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

Title:	Biological and Conference Opinion on (1) U.S. Navy, U.S. Coast Guard, U.S. Army, and U.S. Air Force (Action Proponents) Hawaii-California Training, Testing (HCTT) and other Military Readiness Activities 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 Action Proponents to "Take" Marine Mammals Incidental to HCTT activities through December 2032
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Action Agencies:	U.S. Navy; U.S. Coast Guard; U.S. Army; U.S. Air Force; and Permits and Conservation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Publisher:	Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Approved:	DAMON RANDALL.KIMBERLY.BETH.1365821093 Digitally signed by DAMON RANDALL.KIMBERLY.BETH.1365821093 Date: 2025.10.17 09:23:10 -04'00' Kim Damon-Randall Director, Office of Protected Resources
Date:	10/17/2025
Consultation Tracking number:	OPR-2022-03647
Digital Object Identifier (DOI):	https://doi.org/10.25923/mw24-km06

This page left blank intentionally

TABLE OF CONTENTS

Page

1 Introduction 15
1.1 Background 17
1.2 Consultation History 19
1.3 Analytical Approach 22

2 Description of the Proposed Action 24
2.1 Primary Mission Areas 26
2.2 Proposed Military Readiness Activities 29
2.2.1 Training Activities 29
2.2.2 Testing Activities 30
2.2.3 Modernization and Sustainment of Ranges 31
2.3 Foreign Military Activities 36
2.4 Standard Operating Procedures and Mitigation Measures 38
2.4.1 Standard Operating Procedures 38
2.4.2 Mitigation Measures 41
2.5 Stressors Resulting from the Components of the Proposed Action 65
2.5.1 Acoustic Stressors 65
2.5.2 Explosive Stressors 75
2.5.3 Energy Stressors 77
2.5.4 Physical Disturbance and Strike Stressors 79
2.5.5 Entanglement Stressors 84
2.5.6 Ingestion Stressors 86
2.6 MMPA Regulations and Issuance of a Letter of Authorization 87

3 Action Area 88

4 Species and Designated Critical Habitat that May be Affected 95
4.1 May Affect, Not Likely to be Adversely Affected 97
4.1.1 Stressors Not Likely to Adversely Affect ESA-listed Species 98
4.1.2 Species Not Likely to be Adversely Affected 151
4.1.3 Critical Habitat Not Likely to be Adversely Affected 168
4.2 Status of the Species and Critical Habitat Likely to be Adversely Affected 193
4.2.1 Blue Whale 194
4.2.2 Fin Whale 199
4.2.3 Gray Whale – Western North Pacific Population 204
4.2.4 Humpback Whale – Central America and Mexico DPSs 207
4.2.5 Sei Whale 213
4.2.6 Sperm Whale 217
4.2.7 False Killer Whale – Main Hawaiian Islands Insular DPS 221
4.2.8 Guadalupe Fur Seal 229

4.2.9 Hawaiian Monk Seal 233
4.2.10 Green Sea Turtle – Central North Pacific and East Pacific DPSs 235
4.2.11 Hawksbill Turtle 242
4.2.12 Leatherback Turtle 246
4.2.13 Loggerhead Turtle – North Pacific Ocean DPS 251
4.2.14 Olive Ridley Turtle – Mexico's Pacific Coast Breeding Colonies and All Other Areas 255
4.2.15 Giant Manta Ray 259
4.2.16 Oceanic Whitetip Shark 263
4.2.17 Scalloped Hammerhead Shark – Eastern Pacific DPS 267
4.2.18 Steelhead 270
4.2.19 Steelhead – Southern California DPS 271
4.2.20 Steelhead – California Central Valley DPS 273
4.2.21 Steelhead – Central California Coast DPS 276
4.2.22 Steelhead – Northern California DPS 278
4.2.23 Steelhead – South-Central California Coast DPS 280
4.2.24 Chinook Salmon 283
4.2.25 Chinook Salmon – California Coastal ESU 283
4.2.26 Chinook Salmon – Central Valley Spring-Run ESU 286
4.2.27 Chinook Salmon – Sacramento River Winter-Run ESU 289
4.2.28 Coho Salmon 292
4.2.29 Coho Salmon - Central California Coast ESU 292
4.2.30 Coho Salmon - Oregon Coast ESU 295
4.2.31 Coho Salmon - Southern Oregon and Northern California Coasts ESU 298
4.2.32 Eulachon – Southern DPS 300

5 Environmental Baseline 303
5.1 Large-Scale Changes in Environmental Conditions 303
5.1.1 Acidification 303
5.1.2 Sea Level and Freshwater 304
5.1.3 Range Shift 306
5.1.4 Turtles - Nesting 307
5.1.5 Salmon and Steelhead 308
5.1.6 Sharks and Rays 309
5.1.7 Invasives, Disease, Vulnerable Populations 309
5.1.8 Summary 309
5.2 Oceanic Temperature Regimes 310
5.3 Sound 312
5.3.1 Vessel Sound and Commercial Shipping 314
5.3.2 Acoustic Geophysical Surveys 316
5.3.3 Military Training and Testing Activities 319

5.3.4 Active Sonar 321
5.3.5 Pile Driving and Construction Sound 322
5.3.6 Offshore Industry Operations 322
5.3.7 Acoustic Deterrents 323
5.4 Dredging 323
5.5 Pollution 324
5.5.1 Contaminants and Pesticides 324
5.5.2 Oil spills 327
5.5.3 Nutrient Loading and Algal Blooms 329
5.5.4 Marine Debris 331
5.6 Whaling 334
5.7 Directed Harvest of Sea Turtles 334
5.8 Human-Caused Mortality of Pinnipeds 336
5.9 Unusual Mortality Events 336
5.10 Fisheries Catch and Bycatch 339
5.10.1 Sea Turtles 339
5.10.2 Marine Mammal Fishery Interactions 342
5.10.3 ESA-Listed Salmonids and Sturgeon 346
5.10.4 ESA-listed Elasmobranchs 347
5.11 Aquaculture 349
5.12 Ongoing Military Training and Testing Activities in the Action Area 351
5.12.1 Summary of Hawaii-Southern California Training and Testing Phase III Biological Opinion 351
5.12.2 Surveillance Towed Array Sensor System Low Frequency Active Sonar 352
5.12.3 Point Mugu Sea Range 353
5.13 Vessel Approach 353
5.14 Vessel Strike 356
5.15 Invasive Species 359
5.16 Parasites and Diseases 361
5.17 Scientific Research and Permits 364
5.18 Impact of the Baseline on ESA-listed Resources 365

6 Analysis of Effects 366
6.1 Marine Mammals 367
6.1.1 Acoustic and Explosive Effects Analysis 367
6.1.2 Weapons Firing Noise: In-Air Effects to Hauled Out Pinnipeds 422
6.1.3 Vessel Strike Analysis 426
6.1.4 Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat 451
6.2 Sea Turtles 472
6.2.1 Acoustic and Explosive Effects Analysis 472
6.2.2 Vessel Strike Analysis 483

6.3 Fishes 494
6.3.1 Acoustic and Explosives Effects Analysis Approach 494
6.4 Summary of Effects Determinations 519

7 Cumulative Effects 522

8 Integration and Synthesis 522
8.1 Jeopardy Analysis 523
8.1.1 Blue Whale 523
8.1.2 Fin Whale 526
8.1.3 Gray Whale – Western North Pacific DPS 529
8.1.4 Humpback Whale – Mexico DPS 531
8.1.5 Humpback Whale – Central America DPS 533
8.1.6 Sei Whale 536
8.1.7 Sperm Whale 538
8.1.8 Guadalupe Fur Seal 541
8.1.9 Hawaiian Monk Seal 543
8.1.10 False Killer Whale – Main Hawaiian Islands Insular DPS 545
8.1.11 Green Sea Turtle - Central North Pacific and East Pacific DPSs 548
8.1.12 Hawksbill Sea Turtle 552
8.1.13 Leatherback Sea Turtle 554
8.1.14 Loggerhead Sea Turtle – North Pacific DPS 556
8.1.15 Olive Ridley Sea Turtle 558
8.1.16 Giant Manta Ray 560
8.1.17 Oceanic Whitetip Shark 562
8.1.18 Scalloped Hammerhead Shark – Eastern Pacific DPS 563
8.1.19 Steelhead – Southern California DPS 564
8.1.20 Steelhead – California Central Valley DPS 565
8.1.21 Steelhead – Central California Coast DPS 567
8.1.22 Steelhead – Northern California DPS 568
8.1.23 Steelhead – South-Central California Coast DPS 569
8.1.24 Chinook Salmon – California Coastal ESU 570
8.1.25 Chinook Salmon – Central Valley Spring-Run ESU 572
8.1.26 Chinook Salmon – Sacramento River Winter-Run ESU 573
8.1.27 Coho Salmon - Central California Coast ESU 574
8.1.28 Coho Salmon - Oregon Coast ESU 576
8.1.29 Coho Salmon - Southern Oregon and Northern California Coasts ESU 577
8.1.30 Eulachon – Southern DPS 578
8.2 Destruction and Adverse Modification Analysis 580
8.2.1 Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat 580

9 Conclusion 582

10 Incidental Take Statement 583
10.1 Amount or Extent of Take 584
10.1.1 Marine Mammals and Sea Turtles 584
10.1.2 Fishes 588
10.1.3 Activity Levels as Indicators of Take Resulting from Acoustic Stressors 589
10.2 Reasonable and Prudent Measures 589
10.3 Terms and Conditions 590
11 Conservation Recommendations 592
12 Reinitiation Notice 593
13 References 595

