .

Nieukirk, S. L., and coauthors. 2012. Sounds from airguns and fin whales recorded in the mid-Atlantic Ocean, 1999–2009. The Journal of Acoustical Society of America 131(2):1102–1112.

NMFS. 1991. Final recovery plan for the humpback whale (*Megaptera novaeangliae*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS. 1998. Recovery plan for the blue whale (*Balaenoptera musculus*). Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Silver Spring, Maryland.
NMFS. 2005a. Assessment of Acoustic Exposures on Marine Mammals in Conjunction with USS Shoup Active Sonar Transmissions in the Eastern Strait of Juan de Fuca and Haro Strait, Washington, 5 May 2003. National Marine Fisheries Service, Office of Protected Resources.
NMFS. 2005b. Endangered and Threatened Species; Designation of Critical Habitat for 12 Evolutionarily Significant Units of West Coast Salmon and Steelhead in Washington, Oregon, and Idaho.
NMFS. 2005c. Proposed conservation plan for southern resident killer whales (*Orcinus orca*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region, Seattle, Washington.
NMFS. 2005d. Status review update for Puget Sound steelhead. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington.
NMFS. 2006a. Designation of Critical Habitat for Southern Resident Killer Whales: Biological Report. National Marine Fisheries Service, Northwest Region.
NMFS. 2006b. Eulachon: A review of biology and an annotated bibliography. A. F. S. Center, editor.
NMFS. 2006c. Final supplement to the Shared Strategy's Puget Sound salmon recovery plan. National Marine Fisheries Service, Northwest Region, editor, Seattle.
NMFS. 2007a. Final Supplement to the Hood Canal and Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan, Portland, Oregon.
NMFS. 2007b. Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan.
NMFS. 2008a. Biological opinion for water supply, flood control operations, and channel maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation Improvement District in the Russian River watershed. U.S. Department of Commerce, F/SWR/2006/07316, Santa Rosa, California.
NMFS. 2008b. Endangered Species Act- Section 7 Formal Consulatation Biological Opinion: Effects of the 2008 Pacific Coast Salmon Plan Fisheries on the Southern Resident Killer Whale Distinct Population Segment (Orcinus orca) and their Critical Habitat. National Marine Fisheries Service, Northwest Region.
NMFS. 2008c. Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation; Consultation on the Approval of Revised Regimes under the Pacific Salmon Treaty and the Deferral of Management to Alaska of Certain Fisheries Included in those Regimes. N. Region, editor.
NMFS. 2008d. Preliminary Scientific Conclusions of the Review of the Status of 5 Species of Rockfish: Bocaccio (Sebastes paucispinis), Canary Rockfish (Sebastes pinniger), Yelloweye Rockfish (Sebastes ruberrimus), Greenstriped Rockfish (Sebastes elongatus) and Redstripe Rockfish (Sebastes proriger) in Puget Sound, Washington.
NMFS. 2008e. Recovery Plan for Southern Resident Killer Whales (*Orcinus orca*). National Marine Fisheries Service, Northwest Region, Seattle, Washington.
NMFS. 2008f. Recovery Plan for the Steller Sea Lion (Eumetopias jubatus). Revision., Silver Spring, MD.

NMFS. 2008g. Taking and importing marine mammals; taking marine mammals incidental to a U.S. Navy shock trial. Federal Register 73(143):43130-43138.
NMFS. 2009a. Biological Opinion and Conference Opinion on the long-term operations of the Central Valley Project and State Water Project. U.S. Department of Commerce, 2008/09022, Sacramento, California.
NMFS. 2009b. Middle Columbia River Steelhead Distinct Population Segment ESA Recovery Plan National Marine Fisheries Service Northwest Regional Office, May 4, 2009.
NMFS. 2010a. Final recovery plan for the sperm whale (Physeter macrocephalus). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
NMFS. 2010b. Recovery plan for the fin whale (*Balaenoptera physalus*). Pages 121 *in*. National Marine Fisheries Service, Silver Spring, Maryland.
NMFS. 2011a. 5-Year Review: Summary & Evaluation of Puget Sound Chinook Hood Canal Summer Chum Puget Sound Steelhead. National Marine Fisheries Service Northwest Region Portland, OR.
NMFS. 2011b. 5-Year Review: Summary & Evaluation of Snake River Sockeye, Snake River Spring-Summer Chinook, Snake River Fall-Run Chinook, Snake River Basin Steelhead. N. M. F. S. N. Region, editor, Portland, OR.
NMFS. 2011c. Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Test Pile Program at Naval Base Kitsap Bangor, Kitsap County, Washington (5th field HUC: 17110018, Hood Canal). W. C. Region, editor.
NMFS. 2011d. Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation: National Marine Fisheries Service (NMFS) Evaluation of the 2010-2014 Puget Sound Chinook Harvest Resource Management Plan under Limit 6 of the 4(d) Rule; Impacts of Programs Administered by the Bureau of Indian Affairs that Support Puget Sound Tribal Salmon Fisheries; Salmon Fishing Activities Authorized by the U.S. Fish and Wildlife Service in Puget Sound; NMFS' Issuance of Regulations to Give Effect to In-season Orders of the Fraser River Panel. National Marine Fisheries Service, Northwest Region.
NMFS. 2011e. Final recovery plan for the sei whale (Balaenoptera borealis). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.
NMFS. 2011f. Upper Willamette River conservation and recovery plan for chinook salmon and steelhead.
NMFS. 2012a. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat (EFH) Consultation, Department of the Navy, P-191 Breakwater, Island County. W. C. Region, editor.
NMFS. 2012b. Final Biological Report: Final Rule to Revise the Critical Habitat Designation for Leatherback Sea Turtles.
NMFS. 2012c. Final Recovery Plan for Central California Coast coho salmon Evolutionarily Significant Unit. National Marine Fisheries Service, Southwest Region, Santa Rosa, California.

NMFS. 2012d. Sei whale (Balaenoptera borealis). 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.
NMFS. 2013a. ESA recovery plan for lower Columbia River coho salmon, lower Columbia River Chinook salmon, Columbia River chum salmon, and Lower Columbia River steelhead. National Marine Fisheries Service, Northwest Region, editor, Seattle.
NMFS. 2013b. ESA Recovery Plan for Lower Columbia River Coho Salmon, Lower Columbia River Chinook Salmon, Columbia River Chum Salmon, and Lower Columbia River Steelhead, Portland, Oregon.
NMFS. 2013c. ESA Recovery Plan for the White Salmon River Watershed. June 2013.
NMFS. 2013d. Occurrence of Western Distinct Population Segment Steller Sea Lions East of 144° W. Longitude. National Oceanic and Atmospheric Administration, Silver Spring, Maryland.
NMFS. 2013e. South-Central California steelhead recovery plan.
NMFS. 2014a. Designation of Critical Habitat for the Distinct Population Segments of Yelloweye Rockfish, Canary Rockfish, and Bocaccio: Biological Report. NMFS, West Coast Region, Protected Resources Division.
NMFS. 2014b. Final recovery plan for southern Oregon/Northern California coast coho salmon (*Oncorhynchus kisutch*). National Marine Fisheries Service, editor, Arcata, California.
NMFS. 2014c. Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of California Central Valley Steelhead. California Central Valley Area Office.
NMFS. 2015a. Biological Opinion and Conference Report on Phase II Navy NWTT Activities and NMFS' MMPA Incidental Take Authorization. National Oceanic and Atmospheric Administration, National Marine Fisheries Service Silver Spring.
NMFS. 2015b. ESA Recovery Plan for Snake River Sockeye Salmon (*Oncorhynchus nerka*). June 8, 2015. NMFS West Coast Region, Protected Resources Division.
NMFS. 2015c. ESA Section 7 Biological Opinion and Conference Report on the Navy's Northwest Training and Testing (NWTT) Activities. DOC, NOAA, NMFS, Silver Spring, Maryland.
NMFS. 2015d. Proposed ESA Recovery Plan for Snake River Fall Chinook Salmon (Oncorhynchus tshawytscha). N. F. W. C. Region, editor.
NMFS. 2015e. Southern Distinct Population Segment of the North American Green Sturgeon (Acipenser medirostris); 5-year Review: Summary and Evaluation. W. C. R. National Marine Fisheries Service, editor, Long Beach, CA.
NMFS. 2015f. Sperm whale (Physeter macrocephalus) 5-year review: Summary and evaluation. National Marine Fisheries Service, Office of Protected Resources.
NMFS. 2016a. 5-year Review: Summary and Evaluation of Central Valley Spring-run Chinook Salmon Evolutionarily Significant Unit. N. O. A. A. National Marine Fisheries Service, Commerce, editor.
NMFS. 2016b. 5-Year Status Review : Summary and Evaluation of Sacramento River Winter-Run Chinook Salmon ESU.
NMFS. 2016c. 2016 5-Year Review : Summary & Evaluation of California Coastal Chinook Salmon and Northern California Steelhead.

NMFS. 2016d. 2016 5-Year Review : Summary & Evaluation of Lower Columbia River Chinook Salmon Columbia River Chum Salmon Lower Columbia River Coho Salmon Lower Columbia River Steelhead.
NMFS. 2016e. 2016 5-Year Review: Summary & Evaluation of Southern Oregon/Northern California Coast Coho Salmon National Marine Fisheries Service West Coast Region, Arcata, California.
NMFS. 2016f. 2016 5-Year Review: Summary and Evaluation of Eulachon. West Coast Region, Portland, OR.
NMFS. 2016g. Final Coastal Multispecies Recovery Plan. National Marine Fisheries Service, West Coast Region, Santa Rosa, California.
NMFS. 2016h. Interim Guidance on the Endangered Species Act Term "Harass." December 2016. Protected Resources Management. National Marine Fisheries Service Procedural Instruction 02-110-10. .
NMFS. 2016i. Main Hawaiian Islands Monk Seal Management Plan. Pacific Islands Region.
NMFS. 2016j. Proposed ESA Recovery Plan for Snake River Spring/Summer Chinook Salmon (Oncorhynchus tshawytscha) & Snake River Steelhead (Oncorhynchus mykiss). N. O. A. A. National Marine Fisheries Service, Commerce, editor, West Coast Region.
NMFS. 2016k. Southern Resident Killer Whales (*Orcinus orca*) 5-year Review: Summary and Evaluation. National Marine Fisheries Service, West Coast Region, Seattle, Washington.
NMFS. 2016l. Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing Underwater Acoustic Thresholds for Onset of Permanent and Temporary Threshold Shifts. U. S. D. o. Commerce, editor, Silver Spring, MD.
NMFS. 2016m. Yelloweye rockfish (*Sebastes ruberrimus*), canary rockfish (*Sebastes pinniger*), and bocaccio (*Sebastes paucispinis*) of the Puget Sound/Georgia Basin. Pages 131 *in* Five-Year Review: Summary and Evaluation, West Coast Region Seattle, WA.
NMFS. 2016n. Yelloweye rockfish (Sebastes ruberrimus), canary rockfish (Sebastes pinniger), and bocaccio (Sebastes paucispinis) of the Puget Sound/Georgia Basin 5-Year Review: Summary and Evaluation.
NMFS. 2017a. 2016 5-Year Review: Summary & Evaluation of Lower Columbia River Chinook Salmon, Columbia River Chum Salmon, Lower Columbia River Coho Salmon, and Lower Columbia River Steelhead, Portland, Oregon.
NMFS. 2017b. Biological and Conference Opinion on the Proposed Implementation of a Program for the Issuance of Permits for Research and Enhancement Activities on on Atlantic and Shortnose Sturgeon Pursuant to Section 10(a) of the Endangered Species Act. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.
NMFS. 2017c. Biological and Conference Opinion on the Proposed Implementation of a Program for the Issuance of Permits for Research and Enhancement Activities on Threatened and Endangered Sea Turtles Pursuant to Section 10(a) of the Endangered Species Act. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.
NMFS. 2017d. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion: Reinitiation of Section 7 Consultation Regarding the Pacific Fisheries Management Council's Groundfish Fishery Management Plan. National Marine Fisheries Service, West Coast Region.

NMFS. 2017e. Endangered Species Act Recovery Plan for the Southern Distinct Population Segment of Eulachon (Thaleichthys pacificus).
NMFS. 2017f. ESA Recovery Plan for Snake River Spring/Summer Chinook Salmon (Oncorhynchus tshawytscha) & Snake River Basin Steelhead (Oncorhynchus mykiss). NOAA NMFS West Coast Region, Portland, OR.
NMFS. 2017g. Recovery plan for the southern distinct population segment of Eulachon (*Thaleichthys pacificus*). National Marine Fisheries Service, West Coast Region, Protected Resources Division, Portland, Oregon.
NMFS. 2017h. Rockfish recovery plan Puget Sound/Georgia Basin : yelloweye rockfish (Sebastes ruberrimus) and bocaccio (Sebastes paucispinis).
NMFS. 2018a. 2017 West Coast Entanglement Summary: Overview of Entanglement Data (NMFS MMHSRP Permit #18786-01). Silver Spring, Maryland.
NMFS. 2018b. 2018 Revision to: Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (Version 2.0).
NMFS. 2018c. Endangered Species Act (ESA) Section (a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response: Impacts of the Role of the BIA Under its Authority to Assist with the Development of the 2018-2019 Puget Sound Chinook Harvest Plan, Salmon Fishing Activities Authorized by the U.S. Fish and Wildlife Service, and Fisheries Authorized by the U.S. Fraser Panel in 2018. National Marine Fisheries Service, West Coast Region.
NMFS. 2018d. Recovery Plan for the Southern Distinct Population Segment of North American Green Sturgeon.
NMFS. 2019a. 2015-2016 Large Whale Unusual Mortality Event in the Western Gulf of Alaska, United States and British Columbia (Closed). Marine Life in Distress.
NMFS. 2019b. Biological and Conference Opinion on the Proposed Implementation of a Program for the Issuance of Permits for Research and Enhancement Activities on Cetaceans in the Arctic, Atlantic, Indian, Pacific, and Southern Oceans. O. o. P. R. National Marine Fisheries Service, Endangered Species Act Interagency Cooperation Division, editor, Silver Spring, Maryland.
NMFS. 2019c. Consultation on the Issuance of Eighteen ESA Section 10(a)(1)(A) Scientific Research Permits in Oregon, Washington, and Idaho affecting Salmon, Steelhead, Eulachon, Green Sturgeon, and Rockfish in the West Coast Region National Marine Fisheries Service, West Coast Region, Long Beach, CA.
NMFS. 2019d. Consultation on the Issuance of Thirteen ESA Section 10(a)(1)(A) Scientific Research Permits in California affecting Salmon, Steelhead, and Green Sturgeon in the West Coast Region. National Marine Fisheries Service, West Coast Region, Long Beach, CA.
NMFS. 2019e. Draft Biological Report for the Proposed Revision of the Critical Habitat Designation for Southern Resident Killer Whales. NOAA, National Marine Fisheries Service, West Coast Region.
NMFS. 2019f. ESA Recovery Plan for the Puget Sound Steelhead Distinct Population Segment (Oncorhynchus mykiss). National Marine Fisheries Service, Seattle, WA.
NMFS. 2019g. ESA Section 7(A)(2) Biological Opinion: Continued Authorization of the Hawaii Pelagic Shallow-Set Longline Fishery. NMFS, Pacific Islands Region, Protected Resources Division.

NMFS. 2019h. Proposed Revision of the Critical Habitat Designation for Southern Resident Killer Whales. Draft Biological Report.
NMFS. 2020a. 2015-2020 Guadalupe Fur Seal Unusual Mortality Event in California, Oregon and Washington. Marine Life in Distress.
NMFS. 2020b. Consultation on Implementation of the Pacific Fishery Management Council Salmon Fishery

Management Plan in 2020 for Southern Resident Killer Whales and their Current and Proposed

Critical Habitat.
NMFS. 2020c. Consultation on the Issuance of Sixteen ESA Section 10(a)(1)(A) Scientific Research Permits in Oregon, Washington, Idaho and California affecting Salmon, Steelhead, Eulachon, Green Sturgeon and Rockfish in the West Coast Region. National Marine Fisheries Service, West Coast Region.
NMFS. 2020d. Endangered Species Act status review of the leatherback turtle (Dermochelys coriacea) 2020.
NMFS. 2020e. Taking and Importing Marine Mammals; Taking Marine Mammals Incidental to the U.S. Navy Training and Testing Activities in the Northwest Training and Testing (NWTT) Study Area. Pages 33914-34048 *in* U. S. D. o. Commerce, editor. National Oceanic and Atmospheric Administration.
NMFS, and USFWS. 1998a. Recovery plan for the U.S. Pacific populations of the leatherback turtle (*Dermochelys coriacea*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Washington, D.C.
NMFS, and USFWS. 1998b. Recovery Plan for the U.S. Pacific Populations of the Leatherback Turtles (*Dermochelys coriacea*). Silver Spring, Maryland.
NMFS, and USFWS. 2007a. 5-year review: Summary and evaluation, green sea turtle (*Chelonia mydas*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service.
NMFS, and USFWS. 2007b. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
NMFS, and USFWS. 2007c. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
NMFS, and USFWS. 2013a. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
NMFS, and USFWS. 2013b. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
NMFS, and USFWS. 2015. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.
NMFS, and USFWS. 2020. Endangered Species Act status review of the leatherback turtle (Dermochelys coriacea). Report to the National Marine Fisheries Service Office of Protected Resources and U.S. Fish and Wildlife Service.