LIST OF TABLES

Page

Table 1. Primary Mission Areas and Associated HCTT Activities (Navy 2024d)..... 26

Table 2. Standard Operating Procedures Proposed for HCTT Phase IV with Potential Conservation Benefits to Endangered Species and Their Habitats 38

Table 3. Activity-based Mitigation for Acoustic Stressors (Navy 2024d) 45

Table 4. Activity-based Mitigation for Explosive Stressors (Navy 2024d; Navy 2025e) 46

Table 5. Activity-based Mitigation for Non-Explosive Ordnance (Navy 2024d) 51

Table 6. Activity-based Mitigation for Vessels, Vehicles, and Towed In-Water Devices..... 54

Table 7. Requirements for Activity-based Mitigation Involving Navy Lookouts to Reduce the Risk of Entanglement in Nets used for Obstacle Avoidance Exercises..... 55

Table 8. Shallow-Water Coral Reef and Precious Coral Bed Mitigation Area Requirements (Navy 2024d)..... 58

Table 9. Artificial Reef, Hard Bottom Substrate, and Shipwreck Mitigation Area Requirements (Navy 2024d)..... 59

Table 10. Hawaii Island Marine Mammal Mitigation Area Requirements (Navy 2024d)..... 60

Table 11. Hawaii 4-Islands Marine Mammal Mitigation Area Requirements (Navy 2024d)..... 60

Table 12. Northern California Large Whale Mitigation Area Requirements (Navy 2025e)..... 61

Table 13. Central California Large Whale Mitigation Area Requirements (Navy 2024d)..... 61

Table 14. Southern California Blue Whale Mitigation Area Requirements (Navy 2024d)..... 62

Table 15. California Large Whale Awareness Requirements (Navy 2025e)..... 62

Table 16. California Real-Time Notification Large Whale Mitigation Area Requirements (Navy 2025e)..... 63

Table 17. San Nicolas Island Pinniped Haulout Mitigation Area Requirements (Navy 2024d)..... 64

Table 18. Sound Characteristics of Representative Aircraft Proposed for HCTT (Navy 2024d) 68

Table 19. Sonic Boom Underwater Sound Levels Modeled for F/A-18 Hornet Supersonic Flight (Navy 2024d) 70

Table 20. Sonar and Transducer Source Categories and Activity Levels (Navy 2024d) 71

Table 21. Examples of Noise from Weapons (Navy 2024d) 73

Table 22. Number of Shots from Air Guns Proposed for use in the Action Area (Navy 2024d) 74

Table 23. Number of Explosives within Each Bin that Would Occur Annually (and over seven years) for HCTT Military Readiness Activities (Navy 2024d) 76

Table 24. Typical Air Explosive Munitions Proposed for HCTT (Navy 2024g) 77

Table 25. Proposed Number of HCTT Events with In-Water Electromagnetic Devices (Navy 2024d) 78

Table 26. Number of Proposed HCTT Events Including High-Energy Lasers (Navy 2024d) 78

Table 27. Representative Vessel Types, Lengths, and Speeds (Navy 2024d) 82

Table 28. Representative Types, Sizes, and Speeds of In-Water Devices (Navy 2024d). 83

Table 29. Size Categories for Decelerators and Parachutes Expended During Proposed HCTT Activities (Navy 2024d). 85

Table 30. ESA-Listed or Proposed Species and Designated Critical Habitat That May be Affected by the Proposed Action 95

Table 31. HCTT Activities with Sonar and Explosive Stressors Proposed for Use within NOCAL RC (Navy 2024f) 153

Table 32. Comparison of Phase III and Phase IV Mortality and Injury Thresholds 372

Table 33. Summary of Phase IV TTS/AINJ Thresholds (SEL thresholds are in dB re: 1 μPa^2s underwater) by Marine Mammal Hearing Group 377

Table 34. Estimated Abundances of ESA-Listed Marine Mammal Stocks Present in the HCTT Action Area Used in NAEMO (Navy 2025f) 383

Table 35. Estimated Effects to the Eastern North Pacific Stock of Blue Whales over a Maximum Year of Proposed Activities (Navy 2025f) 384

Table 36. Estimated Effects to the Central North Pacific Stock of Blue Whales over a Maximum Year of Proposed Activities (Navy 2025f) 385

Table 37. Estimated Effects to the California, Oregon, and Washington Stock of Fin Whales over a Maximum Year of Proposed Activities (Navy 2025f) 386

Table 38. Estimated Effects to the Hawaii Stock of Fin Whales over a Maximum Year of Proposed Activities (Navy 2025f) 386

Table 39. Estimated Effects to the Humpback Whale Mexico DPS over a Maximum Year of Proposed Activities (Navy 2025f)..... 387

Table 40. Estimated Effects to the Humpback Whale Central America DPS over a Maximum Year of Proposed Activities (Navy 2025f)..... 388

Table 41. Estimated Effects to the Gray Whale Western North Pacific DPS over a Maximum Year of Proposed Activities (Navy 2025f)..... 389

Table 42. Estimated Effects to the Eastern North Pacific Stock of Sei Whales over a Maximum Year of Proposed Activities (Navy 2025f)..... 390

Table 43. Estimated Effects to the Hawaii Stock of Sei Whales over a Maximum Year of Proposed Activities (Navy 2025f) 391

Table 44. Estimated Effects to the California, Oregon, and Washington Stock of Sperm Whales over a Maximum Year of Proposed Activities (Navy 2025f) 392

Table 45. Estimated Effects to the Hawaii Stock of Sperm Whales over a Maximum Year of Proposed Activities (Navy 2025f)..... 392

Table 46. Estimated Effects to the Main Hawaiian Islands Insular Stock DPS of False Killer Whales over a Maximum Year of Proposed Activities (Navy 2025f)..... 393

Table 47. Estimated Effects to Hawaiian Monk Seals over a Maximum Year of Proposed Activities (Navy 2025f) 394

Table 48. Estimated Effects to Guadalupe Fur Seals over a Maximum Year of Proposed Activities (Navy 2025f) 395

Table 49. Annual Rates of Mortality and Serious Injury (M/SI) for Large Whale Species (or DPSs) from Vessel Strikes in the Central and Eastern North Pacific from 2017-2021, and Relative Probability of Striking each Large Whale Species (or DPS) 443

Table 50. Proportion of Overall Sonar Use in the Hawaii Range Complex with High to Continuous Duty Cycles (Navy 2025l)..... 467

Table 51. Estimated Effects to MHI IFKWs within Critical Habitat over a Maximum Year of Proposed Activities by Activity Type and Sonar Bin (Navy 2025c) 468

Table 53. Comparison of Phase III and Phase IV Sea Turtle TTS and AINJ Onset Levels for Non-impulsive (sonar) and Impulsive (explosives, airguns) Sound

Sources (weighted SEL onset units are dB re: 1 μPa^2s; impulsive peak SPL onsets units are dB re: 1 µPa) 474

Table 53. Estimated Effects to East Pacific DPS Green Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 477

Table 54. Estimated Effects to Central North Pacific DPS Green Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 478

Table 55. Estimated Effects to Hawksbill Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 478

Table 56. Estimated Effects to North Pacific Ocean DPS Loggerhead Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 479

Table 57. Estimated Effects to Olive Ridley Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 480

Table 58. Estimated Effects to Leatherback Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i) 480

Table 59. HCTT Explosive Ranges to Effects for Fishes without a Swim Bladder (Navy 2025a) 505

Table 60. HCTT Explosive Ranges to Effects for Fishes with a Swim Bladder (Navy 2025a) 506

Table 61. NMFS ESA effects determinations by stressor for each ESA-listed species and critical habitat, and overall effects determination for each species and critical habitat (LAA = likely to adversely affect; NLAA = not likely to adversely affect; NE = no effect) 519

Table 62. Number and Type of ESA Takes of Marine Mammals and Sea Turtles Anticipated to Occur as a Result of HCTT Over a Maximum Year of Activities from Acoustic and Explosive Stressors, and Over Every Seven Years from Vessel Strike 585

Table 63. Number and Type of ESA Takes of Marine Mammals and Sea Turtles Anticipated to Occur as a Result of HCTT Over Every Seven Years from Acoustic and Explosive Stressors (Navy 2025i; Navy 2025h) 587

LIST OF FIGURES

Page

Figure 1. Mitigation Areas in the California Portion of the HCTT Action Area (Navy 2025d) 56

Figure 2. Mitigation Areas in the Hawaii Portion of the HCTT Action Area (Navy 2025d) 57

Figure 3. Mean Vessel Speed of U.S. Military Vessels Operating within the Hawaiian Islands Portion of the HCTT Action Area from 2014-2018 (Starcovic and Mintz 2021) 80

Figure 4. Mean Vessel Speed of U.S. Military Vessels Operating within the SOCAL RC Portion of the HCTT Action Area from 2014-2018 (Starcovic and Mintz 2021) 81

Figure 5. Hawaii Range Complex (Navy 2024d) 90

Figure 6. California Portion of the HCTT Action Area (Navy 2024d) 91

Figure 7. Southern California Range Complex (Navy 2024d) 92

Figure 8. Point Mugu Sea Range (Navy 2024d) 93

Figure 9. Northern California Range Complex (Navy 2024d) 94

Figure 10. Aerial Target with Deployed Parachute (Navy 2024d) 131

Figure 11. Areas with Essential Habitat Features of Southern Resident Killer Whale. 152

Figure 12. White and Black Abalone Habitat in the Nearshore Waters of San Clemente Island (Tierra Data 2013). 162

Figure 13. Humpback Whale Critical Habitat in the HCTT Action Area (Navy 2024e) 169

Figure 14. Overlap of Humpback Whale Critical Habitat Units 17 and 18 with the PMSR 170

Figure 15. Southern Resident Killer Whale Designated Critical Habitat 174

Figure 16. Map Showing Overlap between Southern Resident Killer Whale Critical Habitat and the HCTT Action Area (Navy 2024e) 175

Figure 17. Critical Habitat for Hawaiian Monk Seal in the HCTT Study Area 179

Figure 18. Leatherback Sea Turtle Designated Critical Habitat along the U. S. Pacific Coast 184

Figure 19. Area of Overlap Between Leatherback Sea Turtle Designated Critical Habitat (purple area) and the HCTT Action Area (outlined in black) (Navy 2024e) 184

Figure 20. Critical Habitat for the Green Sturgeon Southern DPS 186

Figure 21. Area of Overlap Between Designated Critical Habitat for the Green Sturgeon Southern DPS and the HCTT Action Area (Navy 2024e) 187

Figure 22. Designated Critical Habitat for Black Abalone 190

Figure 23. Map Identifying the Range of the Endangered Blue Whale 195

Figure 24. Map Identifying the Range of the Endangered Fin Whale 199

Figure 25. Map Identifying the Range of the Gray Whale 204

Figure 26. Map Identifying 14 Humpback Whale DPSs, Including One Threatened and Four Endangered, Based on Primarily Breeding Locations, Range, and Feeding Areas (Bettridge et al. 2015) 208

Figure 27. Map Identifying the Range of the Endangered Sei Whale 214

Figure 28. Map Identifying the Range of the Endangered Sperm Whale 217

Figure 29. Map of the Main Hawaiian Islands Insular DPS of False Killer Whale Boundary (NMFS 2021d) 222

Figure 30. MHI IFKW Density by All Five Clusters Representing 38 Satellite-tagged Individuals Deployed through August 2021 (updated by Dr. Robin Baird, Cascadia Research Collective, pers. comm. August 2021 with the original source as Baird et al. (2012) (NMFS 2021e) 225

Figure 31. Cluster-level Space Use of Main Hawaiian Islands Insular False Killer Whales by Social Cluster Based on Satellite-tag Data from 2007 to 2022 (Mahaffy et al. 2023). 226

Figure 32. Designated Critical Habitat for Main Hawaiian Islands Insular DPS False Killer Whale 228

Figure 33. Guadalupe Fur Seal Range off the West Coast of the United States and Mexico (McCue 2021) 230

Figure 34. Hawaiian Archipelago and Range of the Hawaiian Monk Seal (Source: NMFS Pacific Islands Regional Office, Protected Resources Division) 233

Figure 35. Map Depicting Range and DPS Boundaries for Green Sea Turtles 236

Figure 36. Nesting Distribution of Green Turtles in the Central North Pacific DPS (water body labeled '10'). Size of circles indicates estimated nester abundance (Seminoff et al. 2015). 240

Figure 37. Nesting Distribution of Green Turtles in the East Pacific DPS (water body labeled '11'). Size of circles indicates estimated nester abundance (Seminoff et al. 2015). 240

Figure 38. Map Identifying the Range of the Endangered Hawksbill Sea Turtle..... 243

Figure 39. Map identifying the range of the endangered leatherback sea turtle. From NMFS https://www.fisheries.noaa.gov/species/leatherback-turtle#overview 247

Figure 40. Map Identifying the Ranges of the East and West Pacific Leatherback Sea Turtle Populations 247

Figure 41. Map Identifying the Range and DPS Boundaries of the Loggerhead Sea Turtle 252

Figure 42. Map Identifying the Range of the Olive Ridley Sea Turtle..... 255

Figure 43. Extent of Occurrence (dark blue) and Area of Occupancy (light blue) for Giant Manta Rays based on Species Distribution (Lawson et al. 2017) 259

Figure 44. Geographic Range of the Oceanic Whitetip Shark (Young and Carlson 2020) 264

Figure 45. Scalloped Hammerhead Shark DPS Boundaries (Source: 78 Fed. Reg. 20717; April 5, 2013). Note: the Northwest Atlantic/Gulf of Mexico and Central Pacific DPSs are not listed under the ESA. 267

Figure 46. Steelhead Southern California DPS Inland Range and Designated Critical Habitat 271

Figure 47. Geographic Inland Range and Designated Critical Habitat of California Central Valley Steelhead..... 274

Figure 48. Geographic Inland Range and Designated Critical Habitat of Central California Coast Steelhead..... 277

Figure 49. Geographic Inland Range and Designated Critical Habitat of Northern California DPS Steelhead 279

Figure 50. Geographic Inland Range and Designated Critical Habitat of South-Central California Coast Steelhead 281

Figure 51. Geographic Inland Range and Designated Critical Habitat of California Coastal ESU Chinook Salmon 284

Figure 52. Geographic Inland Range and Designated Critical Habitat of Central Valley Spring-Run ESU Chinook Salmon..... 287

Figure 53. Geographic Inland Range and Designated Critical Habitat of the Sacramento River Winter-Run ESU of Chinook Salmon..... 290

Figure 54. Geographic Inland Range of Coho Salmon Central California Coast ESU 293

Figure 55. Geographic Inland Range and Designated Critical Habitat of Coho Salmon Oregon Coast ESU 296

Figure 56. Geographic Inland Range of the Southern Oregon/Northern California ESU of Coho Salmon 299

Figure 57. Map Identifying the Spawning Rivers of the Eulachon Southern DPS (NMFS 2016d) 301

Figure 58. Percent of Vessel Traffic Intensity (i.e., Number of Vessel Transits or Tracks) in HCTT Action Area by Vessel Type, 2014-2018 (Starcovic and Mintz 2021). 315

Figure 59. Relative Distribution of Nonmilitary Vessel Traffic Intensity (i.e., Number of Vessel Transits or Tracks) in HCTT, 2016-2023 (Navy 2025k) 316

Figure 60. Seismic Survey Track Lines within the California Portion of the Action Area (NMFS 2015a) 318

Figure 61. Oil Spill Incidents on the Coast of California from 2004 to 2020 (NOAA Office of Response and Restoration 2021). Marine oil spills locations are indicated by the dark blue circles. 329

Figure 62. Guadalupe Fur Seal Annual Strandings in California, 2013 to 2021. Orange bars indicate unusual mortality event years (NMFS 2021a). 337

Figure 63. Guadalupe Fur Seal Annual Strandings in Oregon and Washington, 2013 to Present. Blue – Oregon; Orange – Washington (NMFS 2021a) 338

Figure 64. Annual Gray Whale Strandings in the United States, Mexico, and Canada, 2015 to 2023 338

Figure 65. Confirmed Whale Entanglement Reports off the U.S. West Coast from 1982-2017 (N=429) by Year and Color-coded by Species (Saez et al. 2021) 343

Figure 66. California Ship Strikes by Whale Species (2017–2024) 357

Figure 67. Navy Phase IV Weighting Functions by Marine Mammal Species Group (amplitude in dB on y-axis; frequency in kHz on x-axis) 374

Figure 68. Comparison of Navy Phase IV TTS Exposure Functions for Sonars and Other Non-impulsive Active Acoustic Sources across Species Groups (exposure SEL in dB on y-axis; frequency in kHz on x-axis) 375

Figure 69. Comparison of Navy Phase IV TTS Exposure Functions for Explosives, Impact Pile Driving, Air Guns and Other Impulsive Acoustic Sources

Across Species Groups (exposure SEL in dB on y-axis; frequency in kHz on x-axis) 376

Figure 70. Phase IV Mysticete Biphasic Behavioral Response Function (50% credible interval shaded in teal) 379

Figure 71. Phase IV Odontocete Biphasic Behavioral Response Function (50% credible interval shaded in teal) 380

Figure 72. Phase IV Pinniped (in Water) Biphasic Behavioral Response Function (50% credible interval shaded in teal) 380

Figure 73. Intrinsic and Extrinsic Factors to Consider when Assessing Likely Consequences to Individuals and Marine Mammal Populations from Anthropogenic Stressors (Keen et al. 2021). 418

Figure 74. Distances to Non-Impulsive Phocid Thresholds from Launch of T3C2 Missile at PMRF (Navy 2025g). Green circle represents a 620 ft. range to effect. 424

Figure 75. Locations of U.S. Navy Vessel Strikes of Large Whales from 2009 through August 2025 within the SOCAL RC Portion of the Action Area (Navy 2023d) 429

Figure 76. MHI IFKW Designated Critical Habitat and Hawaii Range Complex Sub-areas where Most Explosives are Used During Training and Testing 458

Figure 77. Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat and Hawaii Range Complex Subareas Used Most Often for Anti-submarine Warfare Training and Testing Activities (NMFS 2018b) 464

Figure 78. Comparison of Phase III (thin gray lines) and Phase IV (thick black lines) Sea Turtle Composite Audiograms (Navy 2025f) 475

Figure 79. Comparison of Phase III (thin gray lines) and Phase IV (thick black lines) Sea Turtle Exposure Functions for Non-impulsive TTS (solid lines) and AINJ (dashed lines) (Navy 2025e) 475

Figure 80. Hearing Frequency Ranges (pressure) of some ESA-listed Fish Groups Compared to Navy Defined Sonar Classes. Sources: (Tavolga and Wodinsky 1963; Hawkins and Johnstone 1978; Popper et al. 2007; Popper 2008; Popper et al. 2014a) 495

Figure 81. Injury Rates for Severe Injury (a) and Any Injury (b) of Pacific Mackerel Exposed to Explosive Detonations at Varying Sound Pressure Levels 503

1 INTRODUCTION

The Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 et seq.) establishes a national mandate for conserving threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a)(2) of the Act and its implementing regulations require every Federal agency, in consultation with and with the assistance of the Secretary (16 U.S.C. §1532(15)), to insure that any action it authorizes, funds, or carries out, in whole or in part, in the United States or upon the high seas, is not likely to jeopardize the continued existence of any listed species or results in the destruction or adverse modification of critical habitat.