NOAA. 2004. NOAA scientists sight blue whales in Alaska: critically endangered blue whales rarely seen in Alaska waters. NOAA.
NOAA. 2010. Oil and Sea Turtles: Biology, Planning, and Response. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration, Emergency Response Division, Seattle, Washington.
NOAA. 2013. Memorandum - North Central California Coast Salmonid Recovery Priority Populations. Santa Rosa, California.
NOAA. 2014a. 2014 report on the entanglement of marine species in marine debris with an emphasis on species in the United States. National Oceanic and Atmospheric Administration, Marine Debris Program, Silver Spring, Maryland.
NOAA. 2014b. Southern Resident Killer Whales: 10 Years of Research & Conservation.
NOAA. 2016. Species in the Spotlight: Priority Actions, 2016-2020: Pacific Leatherback Turtle, *Dermochelys coriacea*. National Marine Fisheries Service Silver Spring, MD.
NOAA. 2017. 2016 West Coast Entanglement Summary. National Marine Fisheries Service, Seattle, Washington.
NOAA. 2020. 2019 West Coast Whale Entanglement Summary. Pages 4 *in* N. F. W. C. Region, editor.
Noren, D. P. 2011. Estimated field metabolic rates and prey requirements of resident killer whales. Marine Mammal Science 27(1):60-77.
Noren, D. P., A. H. Johnson, D. Rehder, and A. Larson. 2009. Close approaches be vessels elicit surface active behaviors by Southern Resident killer whales. Endangered Species Research 8:179-192.
Norman, S. A., and coauthors. 2004. Cetacean strandings in Oregon and Washington between 1930 and 2002. Journal of Cetacean Research and Management 6(1):87-99.
Normandeau, E., T. Tricas, and A. Gill. 2011. Effects of EMFs from undersea power cables on elasmobranchs and other marine species. U.S. Department of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Pacific Outer Continental Shelf Region, Camarillo, California.
Norris, K. S., and G. W. Harvey. 1972. A theory for the function of the spermaceti organ of the sperm whale. Pages 393–417 *in* S. R. Galler, editor. Animal Orientation and Navigation.
Norris, T. A., and F. R. Elorriaga-Verplancken. 2019. Guadalupe Fur Seal Population Census and Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean, 2018-2019. Technical Report. Prepared for Commander, Pacific Fleet, Environmental Readiness Division. Submitted to Naval Facilities Engineering Command (NAVFAC) Southwest, Environmental Corp, San Diego, CA 9213, under Contract No. N62473-18-2-004.
Norris, T. F., J. N. Oswald, T. M. Yack, and E. L. Ferguson. 2012. An Analysis of Marine Acoustic Recording Unit (MARU) Data Collected off Jacksonville, Florida in Fall 2009 and Winter 2009–2010. Final Report. Submitted to Naval Facilities Engineering Command (NAVFAC) Atlantic, Norfolk, Virginia, under Contract No. N62470-10-D-3011, Task Order 021, issued to HDR Inc., Norfolk, Virginia. Prepared by Bio-Waves Inc., Encinitas, California. 21 November 2012. Revised January 2014.
Nowacek, D. P., F. Christiansen, L. Bejder, J. Goldbogen, and A. Friedlaender. 2016. Studying cetacean behaviour: new technological approaches and conservation applications. Animal Behaviour, 1–10. .

Nowacek, D. P., M. P. Johnson, and P. L. Tyack. 2004. North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. Proceedings of the Royal Society of London Series B Biological Sciences 271(1536):227-231.

Nowacek, D. P., L. H. Thorne, D. W. Johnston, and P. L. Tyack. 2007. Responses of cetaceans to anthropogenic noise. Mammal Review 37(2):81-115.

Nowacek, S. M., R. S. Wells, and A. R. Solow. 2001. Short-term effects of boat traffic on bottlenose dolphins, *Tursiops truncatus*, in Sarasota Bay, Florida. Marine Mammal Science 17(4):673-688.

NRC. 1990. Sea turtle mortality associated with human activities. National Academy Press, National Research Council Committee on Sea Turtle Conservation, Washington, D.C.

NRC. 1994. Low-frequency sound and marine mammals, current knowledge and research needs. (National Research Council). National Academy Press, Washington, D.C.

NRC. 2000. Marine Mammals and Low-Frequency Sound: Progress Since 1994. National Academy Press, Washington, D. C.

NRC. 2003. Ocean Noise and Marine Mammals. National Academy Press, Washington, D.C.

NRC. 2005. Marine Mammal Populations and Ocean Noise: Determining when noise causes biologically significant effects. National Research Council of the National Academies, Washington, D.C.

Nuka Research & Planning Group, LLC. 2012. Southeast Alaska Vessel Traffic Study. Nuka Research & Planning Group, LLC, Seldovia, AK.

NWFSC. 2015a. Status Review Update for Pacific Salmon and Steelhead Listed under the Endangered Species Act: Pacific Northwest. December 21, 2015.

NWFSC. 2015b. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. National Marine Fisheries Service, Northwest Fisheries Science Center:356.

O'Keeffe, D. J. 1984. Guidelines for Predicting the Effects of Underwater Explosions on Swimbladder Fish. Pages 1–28 *in*. Naval Surface Weapons Center, Dahlgren, VA.

O'Keeffe, D. J., and G. A. Young. 1984. Handbook on the Environmental Effects of Underwater Explosions. U.S. Navy, Naval Surface Weapons Center (Code R14), NSWC TR 83-240, Silver Spring, MD.

O'Neill, S. M., G. M. Ylitalo, and J. E. West. 2014. Energy content of Pacific salmon as prey of northern and southern resident killer whales. Endangered Species Research 25:265-281.

O'Shea, T. J., and A. Aguilar. 2001. Cetacea and Sirenia. Pages 427-496 *in* R. F. Shore, and B. A. Rattner, editors. Ecotoxicology of wild mammals. John Wiley & Sons, Chichester, United Kingdom.

ODFW. 2007. Oregon Coast Coho Conservation Plan for the State of Oregon.

ODFW. 2010. Lower Columbia River Conservation and Recovery Plan for Oregon Populations of Salmon and Steelhead. August 6, 2010.

ODFW. 2016. Oregon Adult Salmonid Inventory and Sampling Project. Estimated Total Population, Ocean Harvest Impact Rate, and Spawning Population of Naturally Produced Coho.

ODFW and NMFS. 2011. Upper Willamette River conservation and recovery plan for Chinook salmon and steelhead. Oregon Department of Fish and Wildlife and National Marine Fisheries Service, Northwest Region, editor.

Ohlberger, J., E. Ward, D. Schindler, and B. Lewis. 2018. Demographic changes in Chinook salmon across the Northeast Pacific Ocean. Fish and Fisheries.

Ohsumi, S., and Y. Masaki. 1975. Japanese whale marking in the North Pacific, 1963-1972. Bulletin of the Far Seas Fisheries Research Laboratory 12:171-219.

Ohsumi, S., and Y. Masaki. 1972. Report of the Scientific Committee, Annex J. Eighth memorandum on the results of Japanese stock assessment of whales in the North Pacific. Report of the International Whaling Commission 22:91-95.-Sc/23/5).

Ohsumi, S., and S. Wada. 1974. Status of whale stocks in the North Pacific, 1972. Report of the International Whaling Commission 24:114-126.

Ohsumi, S., and S. Wada. 1972. Stock assessment of blue whales in the North Pacific. Working Paper for the 24th Meeting of the International Whaling Commission. 20 pp.

Olesiuk, P. F., M. A. Bigg, and G. M. Ellis. 1990. Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. Report of the International Whaling Commission Special Issue 12:209-243.

Oleson, E. M., J. Calambokidis, J. Barlow, and J. A. Hildebrand. 2007a. Blue whale visual and acoustic encounter rates in the Southern California Bight. Marine Mammal Science 23(3):574–597.

Oleson, E. M., and coauthors. 2007b. Behavioral context of call production by eastern North Pacific blue whales. Marine Ecology Progress Series 330:269–284.

Oleson, E. M., J. Calambokidis, E. Falcone, G. Schorr, and J. A. Hildebrand. 2009. Acoustic and visual monitoring for cetaceans along the outer Washington coast. Naval Postgraduate School.

Oleson, E. M., J. Calambokidis, M. F. McKenna, and J. A. Hildebrand. 2012. Blue whale behavioral response study & field testing of the new bioacoustic probe. Office of Naval Research.

Oleson, E. M., and J. Hildebrand. 2012. Marine Mammal Demographics Off the Outer Washington Coast and Near Hawaii. U.S. Naval Postgraduate School, Monterey, CA.

Oleson, E. M., and J. A. Hildebrand. 2011. Marine mammal demographics off the outer Washington coast and near Hawaii. Naval Postgraduate School, Monterey, California.

Oleson, E. M., S. M. Wiggins, and J. A. Hildebrand. 2007c. Temporal separation of blue whale call types on a southern California feeding ground. Animal Behaviour 74(4):881–894.

Oliveira, E., T. Jefferson, B. Bartley, and A. Bostwick. 2019. Cruise Report, Marine Species Monitoring and and Lookout Effectiveness Study, Submarine Commanders Course, February 2018, Hawaii Range Complex. Commander, U.S. Pacific Fleet, Pearl Harbor, HI.

Olson, J., and R. W. Osborne. 2017. Southern Resident Killer Whale Sighting Compilation 1948–2016. The Whale Museum and Olympic Natural Resources Center, Friday Harbor, WA.

Olson, J. K. 2013. The effect of human exposure on the anti-predatory response of harbor seals (*Phoca vitulina*). Western Washington University, Bellingham, WA.

Olson, J. K., J. Wood, R. W. Osborne, L. Barrett-Lennard, and S. Larson. 2018. Sightings of southern resident killer whales in the Salish Sea 1976–2014: The importance of a long-term opportunistic dataset. Endangered Species Research 37:105–118.

Olson, J. M. 1998. Temporal and Spatial Distribution Patterns of Sightings of Southern Community and Transient Orcan in the Inland Waters of Washington and British Columbia. Western Washington University.

Omeyer, L. C. M., and coauthors. 2020. Assessing the effects of Banana Pingers as a bycatch mitigation device for harbor porpoises (*Phocoena phocoena*). Frontiers in Marine Science 7:1-10.

Omura, H. 1988. Distribution and migration of the western Pacific stock of the gray whale (Eschrichtius robustus). Scientific Reports of the Whales Research Institute 39:1-10.

Omura, H., T. Ichihara, and T. Kasuya. 1970. Osteology of pygmy blue whale with additional information on external and other characteristics. (Balaenoptera musculus brevicauda). Scientific Reports of the Whales Research Institute Tokyo 22:1-27, +5Pls.

Orca Network. 2012. A review of the sighting archives from January 2003 through July 2012.

Orr, J. W., M. A. Brown, and D. C. Baker. 2000. Guide to rockfishes (Scoraenidae) of the genera *Sebastes*, *Sebastolobus*, and *Abelosebastes* of the Northeast Pacific Ocean. Pages 56 *in*.

Osborne, R., J. Calambokidis, and E. M. Dorsey. 1988. A guide to marine mammals of greater Puget Sound. Island Publishers, Anacortes, Washington.

Osborne, R. W. 1999. A historical ecology of Salish Sea "Resident" killer whales (*Orcinus orca*): with implications for management. Dissertation. University of Victoria, British Columbia.

Pace, R., and G. Silber. 2005. Simple analyses of ship and large whale collisions: Does speed kill? Pages 1 *in*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

Pacific Fishery Management Council. 2014a. Appendix A to the Pacific Coast Salmon Fishery Management Plan as Modified by Amendment 18 to the Pacific Coast Salmon Plan: Identification and Descriptions of Essential Fish Habitat, Adverse Impacts, and Recommended Conservation Measures for Salmon.

Pacific Fishery Management Council. 2014b. Coastal Pelagic Species: Background.

Palacios, D. M., and B. R. Mate. 1996. Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galapagos Islands. Marine Mammal Science 12(4):582-587.

Palacios, D. M., and coauthors. 2020. Humpback Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean: Final Report of Tagging Efforts off the Pacific Northwest in Summer 2018. Prepared for Commander, U.S. Pacific Fleet and Commander, Naval Sea Systems Command. Submitted to Naval Facilities Engineering Command Southwest, under Cooperative Ecosystem Studies Unit, Department of the Navy Cooperative Agreement No. N62473-17-2-0001. Oregon State University, Newport, Oregon.

Palsson, W., and coauthors. 2009. The Biology and Assessment of Rockfishes in Puget Sound.

Panigada, S., and coauthors. 2006. Mediterranean fin whales at risk from fatal ship strikes. Marine Pollution Bulletin 52(10):1287-1298.

Panigada, S., M. Zanardelli, S. Canese, and M. Jahoda. 1999. Deep diving performances of Mediterranean fin whales. Thirteen Biennial Conference on the Biology of Marine Mammals, 28 November - 3 December Wailea Maui HI. p.144.

Papastavrou, V., S. C. Smith, and H. Whitehead. 1989. Diving behaviour of the sperm whale, *Physeter macrocephalus*, off the Galápagos Islands. Canadian Journal of Zoology 67(4):839-846.

Parker, S. J., H. I. McElderry, P. S. Rankin, and R. W. Hannah. 2006. Buoyancy regulation and barotrauma in two species of nearshore rockfish. Transactions of the American Fisheries Society 135:1213-1223.

Parks, S. E. 2009. Assessment of acoustic adaptations for noise compensation in marine mammals. 2009 Office of Naval Research Marine Mammal Program Review. Alexandria, VA.

Parks, S. E., C. W. Clark, and P. L. Tyack. 2007. Short- and long-term changes in right whale calling behavior: The potential effects of noise on acoustic communication. The Journal of the Acoustical Society of America 122(6):3725–3731.

Parks, S. E., M. Johnson, D. Nowacek, and P. L. Tyack. 2011. Individual right whales call louder in increased environmental noise. Biology Letters 7:33–35.

Parsons, K. M., K. C. B. III, J. K. B. Ford, and J. W. Durban. 2009. The social dynamics of southern resident killer whales and conservation implications for this endangered population. (Orcinus orca). Animal Behaviour 77(4):963-971.

Pashin, Y. V., and L. M. Bakhitova. 1979. Mutagenic and Carcinogenic Properties of Polycyclic Aromatic Hydrocarbons. Environmental Health Perspectives 30:185-189.

Patek, S. N. 2002. Squeaking with a sliding joint: Mechanics and motor control of sound production in palinurid lobsters. Journal of Experimental Biology 205:2375-2385.

Patenaude, N. J., and coauthors. 2002. Aircraft sound and disturbance to bowhead and beluga whales during spring migration in the Alaskan Beaufort Sea. Marine Mammal Science 18(2):309-335.

Pater, L. L. 1981. Gun blast far field peak overpressure contours. Department of the Navy, Naval Surface Weapons Center.

Patterson, B., and G. R. Hamilton. 1964. Repetitive 20 cycle per second biological hydroacoustic signals at Bermuda. Marine Bio-acoustics, W N Tavolga ed. Pergamon Press Oxford. p.125-145. Proceedings of a Symposium held at the Lerner Marine Laboratory Bimini Bahamas April.

Pavan, G., and coauthors. 2000. Time patterns of sperm whale codas recorded in the Mediterranean Sea 1985-1996. Journal of the Acoustical Society of America 107(6):3487-3495.

Payne, K., P. Tyack, and R. Payne. 1983. Progressive changes in the songs of humpback whales (*Megaptera novaeangliae*): A detailed analysis of two seasons in Hawaii. Pages 9-57 *in* R. Payne, editor. Communication and Behavior of Whales. Westview Press, Boulder, CO.

Payne, P. M., J. R. Nicolas, L. O'brien, and K. D. Powers. 1986. The distribution of the humpback whale, Megaptera novaeangliae, on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel, Ammodytes americanus. Fishery Bulletin 84(2):271-277.

Payne, P. M., and coauthors. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in prey abundance. Fishery Bulletin 88(4):687-696.

Payne, R., and D. Webb. 1971. Orientation by means of long range acoustic signaling in baleen whales. Annals of the New York Academy of Sciences 188(1):110–141.

Payne, R., and D. Webb. 1971. Orientation by means of long range acoustic signaling in baleen whales. Annals of the New York Academy of Sciences 188(1):110-141.

Payne, R. S., and S. McVay. 1971. Songs of humpback whales. Humpbacks emit sounds in long, predictable patterns ranging over frequencies audible to humans. Science 173(3997):585–597.

Pearcy, W. G. 1992. Ocean ecology of north Pacific salmonids, Univ. Washington Press, Seattle, 179.

Pearcy, W. G., and J. P. Fisher. 1990. Distribution and abundance of juvenile salmonids off Oregon and Washington, 1981-1985. Pages 83p *in*.

Pecl, G. T., and G. D. Jackson. 2008. The potential impacts of climate change on inshore squid: Biology, ecology and fisheries. Reviews in Fish Biology and Fisheries 18:373-385.

Perazio, C. E., and E. Mercado III. 2018. Singing humpback whales *Megaptera novaeangliae* favor specific frequency bands. Proceedings of Meetings on Acoustics, Victoria, Canada.

Perkins, J. S., and P. C. Beamish. 1979. Net entanglements of baleen whales in the inshore fishery of Newfoundland. Journal of the Fisheries Research Board of Canada 36:521-528.

Perry, S. L., D. P. DeMaster, and G. K. Silber. 1999. The great whales: History and status of six species listed as endangered under the U.S. Endangered Species Act of 1973. Marine Fisheries Review 61(1):1-74.

Petersen, J. H., and J. F. Kitchell. 2001. Climate regimes and water temperature changes in the Columbia River: bioenergetic implications for predators of juvenile salmon. Canadian Journal of Fisheries and Aquatic Science 58(9):1831-1841.

Peterson, R. S., C. L. Hubbs, R. L. Gentry, and R. L. Delong. 1968. The Guadalupe fur seal: Habitat, behavior, population size and field identification. Journal of Mammalogy 49(4):665-675.

Peterson, W. T., C. A. Morgan, J. P. Fisher, and E. Casillas. 2010. Ocean distribution and habitat associations of yearling coho (Oncorhynchus kisutch) and Chinook (O. tshawytscha) salmon in the northern California Current. Fisheries Oceanography 19(6):508-525.

Peven, C., and coauthors. 1994. Status of summer/fall Chinook salmon in the mid-Columbia region. Chelan, Douglas, and Grant County PUDs, Boise, Idaho.

PFMC. 2014. Appendix A to the Pacific Coast Salmon Fishery Management Plan. Pacific Fishery Management Council, Portland, OR.

PFMC. 2015. Preseason Report I: Stock adundance analysis and environmental assessment Part 1 for 2015 ocean salmon fishery regulations. P. F. M. Council, editor, Portland, OR.

PFMC. 2020. Pacific Fishery Management Council Salmon Fishery Management Plan Impacts to Southern Resident Killer Whales: Final Draft Risk Assessment Pacific Fishery Management Council, Ad-Hoc Southern Resident Killer Whale Workgroup.

Pierson, M. O. 1978. A study of the population dynamics and breeding behavior of the Guadalupe fur seal, (*Arctocephalus townsendi*). University of California, Santa Cruz.

Pike, G. C., and I. B. Macaskie. 1969. Marine mammals of British Columbia. Bulletin of the Fisheries Research Board of Canada 171:1-54.

Pilot, M., M. E. Dahlheim, and A. R. Hoelzel. 2010. Social cohesion among kin, gene flow without dispersal and the evolution of population genetic structure in the killer whale (Orcinus orca). Journal of Evolutionary Biology 23(1):20-31.

Pimentel, D., R. Zuniga, and D. Morrison. 2004. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics.

Pinela, A. M., and coauthors. 2009. Population genetics and social organization of the sperm whale (*Physeter macrocephalus*) in the Azores inferred by microsatellite analyses. Canadian Journal of Zoology 87(9):802-813.

Piner, K. R., J. R. Wallace, O. S. Hamel, and R. Mikus. 2006. Evaluation of ageing accuracy of bocaccio (*Sebastes paucispinis*) rockfish using bomb radiocarbon. Fisheries Research 77(2):200-206.

Piniak, W. E. D. 2012. Acoustic ecology of sea turtles: Implications for conservation. Duke University.

Pinto De Sa Alves, L. C., A. Andriolo, A. N. Zerbini, J. L. A. Pizzorno, and P. J. Clapham. 2009. Record of feeding by humpback whales (Megaptera novaeangliae) in tropical waters off Brazil. Marine Mammal Science 25(2):416-419.

Pirotta, E., K. L. Brookes, I. M. Graham, and P. M. Thompson. 2014. Variation in harbour porpoise activity in response to seismic survey noise. Biology Letters 10(5):20131090.

Pirotta, E., and coauthors. 2015. Predicting the effects of human developments on individual dolphins to understand potential long-term population consequences. Proceedings of the Royal Society of London Series B Biological Sciences 282(1818):Article 20152109.

Pirotta, E., and coauthors. 2018. A Dynamic State Model of Migratory Behavior and Physiology to Assess the Consequences of Environmental Variation and Anthropogenic Disturbance on Marine Vertebrates. The American Naturalist 191(2):59.

Pirotta, E., and coauthors. 2012. Vessel noise affects beaked whale behavior: Results of a dedicated acoustic response study. PLoS ONE 7(8):e42535.

PISC. 2016. Marine Debris: Removal and Assessment in the Northwestern Hawaiian Islands: 2015. NOAA Fisheries Pacific Islands Fisheries Science Center.

Piscitelli, M. A., and coauthors. 2010. Lung size and thoracic morphology in shallow- and deep-diving cetaceans. Journal of Morphology 271(6):654-673.

Pitcher, T. J. 1986. Functions of shoaling behaviour in teleosts. Springer.

Pitman, R. L., L. T. Ballance, S. I. Mesnick, and S. J. Chivers. 2001. Killer whale predation on sperm whales: Observations and implications. Marine Mammal Science 17(3):494-507.

Plotkin, P. T., (Ed). 1995. National Marine Fisheries Service and the U.S. Fish and Wildlife Service Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, Maryland.

Poeta, G., E. Staffieri, A. Acosta, and C. Battisti. 2017. Ecological effects of anthropogenic litter on marine mammals: A global review with a "black-list" of impacted taxa.

Polagye, B., J. Wood, C. Bassett, D. Tollit, and J. Thomson. 2011. Behavioral response of harbor porpoises to vessel noise in a tidal strait. Journal of the Acoustical Society of America 129(4):2368.

Pomeroy, P., L. O'Connor, and P. Davies. 2015. Assessing use of and reaction to unmanned aerial systems in gray and harbor seals during breeding and molt in the UK. Journal of Unmanned Vehicle Systems 3(3):102–113.

Popov, V. V., A. Y. Supin, V. V. Rozhnov, D. I. Nechaev, and E. V. Sysueva. 2014. The limits of applicability of the sound exposure level (SEL) metric to temporal threshold shifts (TTS) in beluga whales, Delphinapterus leucas. Journal of Experimental Biology 217(10):1804-1810.

Popov, V. V., and coauthors. 2013. Hearing threshold shifts and recovery after noise exposure in beluga whales, Delphinapterus leucas. Journal of Experimental Biology 216(9):1587-1596.

Popov, V. V., and coauthors. 2011. Noise-induced temporary threshold shift and recovery in Yangtze finless porpoises Neophocaena phocaenoides asiaeorientalis. Journal of the Acoustical Society of America 130(1):574-584.

Popper, A., and coauthors. 2014a. Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredicted Standards Committee S3/SC1 and registered with ANSI.

Popper, A. N. 2008. Effects of Mid- and High-Frequency Sonars on Fish. Naval Undersea Warfare Center Division, Newport, RI.

Popper, A. N., and coauthors. 2007a. The effects of high-intensity, low-frequency active sonar on rainbow trout. The Journal of the Acoustical Society of America 122(1):623–635.

Popper, A. N., and coauthors. 2007b. The effects of high-intensity, low-frequency active sonar on rainbow trout. Journal of the Acoustical Society of America 122(1):623-635.

Popper, A. N., and coauthors. 2014b. ASA S3/SC1.4 TR-2014 Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI.

Popper, A. N., and coauthors. 2014c. ASA S3/SC1.4 TR-2014 Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI. Acoustical Society of America Press and Springer Briefs in Oceanography, ISSN 2196-1212 ISSN 2196-1220 (electronic)

ISBN 978-3-319-06658-5 ISBN 978-3-319-06659-2 (eBook), New York, NY and London, United Kingdom.

Popper, M. B. H., T. Carlson, and A. N. 2013. Effects of exposure to pile-driving sounds on fish. Bioacoustics 17:305-307.

Poytress, W. R., and F. D. Carrillo. 2010. Brood-year 2007 winter Chinook juvenile production indices with comparisons to juvenile production estimates derived from adult escapement. Red Bluff: United States Fish and Wildlife Service.

Poytress, W. R., and F. D. Carrillo. 2011. Brood-year 2008 and 2009 winter Chinook juvenile production indices with comparisons to juvenile production estimates derived from adult escapement. Red Bluff: United States Fish and Wildlife Service.

Poytress, W. R., and F. D. Carrillo. 2012. Brood-year 2010 winter Chinook juvenile production indices with comparisons to juvenile production estimates derived from adult escapement. Red Bluff: United States Fish and Wildlife Service.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2009. 2008 Upper Sacramento River Green Sturgeon spawning habitat and larval migration surveys. Final Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2010. 2009 Upper Sacramento River Green Sturgeon spawning habitat and larval migration surveys. Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2013. 2012 Upper Sacramento River Green Sturgeon spawning habitat and young of the year migration surveys. Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Preston, D. 2003. Sei whale carcass provides opportunity for research. NWIFC News 29(3):4-5.

Price, C. S., and coauthors. 2017. Protected Species Marnine Aquaculture Interactions. NOAA Technical Memorandum NOS NCCOS 211.

Price, C. S., and J. A. Morris. 2013. Marine cage culture and the environment: Twenty-first century science informing a sustainable industry.

Price, E. R., and coauthors. 2004. Size, growth, and reproductive output of adult female leatherback turtles *Dermochelys coriacea*. Endangered Species Research 5:1-8.

Price, S. 2017. Rare right whale sightings in Southern California.

Pritchard, P. C. H. 1971. The leatherback or leathery turtle, Dermochelys coriacea. International Union for the Conservation of Nature, Monograph 1:39 pp.

PSAT. 2007. State of the Sound 2007. Puget Sound Action Team, Office of the Governor, State of Washington.
Puget Sound Partnership. 2018. 2018 - 2022 Puget Sound Partnership Action Agenda.
Pughiuc, D. 2010. Invasive species: Ballast water battles. Seaways.
Putman, N. F., P. Verley, C. S. Endres, and K. J. Lohmann. 2015. Magnetic navigation behavior and the oceanic ecology of young loggerhead sea turtles. Journal of Experimental Biology 218(7):1044–1050.
Putnam, N. F., and coauthors. 2013. Evidence for geomagnetic imprinting as a homing mechanism in Pacific salmon. Current Biology 23:312-316.
Pynn, L. 2013. Rare whale sighting stuns B.C. researchers. The Vancouver Sun.
Quick, N., H. Callahan, and A. J. Read. 2017. Two-component calls in short-finned pilot whales (Globicephala macrorhynchus). Marine Mammal Science.
Quinn, T. P. 2005. The Behavior and Ecology of Pacific Salmon and Trout. American Fisheries Society and University of Washington Press, Seattle, Washington.
Quinn, T. P., and K. W. Myers. 2004. Anadromy and the marine migrations of Pacific salmon and trout: Rounsefell revisited. Reviews in Fish Biology and Fisheries 14:421-442.
Raaymakers, S. 2003. The GEF/UNDP/IMO global ballast water management programme integrating science, shipping and society to save our seas. Proceedings of the Institute of Marine Engineering, Science and Technology Part B: Journal of Design and Operations (B4):2-10.
Raaymakers, S., and R. Hilliard. 2002. Harmful aquatic organisms in ships' ballast water - Ballast water risk assessment, 1726-5886, Istanbul, Turkey.
Radtke, L. 1966. Distribution of smelt, juvenile sturgeon, and starry flounder in the Sacramento-San Joaquin Delta with observations on food of sturgeon. Ecological studies of the Sacramento-San Joaquin Estuary, Part II:115-119.
Ralston, S., and J. N. Ianelli. 1998. When lengths are better than ages: The complex case of bocaccio. University of Alaska, Fairbanks, Alaska.
Rankin, S., D. Ljungblad, C. Clark, and H. Kato. 2005. Vocalisations of Antarctic blue whales, *Balaenoptera musculus intermedia*, recorded during the 2001/2002 and 2002/2003 IWC/SOWER circumpolar cruises, Area V, Antarctica. Journal of Cetacean Research and Management 7(1):13–20.
Raum-Suryan, K. L., M. J. Rehberg, G. W. Pendleton, K. W. Pitcher, and T. S. Gelatt. 2004. Development of dispersal, movement patterns, and haul-out use by pup and juvenile Steller sea lions (*Eumetopias jubatus*) in Alaska. Marine Mammal Science 20(4):823-850.
Rawson, K., and coauthors. 2009. Viability criteria for the Lake Ozette sockeye salmon evolutionarily significant unit. Department of Commerce, NMFS-NWFS-99, Seattle, Washington.
Recalde-Salas, A., C. Erbe, C. S. Kent, and M. Parsons. 2020. Non-song vocalizations of humpback whales in western Australia. Frontiers in Marine Science 7:12.
Reeves, G. H., F. H. Everest, and T. E. Nickelson. 1989. Identification of physical habitats limiting the production of coho salmon in western Oregon and Washington.
Reeves, R. R., S. Leatherwood, S. A. Karl, and E. R. Yohe. 1985. Whaling results at Akutan (1912-39) and Port Hobron (1926-37). Report of the International Whaling Commission 35:441-457.

Reeves, R. R., T. D. Smith, and E. A. Josephson. 2008. Observations of western gray whales by ship-based whalers in the 19th century. IWC Scientific Committee, Santiago, Chile.

Reeves, R. R., B. S. Stewart, P. Clapham, and J. Powell. 2002. Guide to marine mammals of the world. Knopf, New York.

Reeves, R. R., and H. Whitehead. 1997. Status of the sperm whale, *Physeter macrocephalus*, in Canada. The Canadian Field-Naturalist 111(2):15.

Reichmuth, C., J. M. Sills, J. Mulsow, and A. Ghoul. 2019. Long-term evidence of noise-induced permanent threshold shift in a harbor seal (Phoca vitulina). Journal of the Acoustic Society of America 146(4):2552.

Reijnders, P., G. Donovan, A. Aguilar, and A. Bjørge. 1999. Report of the workshop on chemical pollution and cetaceans. J. Cetacean Res. Manag. (1):1-42.

Reijnders, P. J. 1986. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature 324:456-457.

Reilly, C. A., T. W. Echeverria, and S. Ralston. 1992. Interannual variation and overlap in the diets of pelagic juvenile rockfish (genus *Sebastes*) off central California. Fishery Bulletin 90(3):505-515.

Reina, R. D., P. A. Mayor, J. R. Spotila, R. Piedra, and F. V. Paladino. 2002. Nesting ecology of the leatherback turtle, *Dermochelys coriacea*, at Parque Nacional Marino Las Baulas, Costa Rica: 1988-1989 to 1999-2000. Copeia 2002(3):653-664.

Rendell, L., S. L. Mesnick, M. L. Dalebout, J. Burtenshaw, and H. Whitehead. 2012. Can genetic differences explain vocal dialect variation in sperm whales, Physeter macrocephalus? Behav Genet 42(2):332-43.

Rendell, L., and H. Whitehead. 2004. Do sperm whales share coda vocalizations? Insights into coda usage from acoustic size measurement. Animal Behaviour 67(5):865-874.

Rice, D. W. 1977. Synopsis of biological data on the sei whale and Bryde's whale in the eastern North Pacific. Report of the International Whaling Commission (Special Issue 1):92-97.

Rice, D. W. 1984. Cetaceans. Orders and Families of Recent Mammals of the World. S. Anderson AND J. Knox Jones, Jr. (eds.). p.447-490. John Wiley AND Sons, Inc., New York.

Rice, D. W. 1989. Sperm whale, *Physeter macrocephalus* (Linnaeus, 1758). Pages 177-233 *in* S. H. Ridway, and S. R. Harrison, editors. Handbook of Marine Mammals Volume 4: River Dolphins and the Larger Toothed Whales, volume 4.

Rice, D. W., A. A. Wolman, B. R. Mate, and J. T. Harvey. 1986. A Mass Stranding of Sperm Whales in Oregon - Sex and Age Composition of the School. Marine Mammal Science 2(1):64-69.

Rice, D. W., and A. A. Wolman. 1982. Whale census in the Gulf of Alaska, June to August 1980. Report of the International Whaling Commission 32:491-497.-Sc/33/O7).

Richardson, W. J., R. A. Davids, C. R. Evans, and P. Norton. 1985a. Distribution of bowheads and industrial activity. Pages 255-306 *in* W. J. Richardson, editor. Behavior, disturbance and distribution of bowhead whales *Balaena mysticetus* in the eastern Beaufort Sea, 1980-84. Report from LGL Ecological Research Associates, Inc. for U.S. Minerals Management Service, Bryan, Texas, and Reston, Virginia.

Richardson, W. J., M. A. Fraker, B. Wursig, and R. S. Wells. 1985b. Behavior of bowhead whales Balaena mysticetus summering in the Beaufort Sea: Reactions to industrial activities. Biological Conservation 32(3):195-230.

Richardson, W. J., C. R. Greene Jr., C. I. Malme, and D. H. Thomson. 1995a. Marine Mammals and Noise. Academic Press, San Diego, California.

Richardson, W. J., C. R. G. Jr., C. I. Malme, and D. H. Thomson. 1995b. Marine Mammals and Noise. Academic Press, Inc., San Diego, California.

Richardson, W. J., G. W. Miller, and C. R. Greene. 1999. Displacement of migrating bowhead whales by sounds from seismic surveys in shallow waters of the Beaufort Sea. Journal of the Acoustical Society of America 106(4):2281.

Richardson, W. J., and B. Wursig. 1995. Significance of responses and noise impacts. Pages 387-424 *in* W. J. Richardson, C. R. Greene Jr., C. I. Malme, and D. H. Thomson, editors. Marine Mammals and Noise. Academic Press, San Diego.

Richerson, K., and coauthors. 2019 Observed and Estimated Bycatch of Green Sturgeon in 2002-2017 U.S. West Coast Groundfish Fisheries. NOAA Fisheries, NWFSC, Observer Program, 2725 Montlake Blvd E., Seattle, WA 98112.

Richmond, D. R., J. T. Yelverton, and E. R. Fletcher. 1973. Far-field underwater-blast injuries produced by small charges. Lovelace Foundation for Medical Education and Research.

Richter, C., S. Dawson, and E. Slooten. 2006. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22(1):46-63.

Richter, C. F., S. M. Dawson, and E. Slooten. 2003. Sperm whale watching off Kaikoura, New Zealand: effects of current activities on surfacing and vocalisation patterns. Department of Conservation, Wellington, New Zealand.

Rick, T. C., and coauthors. 2009. A trans-Holocene archaeological record of Guadalupe fur seals (*Arctocephalus townsendi*) on the California coast. Marine Mammal Science 25(2):487-502.

Riddell, B., and coauthors. 2018. Ocean ecology of Chinook salmon. Pages 555-696 *in*.

Ridgway, S. H. 1972. Homeostasis in the aquatic environment. Pages 590-747 *in* S. H. Ridgway, editor. Mammals of the Sea: Biology and Medicine. Charles C. Thomas, Springfield, Illinois.

Ridgway, S. H., and coauthors. 1997. Behavioral responses and temporary shift in masked hearing threshold of bottlenose dolphins, Tursiops truncatus, to 1-second tones of 141 to 201 dB re 1 µPa. U.S. Department of Navy, Naval Command, Control and Ocean Surveillance Center, RDT&E Division, San Diego, California.

Ridgway, S. H., and R. Howard. 1979. Dolphin lung collapse and intramuscular circulation during free diving: Evidence from nitrogen washout. Science 206(4423):1182-1183.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. 1969. Hearing in the giant sea turtle, Chelonoa mydas. Proceedings of the National Academies of Science 64.

Riesch, R., J. K. Ford, and F. Thomsen. 2006. Stability and group specificity of stereotyped whistles in resident killer whales, Orcinus orca, off British Columbia. Animal Behaviour 71(1):79-91.