Section 7(a)(4) of the ESA requires Federal agencies to confer with the Secretary on any action that is likely to jeopardize the continued existence of proposed species or result in the destruction or adverse modification of proposed critical habitat. For actions that are not likely to jeopardize the continued existence of a proposed species or adversely modify critical habitat, a conference can be requested by the action agency though it is not required. If requested by the Federal action agency and deemed appropriate, the conference may be conducted in accordance with the procedures for formal consultation in 50 CFR §402.14. An opinion issued at the conclusion of the conference may be adopted as the biological opinion when the species is listed or critical habitat is designated.

Section 7(b)(3) of the ESA requires that, at the conclusion of consultation and conference, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify their designated habitat. If NMFS determines that the action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If the action (or reasonable and prudent alternative) is expected to cause incidental take without violating section 7(a)(2), section 7(b)(4), as implemented by 50 CFR §402.14(i), requires NMFS to provide an incidental take statement (ITS) that specifies the amount or extent of incidental taking. Take in the ESA is to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct (16 U.S.C. §1532(19)). The ITS includes reasonable and prudent measures, which are actions necessary or appropriate to minimize impacts of incidental taking and terms and conditions to implement the reasonable and prudent measures.

The Federal action agencies for this consultation are the United States (U.S.) Navy (Navy), U.S. Coast Guard (Coast Guard), U.S. Army (Army), U.S. Air Force (Air Force) and NMFS's Permits and Conservation Division (Permits Division). The Navy, Coast Guard, Army, and Air Force (hereinafter jointly referred to as the Action Proponents) propose to conduct Hawaii-California Training and Testing (HCTT) and other military readiness activities. 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 Action Proponents to "take" marine mammals

incidental to HCTT activities. The regulations propose the issuance of Letters of Authorization (LOAs) that will authorize the Action Proponents to "take" marine mammals incidental to its proposed action, pursuant to the requirements of the MMPA.

Updates to the regulations governing interagency consultation (50 CFR part 402) were effective on May 6, 2024 (89 Fed. Reg. 24268). We are applying the updated regulations to this consultation. The 2024 regulatory changes, like those from 2019, were intended to improve and clarify the consultation process, and, with one exception from 2024 (offsetting reasonable and prudent measures), were not intended to result in changes to the Services' existing practice in implementing section 7(a)(2) of the Act (89 Fed. Reg. 24268; 84 Fed. Reg. 45015). We have considered the prior rules and affirm that the substantive analysis and conclusions articulated in this biological opinion and incidental take statement would not have been any different under the 2019 regulations or pre-2019 regulations. No offsetting reasonable and prudent measures are included in the ITS for this action.

Consultation in accordance with section 7(a)(2) of the statute (16 U.S.C §1536(a)(2)), associated implementing regulations (50 CFR Part 402), and agency policy and guidance (USFWS and NMFS 1998) was conducted by the NMFS Office of Protected Resources (OPR) ESA Interagency Cooperation Division (hereafter referred to as 'we' or 'us'). We prepared this biological and conference opinion (opinion) and ITS in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR Part 402.

This document represents NMFS's opinion on the effects of these actions on the following ESA-listed species: blue whale (*Balaenoptera musculus)*; fin whale (*Balaenoptera physalus)*; Western North Pacific distinct population segment (DPS) gray whale (*Eschrichtius robustus*); Mexico DPS humpback whale (*Megaptera novaeangliae*); Central America DPS humpback whale; sei whale (*Balaenoptera borealis*); sperm whale (*Physeter macrocephalus*); Main Hawaiian Islands Insular DPS false killer whale (*Pseudorca crassidens*); North Pacific Right Whale (*Eubalaena japonica*); Killer Whale (*Orcinus orca*) – Southern Resident DPS; Guadalupe fur seal (*Arctocephalus townsendi*); Hawaiian monk seal (*Neomonachus schauinslandi*); green sea turtle (*Chelonia mydas*) – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle (*Lepidochelys olivacea*) – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle (*Eretmochelys imbricata)*; leatherback sea turtle (*Dermochelys coriacea*); loggerhead sea turtle (*Caretta caretta*) – North Pacific DPS; Southern California DPS steelhead (*Oncorhynchus mykiss*); Steelhead (*Oncorhynchus mykiss*) – California Central Valley DPS; Steelhead – Central California Coast DPS; Steelhead – Northern California DPS; Steelhead – South-Central California Coast DPS; Chinook salmon (*Oncorhynchus tshawytscha*) – California Coastal ESU; Chinook salmon – Central Valley Spring-Run ESU; Chinook salmon (*Oncorhynchus tshawytscha*) – Sacramento River Winter-Run ESU; Coho salmon (*Oncorhynchus kisutch*) – Central California Coast ESU; Coho salmon – Oregon Coast ESU; Coho salmon – Southern Oregon and Northern California Coasts ESU; Eastern Pacific DPS scalloped hammerhead shark (*Sphyrna lewini*); oceanic whitetip shark (*Carcharhinus*

longimanus); giant manta ray (*Manta birostris*); black abalone (*Haliotis cracherodii*); white abalone (Haliotis sorenseni); and proposed sunflower sea star (Pycnopodia helianthoides).

This document also represents NMFS's opinion on the effects of these actions on the following designated critical habitat: Mexico DPS humpback whale designated critical habitat; Central America DPS humpback whale designated critical habitat; Main Hawaiian Islands Insular DPS false killer whale designated critical habitat; Southern Resident killer whale designated critical habitat; Hawaiian monk seal designated critical habitat; leatherback sea turtle designated critical habitat; green sturgeon Southern DPS designated critical habitat; and black abalone designated critical habitat.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). A complete record of this consultation is on file electronically with the NMFS Office of Protected Resources (OPR) in Silver Spring, Maryland and available online in the NOAA Library Institutional Repository https://repository.library.noaa.gov/welcome.

1.1 Background

Between 2007 and 2013, NMFS issued a series of biological opinions on Navy training and testing activities off the coast of Southern California and around Hawaii. Where incidental take of marine mammals was anticipated, these consultations also considered the 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 opinions concluded that the Navy and Permits Division's proposed actions would not jeopardize the continued existence of threatened or endangered species or destroy or adversely modify designated critical habitat. Collectively, NMFS and the Navy referred to the activities that were the subject of these consultations as Phase I.[1]

On December 13, 2013, NMFS issued a biological opinion on proposed Phase II HSTT activities starting in December 2013 and the associated MMPA authorization of incidental take of marine mammals by the NMFS Permits Division from December 2013 to December 2018. For the consultation on Phase II activities, the Navy grouped many of the same training and testing activities considered in previous stand-alone consultations, including activities conducted off the coast of Southern California and Hawaii, into a single proposed action. The opinion concluded that the Navy and Permits Division's proposed actions would not jeopardize the continued existence of threatened or endangered species or destroy or adversely modify designated critical habitat. After issuance of the December 13, 2013 biological opinion, NMFS identified several inadvertent factual errors and omissions regarding amount of incidental take of species from

[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.

vessel strike and omission of potential sea turtle injury or mortality from the ITS. On April 23, 2014, NMFS issued a corrected final biological opinion and ITS that superseded the December 13, 2013 biological opinion.

On March 31, 2015, the United States District Court for the District of Hawaii ruled that NMFS's biological opinion for the Navy's Phase II activities in the HSTT action area included an arbitrary and capricious "no jeopardy" finding for whales and sea turtles and an invalid ITS for sea turtles (Conservation Council for Hawaii v. NMFS; Natural Resources Defense Council v. NMFS). On April 2, 2015, NMFS issued a biological opinion which addressed the issue identified by the court. The reinitiated opinion concluded that the Navy 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.

On December 10, 2018, we issued a biological opinion (NMFS 2018b) under the ESA concluding that the Navy's proposed HSTT activities and the Permits Division's proposal to issue five-year MMPA incidental take regulations and LOAs to the Navy were not likely to jeopardize the continued existence of the ESA-listed species under NMFS's jurisdiction, and were not likely to result in the destruction or adverse modification of critical habitat in the HSTT action area. On January 14, 2020, we issued an amended ITS to ensure the ITS associated with the December 2018 biological opinion was consistent with the Permits Division's issuance of a revised MMPA rule and new LOAs extending the HSTT MMPA rule by an additional two years through 2025.

On May 10, 2021, the Navy notified us that a Royal Australian Navy vessel struck and killed two fin whales while operating within the HSTT action area. On May 17, 2021, the Center for Biological Diversity sent the Navy and NMFS a 60-Day notice of intent (NOI) to sue for violations of the ESA regarding Navy HSTT activities. On July 13, 2021, the Navy reported two large whale vessel strikes by U.S. Navy vessels operating in the Southern California Range. The first occurred on June 29, 2021, and the second occurred on July 11, 2021. As stated in 50 CFR §402.16, reinitiation of formal consultation is required, and shall be requested by the Federal action agency or by NMFS, where discretionary Federal agency involvement or control over the action has been retained or is authorized by law, and one or more of the four reinitiation triggers has been met. Based on our evaluation of the reported military vessel strike incidents in 2021, and other relevant new information, we found that the reinitiation trigger concerning new information (*i.e.*, 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) had been met. On July 13, 2021, the Navy responded to the Center for Biological Diversity indicating that they intended to reinitiate formal consultation for HSTT activities on the basis of new information. On July 14, 2021, we responded to the Center for Biological Diversity that we agreed that reinitiation of the HSTT consultation was required and that we had requested reinitiation from the Navy and the Permits Division.