Risch, D., P. J. Corkeron, W. T. Ellison, and S. M. V. Parijs. 2012. Changes in humpback whale song occurrence in response to an acoustic source 200 km away. PLoS ONE 7(1):e29741.

Risch, D., P. J. Corkeron, W. T. Ellison, and S. M. V. Parijs. 2014. Formal comment to Gong et al: Ecosystem scale acoustic sensing reveals humpback whale behavior synchronous with herring spawning processes and re-evaluation finds no effect of sonar on humpback song occurrence in the Gulf of Maine in fall 2006. PLoS ONE 9(10):e109225.

Ritter, F. 2012. Collisions of sailing vessels with cetaceans worldwide: First insights into a seemingly growing problem. Journal of Cetacean Research and Management 12(1):119-127.

Rivers, J. A. 1997. Blue whale, *Balaenoptera musculus*, vocalizations from the waters off central California. Marine Mammal Science 13(2):186–195.

Robertson, F. C. 2014. Effects of Seismic Operations on Bowhead Whale Behavior: Implications for Distribution and Abundance Assessments. The University of British Columbia, Vancouver, BC.

Robinson, R. A., and coauthors. 2005. Climate change and migratory species. Defra Research, British Trust for Ornithology, Norfolk, U.K. .

Rockwood, R. C., J. Calambokidis, and J. Jahncke. 2017. High mortality of blue, humpback and fin whales from modeling of vessel collisions on the U.S. West Coast suggests population impacts and insufficient protection. PLoS ONE 12(8):e0183052.

Rodgers, E. M., and coauthors. 2019. Integrating physiological data with the conservation and management of fishes: a meta-analytical review using the threatened green sturgeon (Acipenser medirostris). 7(1):coz035.

Roegner, G. C., R. McNatt, D. J. Teel, and D. L. Bottom. 2012. Distribution, size, and origin of juvenile Chinook salmon in shallow-water habitats of the lower Columbia River and estuary, 2002–2007. Marine and Coastal Fisheries 4(1):450-472.

Rolland, R. M., and coauthors. 2017. Fecal glucocorticoids and anthropogenic injury and mortality in North Atlantic right whales Eubalaena glacialis. Endangered Species Research 34:417-429.

Rolland, R. M., and coauthors. 2012. Evidence that ship noise increases stress in right whales. Proceedings of the Royal Society of London Series B Biological Sciences 279(1737):2363-2368.

Roman, J., and S. R. Palumbi. 2003. Whales before whaling in the North Atlantic. Science 301(5632):508-510.

Romano, T. A., and coauthors. 2002. Immune response, stress, and environment: Implications for cetaceans. Pages 253-279 *in* Molecular and Cell Biology of Marine Mammals. Krieger Publishing Co., Malabar, Florida.

Romano, T. A., and coauthors. 2004. Anthropogenic sound and marine mammal health: Measures of the nervous and immune systems before and after intense sound exposure. Canadian Journal of Fisheries and Aquatic Sciences 61:1124-1134.

Romero, L. M. 2004. Physiological stress in ecology: Lessons from biomedical research. Trends in Ecology and Evolution 19(5):249-255.

Romero, L. M., and M. Wikelski. 2001. Corticosterone levels predict survival probabilities of Galapagos marine iguanas during El Nino events. Proceedings of the National Academy of Sciences 98:7366-7370.

Rone, B. K., and coauthors. 2010. Results of the April 2009 Gulf of Alaska Line-Transect Survey (GOALS) in the Navy Training Exercise Area. NOAA.

Rosenbaum, H. C., and coauthors. 2000. World-wide genetic differentiation of *Eubalaena*: Questioning the number of right whale species. Molecular Ecology 9(11):1793-1802.

Rosenberger, A. L. J., M. MacDuffee, A. G. J. Rosenberger, and P. S. Ross. 2017. Oil spills and marine mammals in British Columbia, Canada: Development and application of a risk-based conceptual framework. Archives of Environmental Contamination and Toxicology 73:131-153.

Rosenthal, R. J., L. Haldorson, L. J. Field, V. Moran-O'Connell, and M. G. LaRiviere. 1982. Inshore and shallow offshore bottomfish resources in the southeastern Gulf of Alaska. Alaska Coastal Research, Sitka, Alaska.

Ross, D. 1976. Mechanics of underwater noise. Pergamon Press, New York.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: Effects of age, sex and dietary preference. Marine Pollution Bulletin 40(6):504-515.

Ross, P. S., and coauthors. 2013. Declining concentrations of persistent PCBs, PBDEs, PCDEs, and PCNs in harbor seals (*Phoca vitulina*) from the Salish Sea. Progress in Oceanography 115:160-170.

Rowlett, R. A., G. A. Green, C. E. Bowlby, and M. A. Smultea. 1994. The first photographic documentation of a northern right whale off Washington state. Northwestern Naturalist 75(3):102-104.

Royer, T. C. 2005. Hydrographic responses at a coastal site in the northern Gulf of Alaska to seasonal and interannual forcing. Deep-Sea Research Part Ii-Topical Studies in Oceanography 52(1-2):267-288.

Ruckelshaus, M., and coauthors. 2002. Planning ranges and preliminary guidelines for the delisting and recovery of the Puget Sound Chinook salmon evolutionarily significant unit. Puget Sound Technical Recovery Team. National Marine Fisheries Service, Northwest Fisheries Science Center, editor, Seattle.

Ruggerone, G. T., A. M. Springer, L. D. Shaul, and G. B. van Vliet. 2019. Unprecedented biennial pattern of birth and mortality in an endangered apex predator, the southern resident killer whale, in the eastern North Pacific Ocean. Marine Ecology Progress Series 608:291-296.

Ruud, J. T. 1956. The blue whale. (Balaenoptera musculus). Scientific American 195:46-50.

Saez, L. 2018. Understanding U.S. West Coast Whale Entanglements. National Marine Fisheries Service, West Coast Region, Long Beach, California.

Saez, L., and coauthors. 2012. Co-occurrence of Large Whales and Fixed Commercial Fishing Gear: California, Oregon, and Washington. Southern California Marine Mammal Workshop, Newport Beach, California.

Sakuma, K. M., and S. Ralston. 1995. Distributional patterns of late larval groundfish off central California in relation to hydrographic features during 1992 and 1993. California Cooperative Oceanic Fisheries Investigations Reports 36:179-192.

Salo, E. O. 1991. Life history of chum salmon (Oncorhynchus keta). Pages 231–309 *in* C. G. a. L. Margolis, editor. Pacific salmon life histories. University of British Columbia Press, Vancouver, B.C.

Samaran, F., C. Guinet, O. Adam, J. F. Motsch, and Y. Cansi. 2010. Source level estimation of two blue whale subspecies in southwestern Indian Ocean. Journal of the Acoustical Society of America 127(6):3800–3808.

Sandercock, F. 1991. Life history of coho salmon (Oncorhynchus kisutch). Pacific salmon life histories:396-445.

Sands, N. J., K. Rawson, K.P. Currens, W.H. Graeber, M.H., Ruckelshaus, R.R. Fuerstenberg, and J.B. Scott. 2009. Determination of Independent Populations and Viability Criteria for the Hood Canal Summer Chum Ssalmon Evolutionarily Significant Unit. National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

Santora, J. A., and coauthors. 2020. Habitat compression and ecosystem shifts as potential links between marine heatwave and record whale entanglements. Nature Communications 11(1):536.

Sapolsky, R. M., L. M. Romero, and A. U. Munck. 2000. How do glucocorticoids influence stress responses? Integrating permissive, suppressive, stimulatory, and preparative actions. Endocrine Reviews 21(1):55-89.

Sardella, B. A., D. J. P. Kültz, and B. Zoology. 2014. The physiological responses of green sturgeon (Acipenser medirostris) to potential global climate change stressors. 87(3):456-463.

Sarti M., L. 1996. A note about declining leatherbacks in Mexico. Marine Turtle Newsletter 75:25-26.

Saski, H., and coauthors. 2013. Habitat differentiation between sei (*Balaenoptera borealis*) and Bryde's whales (*B. brydei*) in the western North Pacific. Fisheries Oceanography 22(6):496-508.

Saunders, K. J., P. R. White, and T. G. Leighton. 2008. Models for predicting nitrogen tensions in diving odontocetes. Pages 88 *in* Twenty Second Annual Conference of the European Cetacean Society, Egnond aan Zee, The Netherlands.

Schakner, Z. A., and D. T. Blumstein. 2013. Behavioral biology of marine mammal deterrents: A review and prospectus. Biological Conservation 167:380-389.

Scheidat, M., C. Castro, J. Gonzalez, and R. Williams. 2004. Behavioural responses of humpback whales (*Megaptera novaeangliae*) to whalewatching boats near Isla de la Plata, Machalilla National Park, Ecuador. Journal of Cetacean Research and Management 6(1):63-68.

Scheifele, P. M., and coauthors. 2005. Indication of a Lombard vocal response in the St. Lawrence River beluga. Journal of the Acoustical Society of America 117(3):1486-1492.

Schevill, W. E., W. A. Watkins, and R. H. Backus. 1964. The 20-cycle signals and Balaenoptera (fin whales). Pages 147-152 *in* W. N. Tavolga, editor Marine Bio-acoustics. Pergamon Press, Lerner Marine Laboratory, Bimini, Bahamas.

Schlundt, C. E., J. J. Finneran, D. A. Carder, and S. H. Ridgway. 2000. Temporary shift in masked hearing thresholds of bottlenose dolphins, Tursiops truncatus, and white whales, Delphinapterus leucas, after exposure to intense tones. Journal of the Acoustical Society of America 107(6):3496-3508.

Schoenherr, J. R. 1991. Blue whales feeding on high concentrations of euphausiids around Monterey Submarine Canyon. (*Balaenoptera musculus*). Canadian Journal of Zoology 69(3):583-594.

Scholik, A. R., and H. Y. Yan. 2001. Effects of underwater noise on auditory sensitivity of a cyprinid fish. Hearing Research 152(2-Jan):17-24.

Scholik, A. R., and H. Y. Yan. 2002. The effects of noise on the auditory sensitivity of the bluegill sunfish, Lepomis macrochirus. Comparative Biochemistry and Physiology A Molecular and Integrative Physiology 133(1):43-52.

Schorr, G., E. A. Falcone, D. J. Moretti, and R. Andrews. 2014. First long-term behavioral records from Cuvier's beaked whales (Ziphius cavirostris) reveal record-breaking dives. PLoS ONE, 9(3), e92633. .

Schuyler, Q. A. 2014. Ingestion of marine debris by sea turtles. Doctoral dissertation. The University of Queensland.

Schuyler, Q. A., and coauthors. 2016. Risk analysis reveals global hotspots for marine debris ingestion by sea turtles. Global Change Biology 22(2):567-576.

Schwacke, L. H., and coauthors. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the southeast United States coast. Environmental Toxicology and Chemistry 21:2752-2764.

Scordino, J. J., and coauthors. 2017. Individual gray whale use of coastal waters off northwest Washington during the feeding season 1984–2011: Implications for management. Journal of Cetacean Research and Management 16:57–69.

Scott, T. M., and S. Sadove. 1997. Sperm whale, *Physeter macrocephalus*, sightings in the shallow shelf waters off Long Island, New York. Marine Mammal Science 13(2):4.

Seagars, D. J. 1984. The Guadalupe fur seal: A status review. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Sears, R. 1983. A glimpse of blue whales feeding in the Gulf of St. Lawrence. Whalewatcher 17(3):12-14.

Sears, R. 1987. Study on blue whales - brief description. Unpublished paper to the IWC Scientific Committee. 2 pp. Bournemouth, June (SC/39/PS20).

Seesholtz, A. M., M. J. Manuel, and J. P. Van Eenennaam. 2015. First documented spawning and associated habitat conditions for green sturgeon in the Feather River, California. Environmental Biology of Fishes 98(3):905-912.

Senigaglia, V., and coauthors. 2016. Meta-analyses of whale-watching impact studies: comparisons of cetacean responses to disturbance. Marine Ecology Progress Series 542:251-263.

Sericano, J. L., T. L. Wade, S. T. Sweet, J. Ramirez, and G. G. Lauenstein. 2014. Temporal trends and spatial distribution of DDT in bivalves from the coastal marine environments of the continental United States, 1986–2009. Marine Pollution Bulletin 81:303-316.

Settle, L. R., and coauthors. 2002. Investigation of impacts of underwater explosions on larval and early juvenile fishes.

Shapovalov, L., and A. C. Taft. 1954. The life histories of the steelhead rainbow trout (Salmo gairdneri gairdneri) and silver salmon (Oncorhynchus kisutch): with special reference to Waddell Creek, California, and recommendations regarding their management. California Department of Fish and Game.

Shared Strategy for Puget Sound. 2007. Puget Sound salmon recovery plan. Volume 1, recovery plan, Seattle.

Sharma, R., and T. P. Quinn. 2012. Linkages between life history type and migration pathways in freshwater and marine environments for Chinook salmon, Oncorhynchus tshawytscha. Acta Oecologica 41:1-13.

Shelden, K. E. W., S. E. Moore, J. M. Waite, P. R. Wade, and D. J. Rugh. 2005. Historic and current habitat use by North Pacific right whales *Eubalaena japonica* in the Bering Sea and Gulf of Alaska. Mammal Review 35(2):129-155.

Shelton, A. O., W. H. Satterthwaite, E. J. Ward, B. E. Feist, and B. Burke. 2019. Using hierarchical models to estimate stock-specific and seasonal variation in ocean distribution, survivorship, and aggregate abundance of fall run Chinook salmon. Canadian Journal of Fisheries and Aquatic Sciences 76(1):95-108.

Shields, M. W., J. Lindell, and J. Woodruff. 2018. Declining spring usage of core habitat by endangered fish-eating killer whales reflects decreased availability of their primary prey. Pacific Conservation Biology 24:189–193.

Shine, R., X. Bonnet, M. J. Elphick, and E. G. Barrott. 2004. A novel foraging mode in snakes: browsing by the sea snake *Emydocephalus annulatus* (Serpentes, Hydrophiidae). Functional ecology 18(1):16–24.

Silber, G., J. Slutsky, and S. Bettridge. 2010. Hydrodynamics of a ship/whale collision. Journal of Experimental Marine Biology and Ecology 391:19-Oct.

Silber, G. K. 1986. The relationship of social vocalizations to surface behavior and aggression in the Hawaiian humpback whale (*Megaptera novaeangliae*). Canadian Journal of Zoology 64(10):2075-2080.

Silber, G. K., and coauthors. 2017. Projecting Marine Mammal Distribution in a Changing Climate. Frontiers in Marine Science 4:14.

Silvestre, F., J. Linares-Casenave, S. I. Doroshov, and D. Kültz. 2010. A proteomic analysis of green and white sturgeon larvae exposed to heat stress and selenium. Science of the Total Environment 408(16):3176-3188.

Simao, S. M., and S. C. Moreira. 2005. Vocalizations of a female humpback whale in Arraial do Cabo (Rj, Brazil). Marine Mammal Science 21(1):150-153.

Simmonds, M. P. 2005. Whale watching and monitoring - some considerations. Unpublished paper to the IWC Scientific Committee. 5 pp. Ulsan, Korea, June (SC/57/WW5).

Simmonds, M. P., and W. J. Eliott. 2009. Climate change and cetaceans: Concerns and recent developments. Journal of the Marine Biological Association of the United Kingdom 89(1):203-210.

Simmonds, M. P., and J. D. Hutchinson. 1996. The conservation of whales and dolphins. John Wiley and Sons, Chichester, U.K.

Simmonds, M. P., and S. J. Isaac. 2007. The impacts of climate change on marine mammals: Early signs of significant problems. Oryx 41(1):19-26.

Sims, P. Q., S. K. Hung, and B. Wursig. 2012. High-speed vessel noises in West Hong Kong waters and their contributions relative to Indo-Pacific humpback dolphins (*Sousa chinensis*). Journal of Marine Biology 2012:11.

Širović, A., and coauthors. 2012. Passive Acoustic Monitoring for Marine Mammals in the Northwest Training Range Complex 2011. Marine Physical Laboratory, Scripps Institute of Oceanography, University of California San Diego, La Jolla, CA.

Sirovic, A., J. A. Hildebrand, and S. M. Wiggins. 2007. Blue and fin whale call source levels and propagation range in the Southern Ocean. Journal of the Acoustical Society of America 122(2):1208-1215.

Sirovic, A., and coauthors. 2015. North Pacific right whales (*Eubalaena japonica*) recorded in the northeastern Pacific Ocean in 2013. Marine Mammal Science 31(2):800-807.

Širović, A., and coauthors. 2011. Marine Mammal Demographics of the Otuer Washington Coast During 2008 - 2009. Naval Postgraduate School, Monterey, California.

Sirovic, A., L. N. Williams, S. M. Kerosky, S. M. Wiggins, and J. A. Hildebrand. 2012. Temporal separation of two fin whale call types across the eastern North Pacific. Marine Biology 160(1):47-57.

Sivle, L., and coauthors. 2012. Changes in dive behavior during naval sonar exposure in killer whales, long-finned pilot whales, and sperm whales. Frontiers in Physiology 3(400).

Sivle, L. D., and coauthors. 2015. Severity of expert-identified behavioural responses of humpback whale, minke whale, and northern bottlenose whale to naval sonar. Aquatic Mammals, 41(4), 469–502. .

Sivle, L. D., and coauthors. 2016. Naval sonar disrupts foraging in humpback whales. Marine Ecology Progress Series 562:211-220.

Slijper, E. J. 1962. Whales. English translation Hutchinson & Co. (Publishers). First published in the U.S. by Basic Books Publishing Co., Inc, New York. 475pp.

Smith, A. W., and A. B. Latham. 1978. Prevalence of vesicular exanthema of swine antibodies among feral mammals associated with the southern California coastal zones. American Journal of Veterinary Research 39(2):291-6.

Smith, C. E., and coauthors. 2016. Assessment of known impacts of unmanned aerial systems (UAS) on marine mammals: data gaps and recommendations for researchers in the United States1. Journal of Unmanned Vehicle Systems 4(1):31-44.

Smith, J. N., A. W. Goldizen, R. A. Dunlop, and M. J. Noad. 2008. Songs of male humpback whales, Megaptera novaeangliae, are involved in intersexual interactions. Animal Behaviour 76(2):467-477.

Smith, M. E., A. B. Coffin, D. L. Miller, and A. N. Popper. 2006. Anatomical and functional recovery of the goldfish (Carassius auratus) ear following noise exposure. Journal of Experimental Biology 209(21):4193-4202.

Smith, M. E., A. S. Kane, and A. N. Popper. 2004. Noise-induced stress response and hearing loss in goldfish (Carassius auratus). Journal of Experimental Biology 207(3):427-435.

Smith, S. H., and D. E. Marx Jr. 2016. De-facto marine protection from a Navy bombing range: Farallon De Medinilla, Mariana Archipelago, 1997 to 2012. Marine Pollution Bulletin 102(1):187-198.

SMRU Consulting North America. 2017. Estimating the effects of noise from commercial vessels and whale watch boats on Southern Resident Killer Whales. Prepared for the ECHO Program of Vancouver Fraser Port Authority.

Smultea, M. A., C. E. Bacon, and J. S. D. Black. 2011. Aerial Survey Marine Mammal Monitoring off Southern California in Conjunction with US Navy Major Training Events (MTE), July 27- August 3 and September 23–28, 2010—Final Report, June 2011. Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, HI. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, HI, 96860 3134, under Contract No. N00244-10-C-0021 issued to University of California, San Diego, 7835 Trade St., San Diego, CA 92121. Submitted by Smultea Environmental Sciences (SES), Issaquah, WA, 98027, www.smultea.com, under Purchase Order No. 10309963.

Smultea, M. A., C. E. Bacon, T. F. Norris, and D. Steckler. 2012. Aerial Surveys Conducted in the SOCAL OPAREA From 1 August 2011–31 July 2012. Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, HI. Submitted to Naval Facilities Engineering Command Southwest (NAVFAC SW), EV5 Environmental, San Diego, 92132 under Contract No. N62470-10-D-3011 issued to HDR, Inc., San Diego, CA. Submitted August 2012.

Smultea, M. A., T. A. Jefferson, and A. M. Zoidis. 2010. Rare sightings of a bryde's whale (*Balaenoptera edeni*) and sei whales (*B. borealis*) (Cetacea: Balaenopteridae) northeast of O'ahu, Hawai'i. Pacific Science 64(3):449-457.

Smultea, M. A., K. MacNair-Lomac, G. Campbell, S. Courbis, and T. A. Jefferson. 2017. Aerial Survey of Marine Mammals Conducted in the Inland Puget Sound Waters of Washington, Summer 2013-Winter 2016. Final Report. Smultea Environmental Sciences, LLC, San Diego, California.

Smultea, M. A., and J. R. Mobley. 2009. Aerial Survey Monitoring of Marine Mammals and Sea Turtles in conjunction with SCC OPS Navy Exercises off Kauai, 18–21 August 2008, Final Report, May 2009. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, HI. Prepared by Marine Mammal Research Consultants, Honolulu, HI, and Smultea Environmental Sciences, LLC., Issaquah, WA, under Contract No. N62742-08-P-1942.

Smultea, M. A., J. R. Mobley, D. Fertl, and G. L. Fulling. 2008. An unusual reaction and other observationis of sperm whales near fixed-wing aircraft. Gulf and Caribbean Research 20:75-80.

Smultea, M. A., J. R. Mobley Jr., and K. Lomac-Macnair. 2009. Aerial survey monitoring for marine mammals and sea turtles in conjunction with US Navy major training events off San Diego, California, 15-21 October and 15-18 November 2008, final report. Naval Facilities Engineering Command Pacific, EV2 Environmental Planning, Pearl Harbor, Hawaii.

Snider, B., and R. G. Titus. 2000. Timing, Composition, and Abundance of Juvenile Anadromous Salmonid Emigration in the Sacramento River Near Knights Landing October1997-September 1998. California Department of Fish and Game Stream Evaluation Program Technical Report No. 00-5.

Snoddy, J. E., M. Landon, G. Blanvillain, and A. Southwood. 2009. Blood biochemistry of sea turtles captured in gillnets in the lower Cape Fear River, North Carolina, USA. Journal of Wildlife Management 73(8):1394–1401.

Sogard, S., T. H. Williams, and H. Fish. 2009. Seasonal patterns of abundance, growth, and site fidelity of juvenile steelhead in a small coastal California stream. Transaction of American Fisheries Sociery 138(3):549-563.

Sohn, R. A., F. Vernon, J. A. Hildebrand, and S. C. Webb. 2000a. Field measurements of sonic boom penetration into the ocean. Journal of the Acoustical Society of America 107(6):3073-3083.

Sohn, R. A., F. Vernon, J. A. Hildebrand, and S. C. Webb. 2000b. Field measurements of sonic boom penetration into the ocean. The Journal of the Acoustical Society of America 107(6):3073–3083.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: Evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58:325-333.

Sonoma County Water Agency (SCWA). 2002. Documenting biodiveristy of coastal salmon (*Oncorhynchus* spp.) in Northern California. Bodega Marine Laboratory, University of California at Davis, TW 99/00-110, Bodega Bay, California.

Soule, D. C., and W. S. D. Wilcock. 2013. Fin whale tracks recorded by seismic network on the Juan de Fuca Ridge, Northeast Pacific Ocean. The Journal of the Acoustical Society of America 133(3):1–29.

Sousa-Lima, R. S., and C. W. Clark. 2008. Modeling the effect of boat traffic on the fluctuation of humpback whale singing activity in the Abrolhos National Marine Park, Brazil. Canadian Acoustics 36(1):174-181.

Southall, B., and coauthors. 2007a. Aquatic mammals marine mammal noise exposure criteria: Initial scientific recommendations. Aquatic Mammals 33(4):122.

Southall, B., and coauthors. 2013. Measuring cetacean reponses to military sonar: Behavioral response studies in southern California (SOCAL-BRS). Pages 196 *in* Twentieth Biennial Conference on the Biology of Marine Mammals, Dunedin, New Zealand.

Southall, B., and coauthors. 2012. Biological and behavioral response studies of marine mammals in Southern California, 2011 (SOCAL-11) final project report.

Southall, B., and coauthors. 2011. Biological and Behavioral Response Studies of Marine Mammals in Southern California, 2010 ("SOCAL -10").

Southall, B. L., and coauthors. 2007b. Marine mammal noise exposure criteria: Initial scientific recommendations. Aquatic Mammals 33(4):411-521.

Southall, B. L., and coauthors. 2019a. Behavioral responses of individual blue whales (Balaenoptera musculus) to mid-frequency military sonar. Journal of Experimental Biology 222(Pt 5).

Southall, B. L., and coauthors. 2019b. Behavioral responses of individual blue whales (Balaenoptera musculus) to mid-frequency military sonar. Journal of Experiential Biology 222:15.

Southall, B. L., and coauthors. 2019c. Marine Mammal Noise Exposure Criteria: Updated Scientific Recommendations for Residual Hearing Effects. Aquatic Mammals 45(2):125-232.

Southall, B. L., D. P. Nowacek, P. J. O. Miller, and P. L. Tyack. 2016. Experimental field studies to measure behavioral responses of cetaceans to sonar. Endangered Species Research 31:293-315.

Southall, B. L., and coauthors. 2009. Behavioral responses of beaked whales and other cetaceans to controlled exposures of simulated sonar and other sounds. Paper presented at the 18th Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada. .

Southall, R. B., S. W. Martin, D. L. Webster, and B. L. 2014. Assessment of modeled received sound pressure levels and movements of satellite-tagged odontocetes exposed to mid-frequency active sonar at the Pacific Missile Range Facility: February 2011 through February 2013. U. S. Pacific Fleet.

Southwood, A. L., and coauthors. 1999. Heart rates and diving behavior of leatherback sea turtles in the Eastern Pacific Ocean. Journal of Experimental Biology 202(9):1115-1125.

Sparrow, V. W. 2002a. Review and status of sonic boom penetration into the ocean. The Journal of the Acoustical Society of America 111(1):537–543.

Sparrow, V. W. 2002b. Review and status of sonic boom penetration into the ocean. Journal of the Acoustical Society of America 111(1):537-543.

Spence, B. C. 2016. North-Central California Coast Recovery Domain. T. H. Williams, and coeditors, editors. Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. National Marine Fisheries Service – West Coast Region, Southwest Fisheries Science Center, Fisheries Ecology Division, Santa Cruz, California.

Spence, B. C., and coauthors. 2008. A framework for assessing the viability of threatened and endangered salmon and steelhead in the north-central California coast recovery domain.

Spence, B. C., G. A. Lomnicky, R. M. Hughs, and R. P. Novitzki. 1996. An ecosystem approach to salmonid conservation. Man Tech Environmental Research Services Corps, TR-4501-96-6057, Corvallis, Oregon.

Spotila, J. R. 2004. Sea turtles: a complete guide to their biology, behavior, and conservation. The Johns Hopkins University Press and Oakwood Arts, Baltimore, Maryland.

Spotila, J. R., and coauthors. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? Chelonian Conservation and Biology 2(2):209-222.

Spotila, J. R., R. D. Reina, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 2000. Pacific leatherback turtles face extinction. Nature 405:529-530.

Sremba, A. L., B. Hancock-Hanser, T. A. Branch, R. L. LeDuc, and C. S. Baker. 2012. Circumpolar diversity and geographic differentiation of mtDNA in the critically endangered Antarctic blue whale (*Balaenoptera musculus intermedia*). PLoS ONE 7(3):e32579.

SSSRT. 2010. A Biological Assessment of Shortnose Sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

St Aubin, D. J., and L. A. Dierauf. 2001. Stress and marine mammals. Pages 253-269 *in* L. A. D. F. M. D. Gullands, editor. CRC Handbook of Marine Mammal Medicine, Second edition. CRC Press, Boca Raton, Florida.

St. Aubin, D. J., and J. R. Geraci. 1989. Adaptive changes in hematologic and plasma chemical constituents in captive beluga whales, Delphinapterus leucas. Canadian Journal of Fisheries and Aquatic Sciences 46:796-803.

St. Aubin, D. J., S. H. Ridgway, R. S. Wells, and H. Rhinehart. 1996. Dolphin thyroid and adrenal hormones: Circulating levels in wild and semidomesticated Tursiops truncatus, and influence of sex, age, and season. Marine Mammal Science 12(1):13-Jan.

Stabeno, P. J., and coauthors. 2004. Meteorology and oceanography of the northern Gulf of Alaska. Continental Shelf Research 24-Jan(8-Jul):859-897.

Stafford, K. M. 2003. Two types of blue whale calls recorded in the Gulf of Alaska. Marine Mammal Science 19(4):12.

Stafford, K. M., C. G. Fox, and D. S. Clark. 1998. Long-range acoustic detection and localization of blue whale calls in the northeast Pacific Ocean (*Balaenoptera musculus*). Journal of the Acoustical Society of America 104(6):3616–3625.

Stafford, K. M., D. K. Mellinger, S. E. Moore, and C. G. Fox. 2007. Seasonal variability and detection range modeling of baleen whale calls in the Gulf of Alaska, 1999–2002. The Journal of the Acoustical Society of America 122(6):3378-3390.

Stafford, K. M., and S. E. Moore. 2005. Atypical calling by a blue whale in the Gulf of Alaska. Journal of the Acoustical Society of America 117(5):2724–2727.

Stafford, K. M., S. L. Nieukirk, and C. G. Fox. 2001. Geographic and seasonal variation of blue whale calls in the North Pacific (*Balaenoptera musculus*). Journal of Cetacean Research and Management 3(1):65–76.

Standora, E. A., J. R. Spotila, J. A. Keinath, and C. R. Shoop. 1984. Body temperatures, diving cycles, and movement of a subadult leatherback turtle, *Dermochelys coriacea*. Herpetologica 40:169-176.

Steiger, G. H., and coauthors. 2008. Geographic variation in killer whale attacks on humpback whales in the North Pacific: Implications for predation pressure. Endangered Species Research 4(3):247-256.

Stensland, E., and P. Berggren. 2007. Behavioural changes in female Indo-Pacific bottlenose dolphins in response to boat-based tourism. Marine Ecology Progress Series 332:225-234.

Stimpert, A. K., and coauthors. 2014. Acoustic and foraging behavior of a Baird's beaked whale, Berardius bairdii, exposed to simulated sonar. Scientific Reports 4(7031):8.

Stimpert, A. K., D. N. Wiley, W. W. L. Au, M. P. Johnson, and R. Arsenault. 2007. 'Megapclicks': Acoustic click trains and buzzes produced during night-time foraging of humpback whales (Megaptera novaeangliae). Biology Letters 3(5):467-470.

Stinson, M. L. 1984. Biology of sea turtles in San Diego Bay, California, and in the northeastern Pacific Ocean. San Diego State University.

Stockin, K., D. Lusseau, V. Binedell, N. Wiseman, and M. Orams. 2008. Tourism affects the behavioural budget of the common dolphin *Delphinus* sp. in the Hauraki Gulf, New Zealand. Marine Ecology Progress Series 355:287-295.

Stone, G. S., S. K. Katona, A. Mainwaring, J. M. Allen, and H. D. Corbett. 1992. Respiration and surfacing rates of fin whales (Balaenoptera physalus) observed from a lighthouse tower. Report of the International Whaling Commission 42:739-745.

Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. Chemosphere 70(5):908-913.

Straley, J., and T. O'Connell. 2005. Sperm whale interactions with longline fisheries in the Gulf of Alaska. Oncorhynchus 15(1):1-2.

Straley, J., T. O'Connell, L. Behnken, and A. Thode. 2005. Southeast Alaska Sperm Whale Avoidance Project: Fishermen and Scientists Working to Reduce Whale Depredation on Longlines.

Strayer, D. L. 2010. Alien species in fresh waters: ecological effects, interactions with other stressors, and prospects for the future. Freshwater Biology 55:152-174.

Studebaker, R. S., K. N. Cox, and T. J. Mulligan. 2009. Recent and historical spatial distributions of juvenile rockfish species in rocky intertidal tide pools, with emphasis on black rockfish. Transactions of the American Fisheries Society 138:645-651.

Suarez, A., P. H. Dutton, and J. Bakarbessy. 2000. Leatherback (*Dermochelys coriacea*) nesting on the north Vogelkop coast of Irian Jaya, Indonesia. Pages 260 *in* H. Kalb, and T. Wibbels, editors. Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Summers, T. M., S. L. Martin, J. R. Hapdei, J. K. Ruak, and T. T. Jones. 2018. Endangered green turtles (Chelonia mydas) of the Northern Mariana Islands: Nesting ecology, poaching, and climate concerns. Frontiers in Marine Science 4:428.

Sverdrup, A., and coauthors. 1994. Effects of experimental seismic shock on vasoactivity of arteries, integrity of the vascular endothelium and on primary stress hormones of the Atlantic salmon. Journal of Fish Biology 45(6):973–995.

Swartz, S. L., B. L. Taylor, and D. J. Rugh. 2006. Gray whale Eschrichtius robustus population and stock identity. Mammal Review 36(1):66-84.

Sweeney, K. L., and coauthors. 2015. Flying beneath the clouds at the edge of the world: Using a hexacopter to supplement abundance surveys of Steller sea lions (*Eumetopias jubatus*) in Alaska. Journal of Unmanned Vehicle Systems 4(1):70–81.

Swope, B., and J. McDonald. 2013. Copper-Based Torpedo Guidance Wire: Applications and Environmental Considerations. San Diego, CA.

Szymanski, M. D., and coauthors. 1999. Killer whale (*Orcinus orca*) hearing: auditory brainstem response and behavioral audiograms. Journal of the Acoustical Society of America 106(2):1134-1141.

Tabor, R. A., H. A. Gearns, C. M. McCoy III, and S. Camacho. 2006. Nearshore habitat use by Chinook salmon in lentic systems of the Lake Washington basin. U.S. Fish and Wildlife Service, Lacy, Washington.

Tal, D., H. Shachar-Bener, D. Hershkovitz, Y. Arieli, and A. Shupak. 2015. Evidence for the initiation of decompression sickness by exposure to intense underwater sound. Journal of Neurophysiology 114(3):1521-1529.

Tapilatu, R. F., and coauthors. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: A globally important sea turtle population. Ecosphere 4:15.

Tavolga, W. N., and J. Wodinsky. 1963. Auditory capacities in fishes: Pure tone thresholds in nine species of marine teleosts. Bulletin of the American Museum of Natural History 126(2):179–239.

Taylor, A. H., M. B. Jordon, and J. A. Stephens. 1998. Gulf Stream shifts following ENSO events. Nature 393:68.

Teel, D., B. Burke, D. Kuligowski, C. Morgan, and D. Van Doornik. 2015. Genetic Identification of Chinook Salmon: Stock-Specific Distributions of Juveniles along the Washington and Oregon Coasts. Marine and Coastal Fisheries Dynamics Management and Ecosystem Science 7:274-300.

Teel, D. J., and coauthors. 2014. Genetic identification of Chinook salmon in the Columbia River estuary: stock-specific distributions of juveniles in shallow tidal freshwater habitats. North American Journal of Fisheries Management 34(3):621-641.

Teloni, V., W. M. X. Zimmer, M. Wahlberg, and P. T. Madsen. 2007. Consistent acoustic size estimation of sperm whales using clicks recorded from unknown aspects. Journal of Cetacean Research and Management 9(2):127-136.

Tennessen, J., and coauthors. 2019a. Kinematic signatures of prey capture from archival tags reveal sex differences in killer whale foraging activity. The Journal of Experimental Biology 222.

Tennessen, J., and coauthors. 2019b. Hidden Markov models reveal temporal patterns and sex differences in killer whale behavior. Scientific Reports 9:14951.