On November 25, 2020, the Navy requested initiation of a formal ESA section 7 consultation to assess potential effects of training and testing activities in the Point Mugu Sea Range (PMSR) on ESA-listed species and critical habitat. On January 25, 2022, we completed the PMSR consultation and issued a biological opinion (NMFS 2022b) in which we determined that the training and testing activities on PMSR are not likely to jeopardize the continued existence of any ESA-listed species or result in the destruction of adverse modification of designated critical habitat. We also issued an incidental take statement exempting take of the following species: blue whale, fin whale, humpback whale (Central America DPS and Mexico DPS), sperm whale, Guadalupe fur seal, and leatherback sea turtle.

On October 26, 2023, we notified the Navy and the Permits Division that we were reinitiating formal ESA section 7 consultation for HSTT Phase III. On July 19, 2023, the Navy indicated they wanted to conference on the sunflower sea star, which was proposed for listing as threatened under the ESA on March 16, 2023. On June 3, 2024, we issued a biological and conference opinion (NMFS 2024f) under the ESA concluding that the Navy's proposed HSTT activities and the Permits Division's proposal to issue seven-year MMPA incidental take regulations and LOAs to the Navy were not likely to jeopardize the continued existence of the ESA-listed or proposed species under NMFS's jurisdiction, and were not likely to result in the destruction or adverse modification of critical habitat in the HSTT action area.

1.2 Consultation History

Our communication with the Action Proponents and Permits Division regarding this consultation is summarized as follows:

- In December 2023, the Action Proponents provided NMFS with two sections of the HCTT Draft EIS/OEIS for review: 1. Purpose and Need for the Proposed Action and 2. Description of Proposed Action and Alternatives. We reviewed the documents and provided comments and questions to the Action Proponents in January 2024.
- On April 29, 2024, the Action Proponents notified us that the HCTT Draft EIS/OEIS (Version 2) was ready for NMFS's review. NMFS reviewed the document and provided comments online prior to the June 28, 2024 comment deadline.
- On August 9, 2024, the Navy sent a draft of the HCTT Explosive and Acoustic Analysis Report (Version 1), which included changes from Phase III to the Navy's approach to quantifying acoustic effects from sonar and explosives in Phase IV, and estimates of the acoustic impacts to marine mammals and sea turtles. We reviewed the report and provided comments and questions on August 23, 2024.
- On September 11, 2024, the Action Proponents notified us that their draft HCTT Phase IV Biological Assessment (BA) was available for NMFS review. We reviewed the BA and responded with comments and follow up questions on September 23, 2024.
- On October 9, 2024, we received a letter from the Action Proponents requesting formal ESA section 7 consultation for HCTT Phase IV activities. The consultation initiation package included the following supporting documents: revised final HCTT Phase IV BA

(Navy 2024d); *Acoustic and Explosive Effects Analysis for Marine Mammals in the Hawaii - California Training and Testing Study Area* (Navy 2025a); and the HCTT Phase IV density technical report (Navy 2025n). The Action Proponents also provided us with responses to our comments and questions based on our review of the draft BA.

- On October 15, 2024, we sent the Action Proponents follow up questions based on the revised BA, including a question asking if the Navy would consider identification of designated ship shock trial boxes for this activity involving explosives. On October 29, 2024, the Action Proponents sent us a Memo for the Record (MFR) (Navy 2024g) responding to our follow up questions, including a proposed ship shock trial location within the Southern California Range Complex (SOCAL RC).
- On November 13, 2024, the Navy sent us a MFR with additional information in support of the Action Proponents HCTT BA (Navy 2024f). Additional information included a more detailed description of the Northern California Range Complex (NOCAL RC) and the proposed activities within this portion of the action area, and new information on green sturgeon distribution in relationship to NOCAL RC.
- On November 26, 2024, we notified the Action Proponents that we had accepted their BA as complete. We noted that since the Action Proponents' proposed action is interrelated with the Permits Division's proposed issuance of MMPA regulations, we could not initiate formal ESA section 7 consultation on the HCTT Phase IV proposed action until we receive and accept as complete the Permits Division's initiation package (including the proposed rule).
- On December 9, 2024, we met with the Navy and the Permits Division to discuss their large whale vessel strike analysis as presented in the HCTT BA.
- On December 18, 2024, we met with the Action Proponents and the Permits Division to discuss the HCTT large whale vessel strike analysis. The Coast Guard indicated they would provide us with additional information regarding historical Coast Guard vessel activity and large whale strikes, both within and outside of the action area, to inform our effects analysis. We also indicated that, if possible, we would want to determine the proportion of vessel activity (and/or large whale strikes) that were already accounted for in the Coast Guard's 2023 Offshore Patrol Cutter programmatic biological opinion.
- On January 7, 2025 the Coast Guard provided us with a spreadsheet documenting past vessel strike incidents involving cetaceans in Hawaii, California, the Pacific Northwest and Alaska.
- On February 24, 2025, the Navy sent us a MFR with their revised HCTT vessel strike probability analysis.
- On March 3, 2025, the Navy sent us revised Guadalupe fur seal and Hawaiian monk seal acoustic impacts estimates based on recent abundance changes, and revised acoustic impacts estimates for sea turtles based on correction of an error identified in their previous analysis.

- On March 19, 2025, we emailed the Navy requesting that consultation for the Octopus I and Octopus II fiber optic cable maintenance and installation projects proposed for HCTT Phase IV occur either under the existing Wave Energy Test Site (WETS) Programmatic Letter of Concurrence (LOC) or through a new consultation with the NMFS Pacific Islands Regional Office (PIRO). The Navy responded on March 24 that they were amenable to removing the Octopus I and II projects from the HCTT Phase IV consultation and consulting with PIRO on this action.
- On April 4, 2025, the Navy provided us with an MFR titled *Sea Turtle Auditory Effects Criteria Update and Model Results Adjustments*.
- On April 8, 2025, the Navy sent us the following documents with updated information and responses to our questions: 1) Addendum to the HCTT Phase IV BA; 2) Revised Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area; and 3) Summary of changes to the ranges to effects for marine mammals, sea turtles, and fishes used to estimate acoustic impacts.
- On April 24, 2025, we emailed the Navy with a request for additional information regarding HCTT Phase IV modeled acoustic impacts to Main Hawaiian Islands insular false killer whales (MHI IFKWs) with critical habitat. The Navy responded to this request with an MFR on April 25.
- On May 16, 2025, we emailed the Navy with a request for additional information regarding unpublished vessel traffic information within HCTT. The Navy responded with the requested information in an MFR dated May 19.
- From May 29 through June 2, 2025, we emailed the Navy requesting additional information based on comments received during our internal review of the draft HCTT biological and conference opinion. The Navy responded with the requested information in an MFR dated June 5.
- On June 9, 2025, NMFS OPR and Navy staff participated in a conference call to discuss additional mitigation measures to minimize impacts to MHI IFKWs resulting from proposed HCTT Phase IV activities. On June 25, 2025, the Navy provided us with an MFR with their evaluation of the practicability and effectiveness of additional MHI IFKW mitigation measures.
- On June 24, 2025, we emailed the Navy with a request for additional information based on comments we received from PIRO in their review of the draft HCTT biological and conference opinion. The Navy responded with the requested information in an MFR dated June 30, 2025.
- On July 16, 2025, the Navy notified us of a large whale strike by a Navy vessel in the SOCAL RC on July 15, 2025.
- On July 29, 2025 we notified the Action Proponents and the Permits Division that we were initiating formal ESA section 7 consultation.

- On August 7, 2025, NMFS OPR, PIRO, and Navy staff participated in a conference call to discuss the Navy's MFR (dated June 25, 2025) evaluating the practicability and effectiveness of implementing additional MHI IFKW mitigation measures. During the call NMFS staff recommended that the Navy reconsider additional mitigation measures, particularly in areas identified as high use areas by MHI IFKW, as being beneficial to this extremely small, declining population. On September 11, 2025, the Navy notified us that they were proposing to expand the Hawaii 4-Islands Mitigation Area to include additional areas identified as high use areas by MHI IFKWs.
- On August 11, 2025, the Navy notified us of a large whale strike by a Navy vessel in the SOCAL RC on August 10, 2025.
- From August 19 through September 18, 2025, we consulted with the Navy through conference calls and email on revisions to the large whale vessel strike effects analysis in light of the two strikes in 2025. On September 19, 2025, the Navy sent us an updated version of their vessel strike analysis.

1.3 Analytical Approach

This opinion includes a jeopardy analysis for the ESA-listed and proposed species and an adverse modification or destruction of critical habitat analysis for the designated critical habitat within the action area. "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 CFR §402.02). "Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of an ESA-listed species as a whole (50 CFR §402.02).

Prior to 2016, the designation(s) of critical habitat used the term primary constituent element (PCE), essential features, or generally identified aspects of critical habitat that were essential to the conservation of the species. The 2016 critical habitat regulations (50 CFR §424.12) replaced these terms with the term physical and biological feature (PBF).The shift in terminology does not change the approach used in conducting a ''destruction or adverse modification'' analysis, which is the same regardless of whether the original designation identified PCEs, PBFs, or essential features. In this opinion, we use the term PBF to mean PCE or essential feature, as appropriate for the specific critical habitat.