Tennessen, J., and S. Parks. 2016. Acoustic propagation modeling indicates vocal compensation in noise improves communication range for North Atlantic right whales. Endangered Species Research, 30, 225–237.

Terdalkar, S., A. S. Kulkarni, S. N. Kumbhar, and J. Matheickal. 2005. Bio-economic risks of ballast water carried in ships, with special reference to harmful algal blooms. Nature, Environment and Pollution Technology 4(1):43-47.

Terhune, J. M., and W. C. Verboom. 1999. Right whales and ship noises. Marine Mammal Science 15(1):256-258.

TEWG. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.

The Northwest Seaport Alliance. 2018. The Northwest Seaport Alliance 5-Year Cargo Volume History.

Thode, A., J. Straley, C. O. Tiemann, K. Folkert, and V. O'Connell. 2007. Observations of potential acoustic cues that attract sperm whales to longline fishing in the Gulf of Alaska. Journal of the Acoustical Society of America 122(2):1265-1277.

Thomas, J. A., R. A. Kastelein, and F. T. Awbrey. 1990. Behavior and blood catecholamines of captive belugas during playbacks of noise from ships and oil drilling platform. Zoo Biology 9(5):393-402.

Thomas, P. O., R. R. Reeves, and R. L. Brownell. 2016. Status of the world's baleen whales. Marine Mammal Science 32(2):682-734.

Thompson, P. M., and coauthors. 2013. Short-term disturbance by a commercial two-dimensional seismic survey does not lead to long-term displacement of harbour porpoises. Proceedings of the Royal Society B 280(1771):20132001.

Thompson, P. M., and coauthors. 2010. Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. Marine Pollution Bulletin 60(8):1200–1208.

Thompson, P. O., W. C. Cummings, and S. J. Ha. 1986. Sounds, source levels, and associated behavior of humpback whales, Southeast Alaska. Journal of the Acoustical Society of America 80(3):735-740.

Thompson, P. O., L. T. Findley, O. Vidal, and W. C. Cummings. 1996. Underwater sounds of blue whales, *Balaenoptera musculus*, in the Gulf of California, Mexico. Marine Mammal Science 12(2):288–293.

Thompson, P. O., L. T. Findley, and O. Vidal. 1992a. 20-Hz pulses and other vocalizations of fin whales, Balaenoptera physalus, in the Gulf of California, Mexico. Journal of the Acoustical Society of America 92(6):3051-3057.

Thompson, P. O., F. L. T., and O. Vidal. 1992b. 20-Hz pulses and other vocalizations of fin whales, *Balaenoptera physalus*, in the Gulf of California, Mexico. Journal of the Acoustical Society of America 92(6):3051–3057.

Thomsen, F., D. Franck, and J. K. Ford. 2002. On the communicative significance of whistles in wild killer whales (Orcinus orca). Naturwissenschaften 89(9):404-407.

Thomsen, F., D. Franck, and J. K. B. Ford. 2001. Characteristics of whistles from the acoustic repertoire of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Journal of the Acoustical Society of America 109(3):1240-1246.

Tian, Z., and coauthors. 2019. Suspect and Non-Target Screening for Contaminants of Emerging Concern in Puget Sound. 2019 Salmon Recovery Conference. SRC.

Tillman, M. F. 1977. Estimates of population size for the North Pacific sei whale. (Balaenoptera borealis). Report of the International Whaling Commission Special Issue 1:98-106.-Sc/27/Doc 25).

Tixier, P., N. Gasco, G. Duhamel, and C. Guinet. 2014. Habituation to an acoustic harassment device (AHD) by killer whales depredating demersal longlines. ICES Journal of Marine Science, 72(5), 1673–1681. .

Todd, S., P. Stevick, J. Lien, F. Marques, and D. Ketten. 1996. Behavioural effects of exposure to underwater explosions in humpback whales (*Megaptera novaeanlgiae*). Canadian Journal of Zoology 74:1661–1672.

Tolimieri, N., and P. S. Levin. 2005. The roles of fishing and climate in the population dynamics of bocaccio rockfish. Ecological Applications 15:458-468.

Tonnessen, J. N., and A. O. Johnsen. 1982. The history of modern whaling. University of California Press, Berkeley, CA.

Tougaard, J., and coauthors. 2005. Effects of the Nysted Offshore wind farm on harbour porpoises.

Tougaard, J., J. Carstensen, J. Teilmann, H. Skov, and P. Rasmussen. 2009. Pile driving zone of responsiveness extends beyond 20 km for harbor porpoises (*Phocoena phocoena* (L.)). Journal of the Acoustical Society of America 126(1):11.

Towers, J. R., and coauthors. 2018. Occurrence of fin whales (*Balaenoptera physalus*) between Vancouver Island and continental North America. Northwestern Naturalist 99:49–57.

Townsend, C. H. 1899. Notes on the fur seals of Guadalupe, the Galapagos and Lobos Islands. Pages 265-274 *in* D. S. Jordan, editor. The Fur Seals and Fur-Seal Islands of the North Pacific Ocean, volume Part 3. U.S. Government Printing Office, Washington, D. C.

Townsend, C. H. 1924. The northern elephant seal and the Guadalupe fur seal. Natural History 24(5):567-577.

Tribe, S. I., and W. D. o. F. a. Wildlife. 2007. Skokomish River Chinook Salmon Recovery Plan.

Trickey, J. S., and coauthors. 2015a. Passive Acoustic Monitoring for Marine Mammals in the Northwest Training Range Complex July 2013–April 2014. La Jolla, CA: Marine Physical Laboratory Scripps Institution of Oceanography University of California San Diego.

Trickey, J. S., and coauthors. 2015b. Passive Acoustic Monitoring for Marine Mammals in the Northwest Training Range Complex July 2013–April 2014. Marine Physical Laboratory, Scripps Institution of Oceanography, University of California, San Diego, La Jolla, CA.

Trudel, M., and coauthors. 2009a. Distribution and migration of juvenile Chinook salmon derived from coded wire tag recoveries along the continental shelf of western North America. Transactions of the American Fisheries Society 138(6):1369-1391.

Trudel, M., and coauthors. 2009b. Distribution and Migration of Juvenile Chinook Salmon Derived from Coded Wire Tag Recoveries along the Continental Shelf of Western North America. Transactions of the American Fisheries Society 138(6):1369-1391.

Trumble, S. J., E. M. Robinson, M. Berman-Kowalewski, C. W. Potter, and S. Usenko. 2013. Blue whale earplug reveals lifetime contaminant exposure and hormone profiles. Proceedings of the National Academy of Sciences 110(42):16922-16926.

Tseng, Y.-P., Y.-C. Huang, G. T. Kyle, and M.-C. Yang. 2011. Modeling the impacts of cetacean-focused tourism in Taiwan: Observations from cetacean watching boats: 2002-2005. Environmental Management 47(1):56-66.

Tsidulko, G. A., Q. Zhu, E. Sun, and M. A. Vorontsova. 2005. Scammon Lagoon for the western North Pacific gray whales? Pages 285 *in* Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Tucker, S., and coauthors. 2011. Life history and seasonal stock-specific ocean migration of juvenile Chinook salmon. Transactions of the American Fisheries Society 140(4):1101-1119.

Tucker, S., and coauthors. 2009. Seasonal stock-specific migrations of juvenile sockeye salmon along the west coast of North America: Implications for growth. Transactions of the American Fisheries Society 138(6):1458-1480.

Tyack, P. 1981. Interactions between singing Hawaiian humpback whales and conspecifics nearby. (Megaptera novaeangliae). Behavioral Ecology and Sociobiology 8(2):105-116.

Tyack, P. 1983. Differential response of humpback whales, Megaptera novaeangliae, to playback of song or social sounds. Behavioral Ecology and Sociobiology 13(1):49-55.

Tyack, P., and coauthors. 2011a. Response of Dtagged Cuvier's beaked whale, Ziphius cavirostris, to controlled exposure of sonar sound. Pages 297 *in* Nineteenth Biennial Conference on the Biology of Marine Mammals, Tampa, Florida.

Tyack, P. L. 1999. Communication and cognition. Pages 287-323 *in* J. E. R. III, and S. A. Rommel, editors. Biology of Marine Mammals. Smithsonian Institution Press, Washington.

Tyack, P. L., M. Johnson, N. Aguilar Soto, A. Sturlese, and P. T. Madsen. 2006. Extreme deep diving of beaked whales. Journal of Experimental Biology 209:4238-4253.

Tyack, P. L., and coauthors. 2011b. Beaked whales respond to simulated and actual Navy sonar. PLoS ONE 6(3):e17009.

Tynan, C. T., and coauthors. 2005. Cetacean distributions relative to ocean processes in the northern California Current System. Deep Sea Research Part Ii: Topical studies in Oceanography 52(1-2)145-167.

Tyurneva, O. Y., Y. M. Yakovlev, and V. V. Vertyankin. 2009. Photographic identification of the Korean-Okhotsk gray whale (Eschrichtius robustus) offshore northeast Sakhalin Island and southeast Kamchatka Peninsula (Russia), 2008. Unpublished paper to the IWC Scientific Committee, Madeira, Portugal.

U. S. Department of the Navy. 2015. Environmental Monitoring of Explosive Ordnance Disposal, Underwater Detonation Training, in the Northwest Training Range Complex on August 18, 2015. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Army. 1999. Finding of No Significant Impact for the Life Cycle Environmental Assessment for the HELLFIRE Modular Missile System. U.S. Department of Defense, Washington, DC.

U.S. Department of the Navy. 1981. Gun Blast Far Field Peak Overpressure Contours. Naval Surface Weapons Center, Silver Spring, MD.

U.S. Department of the Navy. 1991. Environmental Assessment for the Naval Sonobuoy Testing in Southeast Alaska. Naval Avionics Center, Indianapolis, Indiana.

U.S. Department of the Navy. 2000. Noise Blast Test Results Aboard the USS Cole. Gun Blast Transmission into Water Test with a 5-Inch/ 54 Caliber Naval Gun (Standard Ordnance). Naval Surface Warfare Center Dahlgren Division, Dahlgren, VA.

U.S. Department of the Navy. 2001. Sonic Boom Parametric Study. Applied Ordnance Technology, Inc. and Operational Environmental Planning Office Naval Air Station Patuxent River, MD.

U.S. Department of the Navy. 2009a. U.S. Navy Explosive Ordnance Disposal (EOD) Training Activities for 2009 in Crescent Harbor. U.S. Department of the Navy, Washington, DC.

U.S. Department of the Navy. 2009b. VACAPES Range Complex Final Environmental Impact Statement/ Overseas Environmental Impact Statement. United States Fleet Forces Command, Washington, DC.

U.S. Department of the Navy. 2010. Laser System Usage in the Marine Environment: Applications and Environmental Considerations. Space and Naval Warfare Systems Command Center Pacific, Technical Report 1996, San Diego, CA.

U.S. Department of the Navy. 2013a. Environmental Monitoring of Explosive Ordnance Disposal Underwater Detonation Training in the Northwest Training Range Complex on August 28, 2013. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Navy. 2013b. Petition for Regulations Pursuant to Section 101(a)(5) of the Marine Mammal Protection Act Covering Taking of Marine Mammals Incidental to Target and Missile Launch Activities for the Period 2014–2019 at San Nicolas Island,

California (50 CFR Part 216, Subpart I). Office of Protected Resources, National Marine Fisheries Service, National Oceanographic and Atmospheric Administration, Point Mugu, CA.

U.S. Department of the Navy. 2014a. Environmental Monitoring of Explosive Ordnance Disposal Underwater Detonation Training in the Northwest Training Range Complex on April 02, 2014. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Navy. 2014b. U.S. Navy Testing of Hypervelocity Projectiles and an Electromagnetic Railgun. Pages 1–260 *in* N. A. a. S. A. s. W. F. Facility, editor. U.S. Department of the Navy, Wallops Island, VA.

U.S. Department of the Navy. 2015a. Environmental Monitoring of Explosive Ordnance Disposal Underwater Detonation Training in the Northwest Training Range Complex on March 25, 2015. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Navy. 2015b. Environmental Monitoring of Explosive Ordnance Disposal Underwater Detonation Training in the Northwest Training Range Complex on September 23, 2015. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Navy. 2015c. Environmental Monitoring of Explosive Ordnance Disposal Underwater Detonation Training in the Northwest Training Range Complex on September 24, 2014. Navy Region Northwest, Naval Facilities Engineering Command, and Naval Undersea Warfare Center, Keyport, WA.

U.S. Department of the Navy. 2017a. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III). Space and Naval Warfare System Command, Pacific.

U.S. Department of the Navy. 2017b. Dive Distribution and Group Size Parameters for Marine Species Occurring in the U.S. Navy's Atlantic and Hawaii-Southern California Training and Testing Study Areas. Naval Undersea Warfare Center Division, Newport, RI.

U.S. Department of the Navy. 2018a. 2017 U.S. Navy Annual Marine Species Monitoring Report for the Pacific: A Multi-Range-Complex Monitoring Report For Hawaii-Southern California Training and Testing (HSTT), Mariana Islands Training and Testing (MITT), Northwest Training and Testing (NWTT), and the Gulf of Alaska Temporary Maritime Activities Area (GOA TMAA). National Marine Fisheries Service, Silver Spring, MD.

U.S. Department of the Navy. 2018b. Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing. Naval Undersea Warfare Center, Newport, RI.

U.S. Department of the Navy, and Department of Defense. 2007. Finding of No Significant Harm and Final Environmental Assessment of the Naval Air Routine Training Exercises in East and Gulf Coast Operation Areas and Seaward. Atlantic Division, Naval Facilities Engineering Command, Norfolk, VA.

U.S. Naval Research Advisory Committee. 2009. Report on Jet Engine Noise Reduction. Department of Defense, Patuxent River, MD.

Ulrich, R. M. 2004. Development of a Sensitive and Specific Biosensor Assay to Detect *Vibrio vulnificus* in Estuarine Waters. Unpublished master's thesis. University of South Florida, Tampa, FL.

Urick, R. J. 1983a. Principles of Underwater Sound. McGraw-Hill.

Urick, R. J. 1983b. *Principles of Underwater Sound*. 3rd edition. Peninsula Publishing, Los Altos, CA.

Urick, R. J. 1983c. Principles of Underwater Sound, Principles of Underwater Sound for Engineers, 3rd edition. Peninsula Publishing, Los Altos Hills, CA.

USCG, EPA, Oregon Department of Environmental Quality, Idaho Office of Emergency Management, and Washington Department of Ecology. 2018. Northwest Area Contingency Plan. Available online: https://www.rrt10nwac.com/Files/NWACP/2018/Northwest%20Area%20Contingency%20Plan%202018.pdf.

USDC. 2009. Endangered and threatened wildlife and plants: Final rulemaking to designate critical habitat for the threatened southern distinct population segment of North American green sturgeon. U.S. Department of Commerce, National Marine Fisheries Service. Federal Register 74(195):52300-52351.

USDC. 2014. Endangered and threatened wildlife; Final rule to revise the Code of Federal Regulations for species under the jurisdiction of the National Marine Fisheries Service. U.S Department of Commerce. Federal Register 79(71):20802-20817.

Vallejo, G. C., and coauthors. 2017. Responses of two marine top predators to an offshore wind farm. Ecology and Evolution 7(21):8698–8708.

van Beest, F. M., L. Kindt-Larsen, F. Bastardie, V. Bartolino, and J. Nabe-Nielsen. 2017. Predicting the population-level impact of mitigating harbor porpoise bycatch with pingers and time-area fishing closures. Ecosphere 8(4):e01785.

Van Doornik, D. M., B. R. Beckman, J. H. Moss, W. W. Strasburger, and D. J. Teel. 2019. Stock specific relative abundance of Columbia River juvenile Chinook salmon off the Southeast Alaska coast. Deep Sea Research Part II: Topical Studies in Oceanography 165:322-328.

Van Doornik, D. M., and coauthors. 2015. Genetic Population Structure of Willamette River Steelhead and the Influence of Introduced Stocks. Transactions of the American Fisheries Society 144(1):150-162.

Van Eenennaam, J. P., and coauthors. 2006. Reproductive conditions of the Klamath River green sturgeon. Transactions of the American Fisheries Society 135(1):151-163.

Van Parijs, S. M., and P. J. Corkeron. 2001. Boat traffic affects the acoustic behaviour of Pacific humpback dolphins, Sousa chinensis. Journal of the Marine Biological Association of the UK 81(3):6.

Van Waerebeek, K., and coauthors. 2007. Vessel collisions with small cetaceans worldwide and with large whales in the Southern Hemisphere, an initial assessment. Latin American Journal of Aquatic Mammals 6(1):43-69.

Vancouver Fraser Port Authority. 2017. Port of Vancouver Statistics Overview 2016. Decision Support Services, Vancouver, BC.

Vanderlaan, A. S., and C. T. Taggart. 2007. Vessel collisions with whales: The probability of lethal injury based on vessel speed. Marine Mammal Science 23(1):144-156.

Vanderlaan, A. S. M., C. T. Taggart, A. R. Serdynska, R. D. Kenney, and M. W. Brown. 2008. Reducing the risk of lethal encounters: Vessels and right whales in the Bay of Fundy and on the Scotian Shelf. Endangered Species Research 4(3):283-283.

Vargo, S., P. Lutz, D. Odell, E. V. Vleet, and G. Bossart. 1986. Study of the effects of oil on marine turtles. U.S. Department of the Interior, Minerals Management Service, Vienna, Virginia.

Veirs, S., V. Veirs, and J. D. Wood. 2016. Ship noise extends to frequencies used for echolocation by endangered killer whales. PeerJ 4:e1657-e1657.

Vilela, R., U. Pena, R. Esteban, and R. Koemans. 2016. Bayesian spatial modeling of cetacean sightings during a seismic acquisition survey. Marine Pollution Bulletin 109(1):512–520.

Villegas-Amtmann, S., L. K. Schwarz, G. Gailey, O. Sychenko, and D. P. Costa. 2017. East or west: The energetic cost of being a gray whale and the consequence of losing energy to disturbance. Endangered Species Research 34:167–183.