We use the following approach to determine whether an action agency is able to insure its proposed action is not likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Identify all aspects of the proposed action (as defined in 50 CFR §402.02), including activities that rely on the action for their occurrence. This section also includes any avoidance and minimization measures that have been incorporated into the action to reduce the effects to ESA-listed species.

- Identify the physical, chemical, and biological modifications to land, water, and air (stressors) that result from those actions and subsequent activities.
- Establish the aerial extent of those stressors, which is the action area (50 CFR §402.02).
- Identify the listed species (as defined at 16 U.S.C. §1532(16)) and designated critical habitat (as defined at 16 U.S.C. §1532(5)) in the action area.
- Identify the species and critical habitats that are not likely to be adversely affected by the action.
- Evaluate the range-wide status of the species and critical habitat expected to be adversely affected by the proposed action. This includes a discussion of the species' abundance, population growth rate, genetic diversity, and spatial distribution, as available.
- Evaluate the environmental baseline (as defined in 50 CFR §402.02) as it pertains to the species and critical habitat.
- Evaluate the effects of the proposed action on listed species and their designated critical habitat using an exposure-response approach. When complete, this section anticipates the amount or extent, as well as the forms (harass, harm, etc.), of take of listed species (or a surrogate) that is reasonably certain to occur as a result of the action, as well as the extent of effects to critical habitat.[2]
- Evaluate cumulative effects (as defined at 50 CFR §402.02).
- Produce an integration and synthesis, where we add the effects of the action and cumulative effects to the environmental baseline, and, in light of the status of the species and critical habitat, analyze whether the proposed action is likely to jeopardize the continued existence of listed species or destroy or adversely modify critical habitat.
- Compile our jeopardy and destruction or adverse modification analysis conclusions relying on the justification in the integration and synthesis.
- If the opinion determines the action agencies failed to insure their actions are not likely to jeopardize the continued existence of listed species or destroy or adversely modify critical habitat, we suggest a reasonable and prudent alternative to the proposed action and assess the effects of that alternative action.
- For actions that do not violate section 7(a)(2) of the ESA or an alternative action is identified that does not violate section 7(a)(2) of the ESA and when take of marine mammals is permitted under section 101(a)(5) of the MMPA, after we conclude our opinion, we provide an incidental take statement that specifies the impact of the take on listed species (amount or extent), reasonable and prudent measures, and terms and conditions to implement those measures.

[2] Because take is categorized differently under ESA and MMPA, the initiation documents provided by Action Proponents, NMFS' Permits Division, and the analysis in this opinion may discuss take differently and identify different numbers of takes as the reasonably certain outcomes of this action (see interim guidance at *https://www.fisheries.noaa.gov/s3/2023-05/02-110-19-renewal-kdr.pdf).*

In each of these steps above, we rely on the best scientific and commercial data available. To conduct the analyses necessary for this opinion, 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. We conducted electronic literature searches throughout this consultation, including within the NMFS Office of Protected Resources' electronic library.

This opinion is based on information provided by the Action Proponents during pre-consultation technical assistance and other supplemental information provided throughout the consultation process. We examined the Action Proponents HCTT Biological Assessment (Navy 2024d), HCTT Draft EIS/OEIS (Navy 2024b) and Final EIS/OEIS (Navy 2025d), the Navy's technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training and Testing* (Navy 2024h), HCTT Phase III annual training and testing reports, literature cited in these documents, and other supplemental information provided throughout the consultation process (Navy 2024g; Navy 2024f; Navy 2025e; Navy 2025b; Navy 2025j; Navy 2025k; Navy 2025l; Navy 2025m). 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. In addition, we engaged regularly with the Action Proponents to discuss new science and technical issues as part of the ongoing adaptive management program for military readiness activities and incorporated new information obtained as a result of these engagements in this consultation.

This opinion also considers information provided by the Permits Division, including its request for initiation of section 7 consultation under the ESA, which included the proposed Federal regulations under the MMPA to authorize the incidental take of marine mammals, including ESA-listed marine mammals, specific to the proposed activities (90Fed. Reg. 321118) and related draft letters of authorization.

We incorporated information from NMFS's draft or final recovery plans, species status reports, 5-year status review updates, and critical habitat reports for the endangered or threatened species considered in this document, and examined scientific literature on ESA-listed species and habitats. We identified information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS's jurisdiction that may be affected by the HCTT Phase IV activities.

Collectively, we consider the information described above to comprise the best scientific and commercial information available for this consultation and in preparation of this opinion.

2 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 in the U.S. or on the high seas. Examples include, but are not limited to: 1) actions intended to conserve listed species or their habitat; 2) the promulgation of regulations; 3) the granting of licenses, contracts, leases, easements, rights-of-way, permits, or

grants in aid; or 4) actions directly or indirectly causing modifications to the land, water, or air (50 CFR §402.02). Two Federal actions were evaluated during this consultation. The first action is the Action Proponents' military readiness activities conducted in the HCTT action area into the foreseeable future. The second action is the Permits Division's implementation of the MMPA through: 1) promulgation of regulations pursuant to the MMPA as amended (16 U.S.C. 1361 et seq.) governing the Action Proponents' "take" of marine mammals incidental to military readiness activities within the HCTT action area through November 2032; and 2) issuance of LOAs pursuant to the regulations that authorize the Action Proponents to "take" marine mammals under the MMPA incidental to military readiness activities within the HCTT action area through December 2032.

The Action Proponents propose to conduct military readiness activities, comprised of training, testing, including the use of active sonar and explosives, and modernization and sustainment of ranges across the north-central Pacific Ocean from the mean high tide line in Southern California west to Hawaii and the International Date Line. The purpose of the military readiness activities the Action Proponents conduct in the HCTT action area is to achieve and maintain fleet readiness and to meet their U.S. Code Title 10 (Navy) and Title 14 (Coast Guard) missions to maintain, train, and equip combat-ready forces capable of winning wars, deterring aggression, and maintaining freedom of the seas. The Action Proponents have historically used the areas off of Hawaii and along the southern Pacific coast of the U.S. for training and testing. The Action Proponents use a combination of undersea ranges, operating areas (OPAREAs), land training ranges, at-sea airspace, and nearshore amphibious landing sites to achieve their training and testing objectives (Navy 2024d). OPAREAs are specific ocean areas within a range complex defined by geographic coordinates with defined surface and subsurface areas and associated special use airspace. Within an OPAREA there is a restricted area that is defined to prohibit or limit public access to the area.

NMFS recognizes that while the Action Proponents' 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 assume that the military readiness activities proposed by the Action Proponents during the 7-year period of the Permits Division's proposed incidental take authorization pursuant to the MMPA will continue into the reasonably foreseeable future at levels similar to those assessed in this opinion.

Below, we present information on: the primary mission areas, specific types of military readiness activities, and activity levels proposed in the different locations throughout the action area (Sections 2.1 and 2.2); the standard operating procedures and mitigation measures that will be implemented by the Action Proponents as part of their military readiness activities (Section 2.4); and the stressors resulting from the proposed activities (Section 2.5). In Section 2.6, we discuss the Permits Division's proposed action under the MMPA.

More details regarding the proposed action can be found in the Action Proponents' HCTT BA (Navy 2024d). We incorporate, by reference, specific sections of the BA into this opinion, where appropriate.

2.1 Primary Mission Areas

The Action Proponents categorize their activities into functional warfare areas called primary mission areas. These activities generally fall into the following seven primary mission areas (Table 1).

Table 1. Primary Mission Areas and Associated HCTT Activities (Navy 2024d)

Primary Mission Area	Description	Example Activities
Air Warfare	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. Testing of air warfare systems is required to ensure the equipment is fully functional.	Air Combat Maneuver; Air Defense exercises; Gunnery and Missile Exercises (air-to-air and surface-to-air); Intelligence, Surveillance, and Reconnaissance Test
Amphibious warfare	Individual and crew training includes shore assaults, boat raids, airfield or port seizures, reconnaissance, and disaster relief. Large-scale amphibious exercises involve ship-to-shore maneuvers, naval fire support (such as shore bombardment), air strikes, and attacks on targets that are in close proximity to friendly forces. Testing is performed to ensure effective ship-to-shore coordination and transport of personnel, equipment, and supplies. Tests may also be conducted periodically to assess operability and to investigate efficacy of new technologies.	Amphibious Assault; Amphibious Operations in Contested Environment; Amphibious Raid; Ready Group Marine Expeditionary Unit Exercise; Squadron Marine Expeditionary Unit Integration Training; Vehicle Maneuvers; Naval Surface Fire Support Exercise (At-sea and Land-based); Non-Combat Evacuation Operation
Anti-submarine warfare	Training includes detecting and classifying submarines, evaluating ocean sounds, tracking a submarine, attacking a target using either exercise torpedoes (i.e., torpedoes that do not contain a warhead) or simulated weapons. Testing uses ships, submarines, and aircraft to demonstrate capabilities of torpedoes, missiles, countermeasure systems, and underwater surveillance and communications systems.	Torpedo Exercises (Helicopter, Maritime Patrol Aircraft, Ship, Submarine); Tracking Exercises (Helicopter, Maritime Patrol Aircraft, Ship, Submarine); Kilo Dip Test; Anti-Submarine Warfare Mission Package Testing; Surface Ship Sonar Testing/Maintenance