Villegas-Amtmann, S., L. K. Schwarz, J. L. Sumich, and D. P. Costa. 2015. A bioenergetics model to evaluate demographic consequences of disturbance in marine mammals applied to gray whales. Ecosphere 6(10).

Visser, F., and coauthors. 2016. Disturbance-specific social responses in long-finned pilot whales, *Globicephala melas*. Scientific Reports 6:28641.

Vladimirov, V. A., and coauthors. 2006a. Results of 2005 western gray whale studies in coastal waters off northeastern Sakhalin. Unpublished paper to the IWC Scientific Committee. 12 pp. St Kitts and Nevis, West Indies, June (SC/58/BRG28).

Vladimirov, V. A., and coauthors. 2006b. Distribution and abundance of western gray whales off the northeast coast of Sakhalin Island (Russia), 2005. Unpublished paper to the IWC Scientific Committee. 12 pp. St Kitts and Nevis, West Indies, June (SC/58/BRG29).

Vladimirov, V. A., and coauthors. 2005. Distribution and abundance of western gray whales off the northeast coast of Sakhalin Island (Russia), 2004. Unpublished paper to the IWC Scientific Committee. 6 pp. Ulsan, Korea, June (SC/57/BRG23).

Vladimirov, V. A., and coauthors. 2008. Distribution and abundance of western gray whales off the northeast coast of Sakhalin Island (Russia), 2007. IWC Scientific Committee, Santiago, Chile.

Vladimirov, V. A., S. P. Starodymov, A. G. Afanasyev-Grigoryev, and V. V. Vertyankin. 2009. Distribution and abundance of western gray whales off the northeast coast of Sakhalin Island, Russia, 2008. Unpublished paper to the IWC Scientific Committee, Madeira, Portugal.

Vladimirov, V. A., S. P. Starodymov, M. S. Kornienko, and J. E. Muir. 2010. Distribution and abundance of western gray whales in the waters off northeast Sakhalin Island, Russia, 2004-2009. Unpublished paper to the IWC Scientific Committee, Agadir, Morocco.

Vogel, D. A., K. R. Marine, and J. G. Smith. 1988. Fish passage action program for Red Bluff Diversion Dam: Final report on fishery investigations. US Fish and Wildlife Service.

von Benda-Beckmann, A., and coauthors. 2016. Assessing the Effectiveness of Ramp-Up During Sonar Operations Using Exposure Models The Effects of Noise on Aquatic Life II (pp. 1197–1203). New York, NY: Springer.

von Benda-Beckmann, A. M., and coauthors. 2014. Modeling effectiveness of gradual increases in source level to mitigate effects of sonar on marine mammals. Conservation Biology 28(1):119-128.

Wada, S., and K.-I. Numachi. 1991. Allozyme analyses of genetic differentiation among the populations and species of the Balaenoptora. Report of the International Whaling Commission Special Issue 13:125-154.

Wade, P. M., and coauthors. 2006. Acoustic detection and satellite tracking leads to discovery of rare concentration of endangered North Pacific right whales. Biology Letters 2:417-419.

Wade, P. R. 2017. Estimates of abundance and migratory destination for North Pacific humpback whales in both summer feeding areas and winter mating and calving areas revision of estimates in SC/66b/IA21. Seattle, Washington.

Wade, P. R., J. W. Durban, J. M. Waite, A. N. Zerbini, and M. E. Dahlheim. 2003. Surveying killer whale abundance and distribution in the Gulf of Alaska and Aleutian Islands. NOAA.

Wade, P. R., and coauthors. 2011a. The world's smallest whale population? Biology Letters 7(1):83-85.

Wade, P. R., and coauthors. 2011b. Rare detections of North Pacific right whales in the Gulf of Alaska, with observations of their potential prey. Endangered Species Research 13(2):99-109.

Wainwright, T. C., M. W. Chilcote, and P. W. Lawson. 2008. Biological recovery criteria for the Oregon Coast coho salmon evolutionarily significant unit.

Wainwright, T. C., and L. A. Weitkamp. 2013. Effects of Climate Change on Oregon Coast Coho Salmon: Habitat and Life-Cycle Interactions. Northwest Science 87(3):219-242.

Waite, J. 2003. Cetacean Survey. National Marine Mammal Laboratory (NMML), Cetacean Assessment and Ecology Program, Quarterly Report. Available online at: http://www.afsc.noaa.gov/Quarterly/jas2003/divrptsNMML2.htm Accessed 5/30/08.

Walker, M. M., J. L. Kirschvink, G. Ahmed, and A. E. Dizon. 1992. Evidence that fin whales respond to the geomagnetic field during migration. Journal of Experimental Biology 171(1):67-78.

Walker, R. L., and J. S. Foott. 1993. Disease survey of Klamath River salmonid smolt populations, 1992. U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [150 Kb] at KRIS Website:46 p.

Walker, R. V., V. V. Sviridov, S. Urawa, and T. Azumpaya. 2007. Spatio-Temporal Variation in Vertical Distributions of Pacific Salmon in the Ocean. North Pacific Anadromous Fish Commission Bulletin 4:193-201.

Walker, W. A., and J. M. Coe. 1990. Survey of marine debris ingestion by odontocete cetaceans. Pages 747-774 *in* Second International Conference on Marine Debris, Honolulu, Hawaii.

Wallace, B. P., and coauthors. 2010. Regional management units for marine turtles: A novel framework for prioritizing conservation and research across multiple scales. PLoS ONE 5(12):e15465.

Wallace, B. P., S. S. Kilham, F. V. Paladino, and J. R. Spotila. 2006. Energy budget calculations indicate resource limitation in Eastern Pacific leatherback turtles. Marine Ecology Progress Series 318:263-270.

Wallace, B. P., and coauthors. 2007. Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia 152(1):37-47.

Wambiji, N., P. Gwada, E. Fondo, S. Mwangi, and M. K. Osore. 2007. Preliminary results from a baseline survey of the port of Mombasa: with focus on molluscs. 5th Western Indian Ocean Marine Science Association Scientific Symposium; Science, Policy and Management pressures and responses in the Western Indian Ocean region, Durban, South Africa.

Wang, W. X., and P. S. Rainbow. 2008. Comparative approaches to understand metal bioaccumulation in aquatic animals. Comparative Biochemistry and Physiology, Part C 148(4):315–323.

Ward, D. W. 1960. Recovery from high values of temporary threshold shift. Journal of the Acoustical Society of America, 32(4), 497–500. .

Ward, S. L. C., and J. W. 1943. The effects of rapid compression waves on animals submerged in water. Surgery, Gynecology and Obstetrics 77:403-412.

Waring, G. T. 1993. Spatial patterns of six cetaceans along a linear habitat. Tenth Biennial Conference on the Biology of Marine Mammals, 11-15 November Galveston TX. p.2. Symposium: Cetacean Habitats.

Waring, G. T., C. P. Fairfield, C. M. Ruhsam, and M. Sano. 1993. Sperm whales associated with Gulf Stream features off the north-eastern USA shelf. Fisheries Oceanography 2(2):101-105.

Waring, G. T., E. Josephson, C. P. Fairfield, and K. M.-F. (Eds). 2009. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2008. NOAA Technical Memorandum NMFS-NE-210. 440pp.

Waring, G. T., E. Josephson, C. P. Fairfield, and K. Maze-Foley. 2007. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2006. U.S. Department of Commerce, NOAA, NMFS.

Waring, G. T., R. M. Pace, J. M. Quintal, C. P. Fairfield, and K. Maze-Foley. 2004. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2003, Woods Hole, Massachusetts.

Warren, R. 2019. Actions taken in development of WDFW managed fishery season for 2018-2019 beneficial for Southern Resident killer whales.

Wartzok, D., A. N. Popper, J. Gordon, and J. Merrill. 2003. Factors affecting the responses of marine mammals to acoustic disturbance. Marine Technology Society Journal 37(4):15-Jun.

Washington Department of Fish and Wildlife. 2013. Threatened and Endangered Wildlife. Annual Report 2012. Washington Department of Fish and Wildlife Listing and Recovery Section, Diversity Division, Wildlife Program, Olympia, WA.

Washington Department of Fish and Wildlife (WDFW). 1993. 1992 Washington state salmon and steelhead stock inventory (SASSI) WDFW and Western Washington Treaty Indian Tribes, Olympia, Washington.

Washington State Department of Transportation. 2018. Washington State Ferries Federal Transit Administration Asset Management Plan, Olympia, WA.

Wasser, S. K., and coauthors. 2017. Population growth is limited by nutritional impacts on pregnancy success in endangered Southern Resident killer whales (Orcinus orca). PLoS ONE 12(6):e0179824.

Watkins, W. A. 1977. Acoustic behavior of sperm whales. Oceanus 20:50-58.

Watkins, W. A. 1981a. Activities and underwater sounds of fin whales. (Balaenoptera physalus). Scientific Reports of the Whales Research Institute Tokyo 33:83-118.

Watkins, W. A. 1981b. Radio tagging of finback whales - Iceland, June-July 1980. Woods Hole Oceanagraphic Institution.

Watkins, W. A. 1981c. Reaction of three species of whales, *Balaenoptera physalus*, *Megaptera novaeangliae*, and *Balaenoptera edeni* to implanted radio tags. Deep Sea Research Part I: Oceanographic Research Papers 28(6):589-599.

Watkins, W. A. 1985. Changes observed in the reaction of whales to human activities. National Marine Fisheries Service.

Watkins, W. A. 1986. Whale Reactions to Human Activities in Cape-Cod Waters. Marine Mammal Science 2(4):251-262.

Watkins, W. A., M. A. Daher, K. M. Fristrup, T. J. Howald, and G. N. Disciara. 1993. Sperm whales tagged with transponders and tracked underwater by sonar. Marine Mammal Science 9(1):55-67.

Watkins, W. A., K. E. Moore, and P. L. Tyack. 1985. Sperm whale acoustic behaviors in the southeast Caribbean. Cetology 49:1-15.

Watkins, W. A., and W. E. Schevill. 1975. Sperm whales (Physeter catodon) react to pingers. Deep Sea Research and Oceanogaphic Abstracts 22(3):123-129 +1pl.

Watkins, W. A., and W. E. Schevill. 1977. Spatial distribution of *Physeter catodon* (sperm whales) underwater. Deep Sea Research 24(7):693-699.

Watkins, W. A., P. Tyack, K. E. Moore, and J. E. Bird. 1987. The 20-Hz signals of finback whales (*Balaenoptera physalus*). Journal of the Acoustical Society of America 82(6):1901-1912.

Watwood, S., M. Fagain, A. D'Amico, and T. Jefferson. 2012. Cruise Report, Marine Species Monitoring and Lookout Effectiveness Study, Koa Kai, November 2011, Hawaii Range Complex. Prepared for Commander, U.S. Pacific Fleet.

Watwood, S. L., P. J. O. Miller, M. Johnson, P. T. Madsen, and P. L. Tyack. 2006. Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). Journal of Animal Ecology 75:814-825.

WDFW. 2017. Draft conservation plan for reducing the impact of selected fisheries on ESA listed species in Puget Sound, with an emphasis on bocaccio and yelloweye rockfish. Prepared for the National Marine Fisheries Service by the Washington Department of Fish and Wildlife.

WDFW, and Northwest Indian Fisheries Commission. 1990. Puget Sound Salmon Management Periods and their Derivations.

WDFW, and ODFW. 2001. Washington and Oregon eulachon management plan. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

WDFW, N. a. 2018. Southern Resident Killer Whale Priority Chinook Stocks Report. NOAA Fisheries West Coast Region and Washington Department of Fish and Wildlife.

Weaver, A. 2015. Sex difference in bottlenose dolphin sightings during a long-term bridge construction project. Animal Behavior and Cognition 2(1):1–13.

Weber, D. S., B. S. Stewart, and N. Lehman. 2004. Genetic consequences of a severe population bottleneck in the Guadalupe fur seal (*Arctocephalus townsendi*). Journal of Heredity 95(2):144-153.

Weilgart, L., and H. Whitehead. 1993. Coda communication by sperm whales (*Physeter macrocephalus*) off the Galápagos Islands. Canadian Journal of Zoology 71(4):744–752.

Weilgart, L. S., and H. Whitehead. 1997. Group-specific dialects and geographical variation in coda repertoire in South Pacific sperm whales. Behavioral Ecology and Sociobiology 40(5):277-285.

Weinrich, M. T., and coauthors. 1992. Behavioral reactions of humpback whales *Megaptera novaeangliae* to biopsy procedures. Fishery Bulletin 90(3):588-598.

Weir, C. R. 2008. Overt responses of humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter macrocephalus*), and Atlantic spotted dolphins (*Stenella frontalis*) to seismic exploration off Angola. Aquatic Mammals 34(1):71–83.

Weir, C. R., A. Frantzis, P. Alexiadou, and J. C. Goold. 2007. The burst-pulse nature of 'squeal' sounds emitted by sperm whales (*Physeter macrocephalus*). Journal of the Marine Biological Association of the U.K. 87(1):39-46.

Weirathmueller, M. J., W. S. D. Wilcock, and D. C. Soule. 2013. Source levels of fin whale 20Hz pulses measured in the Northeast Pacific Ocean. Journal of the Acoustical Society of America 133(2):741-749.

Weirathmueller, M. J. W. S. D. W. D. C. S. 2013. Source levels of fin whale 20Hz pulses measured in the Northeast Pacific Ocean. Journal of the Acoustical Society of America 133(2):741-749.

Weis, L. J. 2004. The effects of San Juan County, Washington, marine protected areas on larval rockfish production. University of Washington, Seattle, WA.

Weitkamp, L. A. 2010. Marine Distributions of Chinook Salmon from the West Coast of North America Determined by Coded Wire Tag Recoveries. Transactions of the American Fisheries Society 139(1):147-170.

Weitkamp, L. A., and K. Neely. 2002. Coho salmon (Oncorhynchus kisutch) ocean migration patterns: insight from marine coded-wire tag recoveries. Can. J. Fish. Aquat. Sci. 59.

Weller, D. W., and coauthors. 2013. Report of the National Marine Fisheries Service gray whale stock identification workshop. National Marine Fisheries Service Gray Whale Stock Identification Workshop. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Weller, D. W., and coauthors. 2008. A photographic match of a western gray whale between Sakhalin Island, Russia, and Honshu, Japan: The first link between the feeding ground and a migratory corridor. Journal of Cetacean Research and Management 10(1):89-91.

Weller, D. W., and coauthors. 2007. Status of western gray whales off northeastern Sakhalin Island, Russia. Unpublished paper to the IWC Scientific Committee. 11 pp. Anchorage, AK, May (SC/59/BRG19).

Weller, D. W., and coauthors. 2004. Status of western gray whales off northeastern Sakhalin Island, Russia, in 2003. IWC Scientific Committee, Sorrento, Italy.

Weller, D. W., and coauthors. 2006. Status of western gray whales off northeastern Sakhalin Island, Russia, in 2005. Unpublished paper to the IWC Scientific Committee. 10 pp. St Kitts and Nevis, West Indies, June (SC/58/BRG3).

Weller, D. W., and coauthors. 2003. Summer sightings of western gray whales in the Okhotsk and western Bering Seas. International Whaling Commission Scientific Committee, Berlin.

Weller, D. W., and coauthors. 2012a. Movements of gray whales between the western and eastern North Pacific. Endangered Species Research 18:193-199.

Weller, D. W., and coauthors. 2012b. Movements of gray whales between the western and eastern North Pacific. Endangered Species Research 18(3):193-199.

Weller, D. W., and coauthors. 1999. Gray whales (*Eschrichtius robustus*) off Sakhalin Island, Russia: Seasonal and annual patterns of occurrence. Marine Mammal Science 15(4):1208-1227.

Weller, D. W., and coauthors. 1996. Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. Marine Mammal Science 12(4):588-593.

Wells, J. V., and M. E. Richmond. 1995. Populations, metapopulations, and species populations: What are they and who should care? Wildlife Society Bulletin 23(3):458-462.

Wells, R. S., and coauthors. 2008. Consequences of injuries on survival and reproduction of common bottlenose dolphins (Tursiops truncatus) along the west coast of Florida. Marine Mammal Science 24(4):774-794.

Wensveen, P., and coauthors. 2017. Lack of behavioural responses of humpback whales (Megaptera novaeangliae) indicate limited effectiveness of sonar mitigation. Journal of Experimental Biology 220:4150-4161.

Wensveen, P. J., and coauthors. 2015. How effectively do horizontal and vertical response strategies of long-finned pilot whales reduce sound exposure from naval sonar? Marine Environmental Research 106:68-81.

Wensveen, P. J., and coauthors. 2019. Northern bottlenose whales in a pristine environment respond strongly to close and distant navy sonar signals. Proceedings of the Royal Society of London B: Biological Science 286(1899):20182592.

West, J. E., S. M. O'Neill, J. Lanksbury, G. M. Ylitalo, and S. Redman. 2011. Current conditions, time trends and recovery targets for toxic contaminants in Puget Sound fish: the Toxics in Fish Dashboard Indicator.

West, K. L., and coauthors. 2015. Coinfection and vertical transmission of Brucella and Morbillivirus in a neonatal sperm whale (Physeter macrocephalus) in Hawaii, USA. Journal of Wildlife Diseases 51(1):227-232.

Westrheim, S. J., and W. R. Harling. 1975. Age-length relationships for 26 scorpaenids in the northeast Pacific Ocean. Fisheries and Marine Service Research Division, Nanaimo, British Columbia.

Wever, E. G., and J. A. Vernon. 1956. The sensitivity of the turtle's ear as shown by its electrical potentials. Proceedings of the National Academies of Science 42.

Whale Museum. 2012. The Whale Museum.

Whitehead, H. 2002. Sperm whale *Physeter macrocephalus*. Pages 1165-1172 *in* W. F. Perrin, B. Würsig, and J. G. M. Thewissen, editors. Encyclopedia of Marine Mammals. Academic Press, San Diego, California.

Whitehead, H. 2003a. Society and culture in the deep and open ocean: The sperm whale and other cetaceans. Pages 616 *in* F. B. M. d. Waal, and P. L. Tyack, editors. Animal Social Complexity: Intelligence, Culture, and Individualized Societies. Harvard University Press.