Primary Mission Area	Description	Example Activities
Electronic warfare	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.	Counter Targeting Chaff Exercises – Aircraft/Ship; Counter Targeting Flare Exercise; Electronic Warfare Operations; High-Speed Anti-Radiation Missile Exercise; Electronic Systems Test; Radar and Other System Testing
Expeditionary warfare	Includes defense of harbors, operation of remotely operated vehicles, defense against swimmers, boarding/seizure operations, underwater construction, dive and salvage operations, and insertion/extraction via air, surface, and subsurface platforms.	Dive and Salvage Operations; Personnel Insertion/Extraction – Air, Surface and Subsurface, and Swimmer/Diver training; Port Damage Repair and Underwater Construction Training
Mine warfare	Training includes mine detection and classification, mine countermeasure and neutralization, and offensive mine laying. Testing focuses on the development of mine warfare systems to improve sonar, laser, and magnetic detectors intended to hunt, locate, and record the positions of mines.	Airborne Mine Countermeasures – Mine Detection and Towed Mine Neutralization; Airborne Mine Laying; Civilian Port Defense Exercises; Coordinated Unit-Level Helicopter Airborne Mine; Countermeasures Exercise; Installation and Maintenance of Mine Training Areas; Mine Countermeasures/ Neutralization – Remotely Operated Vehicles; Mine Countermeasures – Ship Sonar; Mine Neutralization – Explosive Ordnance Disposal; Submarine Mobile Mine and Mine Laying Exercise; Surface Ship Object Detection; Underwater Mine Countermeasures Operations; Airborne Dipping Sonar Minehunting Test; Mine Laying Test
Surface warfare	Training includes gunnery and missile exercises, submarine missile or torpedo launch events, and other munitions against surface targets. Testing of weapons is conducted to develop new technologies and to assess weapon performance and operability.	Bombing Exercise Air-to-Surface; Gunnery Exercises - Air-to-Surface and Surface-to-Surface; Integrated Live Fire Exercise; Laser Targeting – Aircraft and Ship; Long-Range Unmanned Surface Vessel Training; Maritime Security Operations; Missile

Primary Mission Area	Description	Example Activities
		Exercises - Air-to-Surface, Air-to-Surface Rocket, and Surface-to-Surface; Sinking Exercise; Small Boat Attack; Long-Range Weapons Delivery Systems (Over-the-Horizon)/Hypersonic Vehicle Test; Missile and Rocket Testing

For the training activities considered during consultation, military personnel (e.g., Sailors, Soldiers, Marines, or Coast Guardsmen) 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. Personnel then train within their warfare community at sea in preparation for deployment (Navy 2024d).

A more detailed description of each of the seven primary mission areas, and associated activities or exercises is provided in the HCTT BA (Section 2.2, pp. 2-28 through 2-31), which we incorporate into this opinion by reference (Navy 2024d).

While most activities proposed by the Action Proponents are categorized into one of these primary mission areas, other activities (i.e., that don't fall within one of these primary mission areas) include:

- Aerial Firefighting
- At-Sea Vessel Refueling Training
- Combat Swimmer/Diver Training and Certification
- Kilo Dip
- Multi-Domain Unmanned Autonomous Systems
- Port Damage Repair
- Ship-to-Shore Fuel Transfer System Training
- Precision Anchoring
- Underwater Survey
- Search and Rescue
- Submarine Navigation
- Submarine Sonar Maintenance and Systems Checks
- Submarine Under Ice Certification

- Submarine and Unmanned Undersea Vehicle (UUV) Subsea and Seabed Warfare Exercise
- Surface Ship Sonar Maintenance and Systems Checks
- Training and End-to-End Mission Capability Verification – Subsea and Seabed Warfare Kinetic Effectors
- Training and End-to-End Mission Capability Verification – Unmanned Aerial Vehicle (UAV)
- Unmanned Aerial System Training and Certification – Submarine
- Unmanned Underwater Vehicle Training - Certification and Development
- Waterborne Training
- Acoustic and Oceanographic Research
- Air Platform Shipboard Integration Test
- Undersea Range System Test (Navy 2024d)

2.2 Proposed Military Readiness Activities

While the Action Proponents have been conducting military readiness activities in the HCTT action area for decades, the tempo and types of training and testing activities have changed over time due to 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 military readiness activities. The types and numbers of activities proposed by the Action Proponents 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.

In addition to the military readiness activities of the Navy (including Marine Corps) and Coast Guard, the proposed action also includes a limited subset of Army and Air Force activities with effects at sea as a result of activities conducted in the Hawaii portion of the action area. Military readiness activities include three separate groups of activities: training (Section 2.2.1), testing (Section 2.2.2), and modernization and sustainment of ranges (Section 2.2.3).

2.2.1 Training Activities

The HCTT BA describes all of the Navy (Section 2.3.2, Table 2-10, pp 2-33 through 2-44), Coast Guard (Section 2.3.2, Table 2-11, pp. 2-45 through 2-46), Army (Section 2.3.2, Table 2-12, pp. 2-46 through 2-47), and Air Force (Section 2.3.2, Table 2-13, pp. 2-47 through 2-48) training activities and activity levels (by general location) as part of the proposed action. We incorporate these tables into this opinion by reference (Navy 2024d). A more detailed description of the sonar, munitions, targets, systems, and other material used during training activities can be found in Appendix A *Activity Descriptions* of the HCTT Final EIS/Overseas EIS (Navy 2025d).

While most HCTT training exercises are conducted by a single military unit, the proposed action also includes major training exercises (MTEs), and integrated and coordinated exercises, which are comprised of multiple "unit-level" exercises conducted by several units operating together under the command and control of a single commander. Integrated or coordinated anti-submarine warfare exercises are generally on a smaller scale than MTEs, are shorter in duration, use fewer assets, and use fewer hours of hull-mounted sonar per exercise. Coordinated training exercises involve multiple units working together to meet unit-level training requirements, whereas integrated training exercises involve multiple units working together for deployment (Navy 2024d).

2.2.2 Testing Activities

Testing activities occur in response to emerging science or fleet operational needs. 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 (e.g., missiles, radar, and sonar) and platforms (e.g., surface ships, submarines, and aircraft); and acquisition of systems and platforms to support Navy missions. 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. The individual commands within the research and acquisition community included in the proposed action are Naval Air Systems Command (NAVAIR), Naval Facilities Engineering and Expeditionary Warfare Center (EXWC), Naval Sea Systems Command (NAVSEA), Naval Information Warfare Systems Command (NAVWAR), and the Office of Naval Research (ONR).

NAVAIR testing activities support its mission to provide full life cycle support of naval aviation aircraft, weapons, and systems to be operated by the Action Proponents (Navy 2024d). NAVAIR activities closely follow fleet primary mission areas, such as the testing of airborne mine warfare and anti-submarine warfare weapons and systems. NAVAIR activities include, but are not limited to, the testing of new aircraft platforms, weapons, and systems that have not yet been integrated into the fleet and Coast Guard (Navy 2024d). In addition to testing new platforms and weapon systems, most aircraft and weapon systems that have been integrated into the fleet also require follow-on testing throughout their lifecycle in conjunction with maintenance and upgrades, such as software revisions, to ensure that they function as designed. Table 2-14 of the HCTT BA (Section 2.3.3.1, pp. 2-49 through 2-53) describes the proposed NAVAIR testing activities, activity levels, and general location of these activities within the action area. We incorporate this table here by reference (Navy 2024d).

EXWC provides research, development, testing, and evaluation, as well as in-service engineering and lifecycle management for the shore, oceans, and expeditionary domains. EXWC's proposed activities include ocean energy and cable systems research; undersea range system testing; and underwater search, deployment, and recovery. Table 2-15 of the HCTT BA (Section 2.3.3.2, p.

2-54) describes the proposed EXWC testing activities, activity levels, and general location of these activities within the action area. We incorporate this table here by reference (Navy 2024d).

NAVSEA activities are aligned with its mission of new ship construction, life cycle management, and weapon systems development (Navy 2024d). NAVSEA activities include pierside and at-sea testing of ship systems, including sonar, acoustic countermeasures, radars, launch systems, weapons, unmanned systems, and radio equipment; tests to determine how the ship performs at sea (sea trials); developmental and operational test and evaluation programs for new technologies and systems; and testing on all ships and systems that have undergone overhaul or maintenance (Navy 2024d). At-sea test firing of shipboard weapon systems, including guns, torpedoes, and missiles, is also proposed as part of NAVSEA testing activities. Table 2-16 of the HCTT BA (Section 2.3.3.3, pp. 2-55 through 2-58) describes the proposed NAVSEA testing activities, activity levels, and general location of these activities within the action area. We incorporate this table here by reference (Navy 2024d).

NAVWAR is the information warfare systems command for the Navy. The mission of NAVWAR is to identify, develop, deliver, and sustain information warfare capabilities and services that enable naval, joint, coalition, and other national missions operating in warfighting domains from seabed to space; and to perform such other functions and tasks as directed (Navy 2024d). Table 2-17 of the HCTT BA (Section 2.3.3.4, p. 2-59) describes the proposed NAVWAR testing activities, activity levels, and general location of these activities within the action area. We incorporate this table here by reference (Navy 2024d).

ONR manages the Navy's basic, applied, and advanced research. ONR is also a parent organization for the Naval Research Laboratory, which operates as the Navy's corporate research laboratory and conducts a broad multidisciplinary program of scientific research and advanced technological development (Navy 2024d). Testing activities conducted by ONR and the Naval Research Laboratory include acoustic and oceanographic research, unmanned underwater vehicle research, and next generation mine countermeasures research. Table 2-18 of the HCTT BA (Section 2.3.3.5, p. 2-60) describes the proposed ONR testing activities, activity levels, and general location of these activities within the action area. We incorporate this table here by reference (Navy 2024d).