Whitehead, H. 2003b. Sperm Whales: Social Evolution in the Ocean. University of Chicago Press.

Whitehead, H. 2008. Social and cultural evolution in the ocean: Convergences and contrasts with terrestrial systems. The Deep Structure of Biology: Is Convergence Sufficiently Ubiquitous to Give a Directional Signal? p.143-160. Simon conway Morris (ed.). Templeton Foundation Press, West Conshohocken, Pennsylvania. ISBN 978-1-59947-138-9. 256pp.

Whitehead, H. 2009. Sperm whale: Physeter macrocephalus. Pages 1091-1097 *in* W. F. P. B. W. J. G. M. Thewissen, editor. Encyclopedia of Marine Mammals, Second edition. Academic Press, San Diego.

Whitehead, H., J. Christal, and S. Dufault. 1997. Past and distant whaling and the rapid decline of sperm whales off the Galapagos Islands. (Physeter macrocephalus). Conservation Biology 11(6):1387-1396.

Whitehead, H., and L. Weilgart. 1991. Patterns of visually observable behaviour and vocalizations in groups of female sperm whales. Behaviour 118(3/4):275-295.

Wieland, M., A. Jones, and S. C. P. Renn. 2010. Changing durations of southern resident killer whale (*Orcinus orca*) discrete calls between two periods spanning 28 years. Marine Mammal Science 26(1):195-201.

Wiggins, S., A. Krumpel, L. Dorman, J. Hildebrand, and S. Baumann-Pickering. 2019. Seal Bomb Sound Source Characterization. Marine Physical Laboratory of the Scripps Institution of Oceanography, La Jolla, California.

Wiggins, S. M., and coauthors. 2017a. Summary of Anthropogenic Sound in the Gulf of Alaska and Northwest Coast (MPL Technical Memorandum #611). Marine Physical Laboratory, La Jolla, California.

Wiggins, S. M., and coauthors. 2017b. Summary of Ambient and Anthropogenic Sound in the Gulf of Alaska and Northwest Coast. Marine Physical Laboratory, La Jolla, CA.

Wiggins, S. M., E. M. Oleson, M. A. McDonald, and J. A. Hildebrand. 2005. Blue whale (*Balaenoptera musculus*) diel call patterns offshore of southern California. Aquatic Mammals 31(2):161–168.

Wiggins, S. M., and coauthors. 2020. Comparison of Fisheries and Naval Explosions in SOCAL Range Complex. University of California San Diego, Scripps Institution of Oceanography, Marine Physical Laboratory, La Jolla, California.

Wilcove, D. S., and L. Y. Chen. 1998. Management costs for endangered species. Conservation Biology 12(6):1405-1407.

Wilcox, C., and coauthors. 2015. Understanding the sources and effects of abandoned, lost, and discarded fishing gear on marine turtles in northern Australia. Conservation Biology 29(1):198-206.

Wildlife, W. D. o. F. a., and P. N. P. T. Tribes. 2000. Summer Chum Salmon Conservation Initiative: An Implementation Plan to Recover Summer Chum in the Hood Canal and Strait of Juan de Fuca Region.

Wiles, G. J. 2015. Periodic Status Review for the Steller Sea Lion. Washington Department of Fish and Wildlife, Olympia, Washington.

Wiles, G. J. 2016. Periodic Status Review for the Killer Whale. Washington Department of Fish and Wildlife, Olympia, WA.

Wiley, M. L., J. B. Gaspin, and J. F. Goertner. 1981. Effects of underwater explosions on fish with a dynamical model to predict fishkill. Ocean Science and Engineering 6(2):223–284.

Wilkinson, K. 2013. Personal communication between Kristin Wilkinson (NOAA Fisheries, Protected

Resources Division Marine Mammal Stranding Specialist) and Andrea Balla-Holden (Navy) in July

2013, regarding Stranding Database. A. B.-H. (Navy), editor.

Williams Jr, E. H., and coauthors. 1994. An epizootic of cutaneous fibropapillomas in green turtles Chelonia mydas of the Caribbean: part of a panzootic? Journal of Aquatic Animal Health 6(1):70-78.

Williams, R., and E. Ashe. 2007. Killer whale evasive tactics vary with boat number. (Orcinus orca). Journal of Zoology 272(4):390-397.

Williams, R., E. Ashe, and P. D. O'hara. 2011a. Marine mammals and debris in coastal waters of British Columbia, Canada. Marine Pollution Bulletin 62(6):1303-1316.

Williams, R., D. E. Bain, J. K. B. Ford, and A. W. Trites. 2002a. Behavioural responses of male killer whales to a 'leapfrogging' vessel. Journal of Cetacean Research and Management 4(3):305-310.

Williams, R., D. E. Bain, J. K. B. Ford, and A. W. Trites. 2002b. Behavioural responses of male killer whales to a 'leapfrogging' vessel. Journal of Cetacean Research and Management 4(3):305-310.

Williams, R., D. E. Bain, J. C. Smith, and D. Lusseau. 2009a. Effects of vessels on behaviour patterns of individual southern resident killer whales *Orcinus orca*. Endangered Species Research 6:199-209.

Williams, R., C. W. Clark, D. Ponirakis, and E. Ashe. 2014a. Acoustic quality of critical habitats for three threatened whale populations. Animal Conservation 17(2):174-185.

Williams, R., C. Erbe, E. Ashe, A. Beerman, and J. Smith. 2014b. Severity of killer whale behavioral responses to ship noise: A dose-response study. Marine Pollution Bulletin 79(1-2):254-260.

Williams, R., and coauthors. 2011b. Competing conservation objectives for predators and prey: estimating killer whale prey requirements for Chinook salmon. PLoS ONE 6(11).

Williams, R., D. Lusseau, and P. S. Hammond. 2006a. Estimating relative energetic costs of human disturbance to killer whales (*Orcinus orca*). Biological Conservation 133:301-311.

Williams, R., D. Lusseau, and P. S. Hammond. 2009b. The role of social aggregations and protected areas in killer whale conservation: The mixed blessing of critical habitat. Biological Conservation 142:709-719.

Williams, R., and D. P. Noren. 2009. Swimming speed, respiration rate, and estimated cost of transport in adult killer whales. Marine Mammal Science 25(2):327-350.

Williams, R., and L. Thomas. 2007. Distribution and abundance of marine mammals in the coastal waters of British Columbia, Canada. Journal of Cetacean Research and Management 9(1):15–28.

Williams, R., A. W. Trites, and D. E. Bain. 2002c. Behavioural responses of killer whales (*Orcinus orca*) to whale-watching boats: opportunistic observations and experimental approaches. Journal of Zoology 256:255-270.

Williams, R. M., A. W. Trites, and D. E. Bain. 2002d. Behavioral responses of killer whales (*Orcinus orca*) to whale-watching boats: Opportunistic observations and experimental approaches. Journal of Zoology 256(2):255-270.

Williams, T. H., and coauthors. 2006b. Historical population structure of coho salmon in the Southern Oregon/Northern California coasts evolutionarily significant unit. Pages 71 p *in* U.S. Department of Commerce,.

Williams, T. H., S. T. Lindley, B. C. Spence, and D. A. Boughton. 2011c. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. National Marine Fisheries Service, Southwest Fisheries Science Center, Fisheries Ecology Division, editor, Santa Cruz, California.

Williams, T. H., and coauthors. 2008. Framework for assessing viability of threatened coho salmon in the Southern Oregon/Northern California Coast Evolutionarily Significant Unit. NOAA Technical Memorandum NMFS-SWFSC 432.

Williams, T. M., and coauthors. 2017. Swimming and diving energetics in dolphins: a stroke-by-stroke analysis for predicting the cost of flight responses in wild odontocetes. Journal of Experimental Biology 220(6):1135–1145.

Willis-Norton, E., and coauthors. 2015. Climate change impacts on leatherback turtle pelagic habitat in the Southeast Pacific. Deep Sea Research Part II: Topical Studies in Oceanography 113:260-267.

Wilson, J. A., and R. S. McKinley. 2004. Distribution, habitat and movements. Pages 40-69 in: G.T.O. LeBreton, F.W.H. Beamish, and R.S. McKinley, eds. Sturgeon and paddlefish of North America. Kluwer Academic Publishers.

Winn, H. E., P. J. Perkins, and T. C. Poulter. 1970. Sounds of the humpback whale. Proceedings of the 7th Annual Conference on Biological Sonar and Diving Mammals, Stanford Research Institute Menlo Park CA. p.39-52.

Winn, H. E., and N. E. Reichley. 1985. Humpback whale - *Megaptera novaeangliae*. Pages 241-274 *in* S. H. Ridgway, and S. R. Harrison, editors. Handbook of Marine Mammals: Vol. 3 The Sirenians and Baleen Whales. Academic Press Ltd., London.

Wise, J. P., Sr., and coauthors. 2009. A global assessment of chromium pollution using sperm whales (Physeter macrocephalus) as an indicator species. Chemosphere 75(11):1461-1467.

Witteveen, B. H., R. J. Foy, K. M. Wynne, and Y. Tremblay. 2008. Investigation of foraging habits and prey selection by humpback whales (*Megaptera novaeangliae*) using acoustic tags and concurrent fish surveys. Marine Mammal Science 24(3):516-534.

Witteveen, B. H., G. A. J. Worthy, K. M. Wynne, and J. D. Roth. 2009. Population structure of North Pacific humpback whales on their feeding grounds revealed by stable carbon and nitrogen isotope ratios. Marine Ecology Progress Series 379:299-310.

Witzell, W. N., A. A. Geis, J. R. Schmid, and T. Wibbels. 2005. Sex ratio of immature Kemp's ridley turtles (*Lepidochelys kempii*) from Gullivan Bay, Ten Thousand Islands, south-west Florida. Journal of the Marine Biological Association of the U.K. 85:205-208.

Woodbury, D., and S. Ralston. 1991. Interannual variation in growth rates and back-calculated birth-date distributions of pelagic juvenile rockfishes (*Sebastes* spp.) off the central California coast. Fishery Bulletin 89(3):523-533.

Woodworth-Jefcoats, P. A., J. J. Polovina, and J. C. Drazen. 2017. Climate change is projected to reduce carrying capacity and redistribute species richness in North Pacific pelagic marine ecosystems. Global Change Biology 23:1000-1008.

Wright, A. J. 2005. Lunar cycles and sperm whales (*Physeter macrocephalus*) strandings on the North Atlantic coastlines of the British Isles and Eastern Canada. Marine Mammal Science 21(1):145-149.

Wright, A. J., and coauthors. 2007. Anthropogenic noise as a stressor in animals: A multidisciplinary perspective. International Journal of Comparative Psychology 201(2-3):250-273.

Wright, B., T. Murtagh, R. Brown, and S. Riemer. 2017a. Willamette Falls Pinniped Monitoring Project, 2017. Oregon Department of Fish and Wildlife, Oregon City, Oregon.

Wright, B. M., and coauthors. 2017b. Fine-scale foraging movements by fish-eating killer whales (*Orcinus orca*) relate to the vertical distributions and escape responses of salmonid prey (*Oncorhynchus* spp.). Movement Ecology 5(3):1–18.

Wright, D. G. 1982. A Discussion Paper on the Effects of Explosives on Fish and Marine Mammals in the Waters of the Northwest Territories. Western Region Department of Fisheries and Oceans, Winnipeg, Canada.

Wursig, B., S. K. Lynn, T. A. Jefferson, and K. D. Mullin. 1998. Behaviour of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals 24(1):41-50.

Wydoski, R., and R. Whitney. 1979. Inland fishes of Washington. University of Washington Press.

Wysocki, L. E., J. P. Dittami, and F. Ladich. 2006. Ship noise and cortisol secretion in European freshwater fishes. Biological Conservation 128(4):501-508.

Yablokov, A. V. 2000. Consequences and perspectives of whaling (instead of a preface). Pages 6-10 *in* Soviet Whaling Data (1949-1979). Center for Russian Environmental Policy Marine Mammal Council, Moscow.

Yablokov, A. V., and L. S. Bogoslovskaya. 1984. A review of Russian research on the biology and commercial whaling of the gray whale. Pages 465-486 *in* M. L. Jones, S. L. Swartz, and S. Leatherwood, editors. The Gray Whale, *Eschrichtius robustus*. Academic Press, New York.

Yablokov, A. V., V. A. Zemsky, Y. A. Mikhalev, V. V. Tormosov, and A. A. Berzin. 1998. Data on Soviet whaling in the Antarctic in 1947–1972 (population aspects). Russian Journal of Ecology 29:38–42.

Yagla, J., and R. Stiegler. 2003a. Gun blast noise transmission across the air-sea interface. 5th European Conference on Noise Control. Congress of the Acoustical Society of Italy, Naples, Italy.

Yagla, J., and R. Stiegler. 2003b. Gun blast noise transmission across the air-sea interface. Pages 9-Jan *in* Euronoise, Naples.

Yakovlev, Y. M., and O. Y. Tyurneva. 2004. Photo-identification of the western gray whale (*Eschrichtius robustus*) on the northeastern Sakhalin shelf, Russia, 2002-2003. International Whaling Commission Scientific Committee, Sorrento, Italy.

Yakovlev, Y. M., and O. Y. Tyurneva. 2005. A note on photo-identification of the western gray whale (*Eschrictius robustus*) on the northeastern Sakhalin shelf, Russia, 2002-2004. International Whaling Commission Scientific Committee, Ulsan, Korea.

Yamanaka, K. L., and A. R. Kronlund. 1997. Inshore rockfish stock assessment for the west coast of Canada in 1996 and recommended yields for 1997.

Yamanaka, K. L., and coauthors. 2006. A review of yelloweye rockfish *Sebastes ruberrimus* along the Pacific coast of Canada: Biology, distribution, and abundance trends

Yazvenko, S. B., and coauthors. 2007. Feeding of western gray whales during a seismic survey near Sakhalin Island, Russia. Environmental Monitoring and Assessment 134(1–3):93–106.

Yelverton, J. T., and D. R. Richmond. 1981. Underwater Explosion Damage Risk Criteria for Fish, Birds, and Mammals. Pages S84 *in* 102nd Meeting of the Acoustical Society of America. Miami Beach, FL. Journal of the Acoustical Society of America.

Yelverton, J. T., D. R. Richmond, E. R. Fletcher, and R. K. Jones. 1973. Safe distances from underwater explosions for mammals and birds. Defense Nuclear Agency.

Yelverton, J. T., D. R. Richmond, W. Hicks, K. Saunders, and E. R. Fletcher. 1975. The Relationship between Fish Size and Their Response to Underwater Blast. Lovelace Foundation for Medical Education and Research, DNA 3677T, Washington, DC.

Ylitalo, G. M., and coauthors. 2001. Influence of life-history parameters on organochlorine concentrations in free-ranging killer whales (*Orcinus orca*) from Prince William Sound, AK. Science of the Total Environment 281:183-203.

Ylitalo, G. M., and coauthors. 2005. The role of organochlorines in cancer-associated mortality in California sea lions (*Zalophus californianus*). Marine Pollution Bulletin 50:30-39.

Yochem, P. K., and S. Leatherwood. 1985. Blue whale *Balaenoptera musculus* (Linnaeus, 1758). Pages 193-240 *in* S. H. Ridgway, and R. Harrison, editors. Handbook of Marine Mammals, vol. 3: The Sirenians and Baleen Whales. Academic Press, London.

Yoshiyama, R. M., F. W. Fisher, and P. B. Moyle. 1998. Historical abundance and decline of chinook salmon in the Central Valley region of California. North American Journal of Fisheries Management 18(3):487-521.

Zabel, R. W. 2015. Memorandum to Donna Weiting: Estimation of Percentages for Listed Pacific Salmon and Steelhead Smolts Arriving at Various Locations in the Columbia River Basin in 2015.

Zabel, R. W. 2017a. Memorandum for Chris Yates: Estimation of Percentages for Listed Pacific Salmon and Steelhead Smolts Arriving at Various Locations in the Columbia River Basin in 2017.

Zabel, R. W. 2017b. Memorandum for Christopher E. Yates: Update, Corrected Estimation of Percentages for Listed Pacific Salmon and Steelhead Smolts Arriving at Various Locations in the Columbia River Basin in 2016.

Zabel, R. W. 2018. Memorandum for Chris Yates: Estimation of Percentages for Listed Pacific Salmon and Steelhead Smolts Arriving at Various Locations in the Columbia River Basin in 2018.

Zabel, R. W. 2020. Memorandum for Chris Yates: Estimation of Percentages for Listed Pacific Salmon and Steelhead Smolts Arriving at Various Locations in the Columbia River Basin in 2019. Northwest Fisheries Science Center.

Zedonis, P. A. 1992. The biology of steelhead (*Onchorynchus mykiss*) in the Mattole River estuary/lagoon, California. Humboldt State University, Arcata, CA.

Zerbini, A., A. S. Kennedy, B. K. Rone, C. L. Berchok, and P. J. Clapham. 2010. Habitat use of North Pacific right whales in the Bering Sea during summer as revealed by sighting and telemetry data. Pages 153 *in* Alaska Marine Science Symposium, Anchorage, Alaska.

Zerbini, A. N., J. M. Waite, J. L. Laake, and P. R. Wade. 2006. Abundance, trends and distribution of baleen whales off Western Alaska and the central Aleutian Islands. Deep-Sea Research I 53:1772-1790.

Zimmer, W. M. X., and P. L. Tyack. 2007. Repetitive shallow dives pose decompression risk in deep-diving beaked whales. Marine Mammal Science 23(4):888-925.

Zoidis, A. M., and coauthors. 2008. Vocalizations produced by humpback whale (*Megaptera novaeangliae*) calves recorded in Hawaii. The Journal of the Acoustical Society of America 123(3):1737-1746.

Zorn, H. M., J. H. Churnside, and C. W. Oliver. 2000. Laser safety thresholds for cetaceans and pinnipeds. Marine Mammal Science 16(1):186-200.