2.2.3 Modernization and Sustainment of Ranges

Technological advancements and improved weapons and systems capabilities often require modifications to training and testing ranges. In addition, existing components of the ranges require maintenance or replacement as they come to the end of their service life. These modernization and sustainment activities were not included in the HSTT Phase III consultation, but are part of the HCTT Phase IV action.

Southern California Offshore Anti-Submarine Range Modernization

The proposed action includes upgrading the existing Southern California Offshore Anti-Submarine Warfare Range (SOAR) underwater tracking and communication range with new hydrophones. This modernization includes the installation of undersea cables integrated with hydrophones and underwater telephones to sustain the capabilities of the SOAR. The new system would provide sufficient instrumentation to maintain the full underwater telephone coverage and hydrophone tracking capability within the existing approximately 670 square nautical miles (nmi^2) instrumented area (Navy 2024d). There are two types of sensors that will be installed during this modernization: hydrophones and underwater telephones. Hydrophones are used to convert acoustic energy into electrical energy, and to receive and record sounds on the range. Underwater telephones are used for two-way underwater communications (Navy 2024d).

The SOAR Modernization would consist of installing new arrays and leaving the old arrays in place. All arrays will connect to the existing nearshore junction box, and no additional onshore construction is anticipated. From the underwater junction box, the existing single trunk cable will connect to the onshore cable termination shelter via Sea Shore Interface components, including four horizontal directionally drilled conduits and their underwater exit points. Only one of the four conduits is used for SOAR cables, while the other three are unused and contain only a corrosion inhibitor solution. These existing components (junction box, trunk cable, and cable termination shelter) will remain in use and will not be replaced. Only the arrays, nodes, and hydrophones will be replaced. The seafloor within SOAR is mostly flat and of constant depth, with little if any underwater obstructions or seafloor anomalies. The majority of the deep water cable and node deployment would be in the overseas environment, beyond 12 nm from shore (Navy 2024d).

The array cables would follow the same existing general path, avoiding steep terrains. The spreading out or turning of cables is planned to occur at a constant depth, thus avoiding placing any cables at angles to the downward slope. The underwater junction box is equivalent to a distribution panel, connecting the trunk cable to the arrays (Navy 2024d). Internode cable is defined as the cable sections from the junction box to the first node and all subsequent sections between nodes. The design of the internode cable would be similar to that of standard submarine telecommunication cables, including up to six fibers encased in multiple layers of protective sheathing steel, nylon, or high-density polyethylene, and water blocking material, plus a copper conductor. The cable from the junction box to the first node would be double-armored and approximately two inches in diameter.

The deployment of the cables would utilize conventional cable-laying machinery, including a linear cable engine and cable pans. The new SOAR cables may follow the existing cable routes as the sensors are deployed. It is expected that the new cables would lie across existing cables as they are deployed. Due to the configuration of nodes on an array (some of the existing nodes are individually cabled), the cable routes in water depths greater than about 4,000 feet (ft; deep

water) would not be exactly the same as the existing arrays. During cable laying, vessels move very slowly, typically 1 to 3 knots (Navy 2024d).

During the offshore deployment of the arrays, the bi-directional and uni-directional nodes would be tested to ensure that the sensors are functional. After the installation is complete, a positional survey would be performed to locate the exact geodetic locations of the sensors on the range. The verification equipment used for the deployment verification and for the survey would use existing SOAR resources that are routinely used during normal range maintenance activities (Navy 2024d).

Shallow Water Training Ranges Installation

The proposed action includes installation of two Shallow Water Training Ranges (SWTRs) as an extension to the SOAR. The SWTR instrumentation would be used to evaluate the performance of participants in shallow water training exercises. Tanner Bank SWTR would encompass an area of 388 nm^2, and San Clemente Island (SCI) SWTR would encompass an area of 129 nm^2. When installed, the underwater instrumentation would increase the use of these areas for ASW training involving mid-frequency active sonar (Navy 2024d).

The SWTR instrumentation would consist of a system of undersea telecommunication cables, referred to as array cables, arranged on the seafloor and connecting a series of nodes. Each node may contain one or more transducers, which enable the transmission of sound; or a hydrophone, which receives sound and converts it into an electrical signal. The array cables would be connected to an existing underwater junction box close to shore and pulled through existing bores on the western side of SCI. Each range would require a new trunk cable and a new junction box; thus, the installation of three trunk cables and three junction boxes would be a part of the proposed action (Navy 2024d).

The SWTR extension would consist of sensor nodes spread on the ocean floor over a 500 nm^2 area. Sensor nodes are small spherical shapes of less than six inches in diameter that would be either suspended up to 15 ft in the water column or lie flat on the seafloor. Sensor nodes located in shallow water with a presence of commercial fishing activity would have an additional protective device surrounding or overlaying a sensor that would be 3 to 4 ft, round or rectangular, with a shallow height. Sensor nodes would be connected to each other by interconnect telecommunications cable with a diameter of less than one inch (Navy 2024d).

A series of sensor nodes would be connected via the interconnect cable to underwater junction boxes located in diver-accessible water depths. A junction box is rectangular in shape with dimensions of 10 to 15 ft on each side. The junction boxes would connect to a shore-based facility via trunk cables (submarine cables up to 2-inch diameter with additional data capacity). The trunk cables eliminate the need to have numerous interconnect cables running to shore. Up to eight trunk cables with a combined length of 375 nm would be employed. Trunk cables would be protected in the seashore area by horizontal directionally drilled pipes running beneath the

shoreline (Navy 2024d). The interconnecting cables and trunk cables would be deployed using a ship. The trunk cable paths would be routed through the deep water as much as is possible.

Sustainment of Undersea Ranges

Undersea ranges provide essential mission readiness capabilities. Range sustainment includes maintenance of systems and associated components. Maintenance may include, but is not limited to inspections, system replacement to extend service life (e.g., anodes and clamps), replacement of corrosion inhibitor solutions, and catastrophic repairs. Sustainment activities at undersea training ranges may require the use of divers, vessels, and unmanned underwater vehicles. Vessels may be required to anchor to the seafloor. Activities may take up to several weeks at a time.

During SOAR modernization and recurring maintenance, divers will replace old zinc anodes from the four existing electrical seafloor cable conduits and replace corrosion inhibitor solutions from the three unused conduits. To replace the zinc anodes, divers will remove a bolt, clean the area with a wire brush to ensure electrical connectivity, and then place a new zinc anode on the conduit. A Vapor Phase Corrosion Inhibitor (VpCI) solution is used in the conduits at a dilution up to 1.5% VpCI; 98.5% potable water (Navy 2024d). Solutions are effective for approximately 24 months after which time they would need to be replaced again (2-year cycle). To replace corrosion inhibitor solutions, divers will open the valve on the underwater termination point of each conduit and new corrosion inhibitor solution will be pumped into the conduits at the cable vaults. For three conduits with the solution, there is approximately 6,000 gallons of total solution (i.e., about 2,000 gallons per conduit) that could be released up to three times every seven years. For each event, it is estimated this work can be completed in approximately one week during daytime hours. The proposed release of solution would occur approximately 3,000 ft offshore and in water depths of approximately 90 ft (Navy 2025k). Steel clamps may be used in shallow water areas to secure cable infrastructure to the seafloor. Underwater drilling (using hand-held hydraulic drills) may be required to replace the clamps if large-scale maintenance is necessary (e.g., after storm destruction).

Deployment of Seafloor Cables

The Navy proposes to deploy fiber optic cables along the seafloor south and west of SCI in the California Study Area[3]. The proposed activity would increase the reach of the existing cable by installing an additional 373 miles of fiber optic cable in deep water (typically greater than 1,500 ft deep) approximately 100 nm from SCI. All installation and maintenance of cables would occur in the water, connecting to existing in-water infrastructure (Navy 2024d).

[3] The Navy also included two fiber optic cable deployments within the Hawaii portion of the action area (northeast of Oahu and west of Kauai) in their HCTT EIS/OEIS. However, the Navy has decided not to include these actions in this opinion, but rather will consult on them through separate ESA section 7 consultations with PIRO.

Installation and Maintenance of Mine Warfare and Other Training Areas

As part of the proposed action, support crews will deploy, move, and retrieve mine target shapes, sometimes referred to as mine countermeasure (MCM) targets. Mine targets could be placed on the seafloor (bottom targets) or tethered to anchors that are on the seafloor (moored). Targets are primarily inert and non-explosive, and emulate real world threats with a variety of sizes and shapes including spheres, cylinders, clamshells, and truncated cones. Minefields and mine training areas occur from the very shallow water (0-40 ft) to deep water (>500 ft).

MCM targets will be replaced every 1–2 years. The shape and mooring line would be retrieved for refurbishment and redeployed with a new anchor nominally once per year. The concrete anchors would typically be abandoned in place on the bottom after each installation (Navy 2024d). Moored shapes are typically deployed in two stages. The shape (with attached mooring tether) is first lowered into the water and released to float on the surface. The vessel then positions over the installation site and releases the anchor to settle to the bottom. As the anchor falls, it pulls any slack out of the mooring line and then pulls the shape under. Bottom shapes are initially lowered into the water by crane, then released to settle to the bottom. For underwater detonation training, individual target mines are inserted either by small boat, by diver, or both, depending on the training scenario (Navy 2024d).

The location of existing and proposed minefields are provided in Section 2.3.4.5 (pp. 2-73 through 2-81, Figures 2-23 through 2-28) of the HCTT Phase IV BA (Navy 2024d), which we incorporate here by reference.

Installation and Maintenance of Underwater Platforms

As part of the proposed action, underwater landing platforms will be installed in SOCAL RC and HRC to facilitate underwater vehicle pilot proficiency training. In HRC the platform location will be the same as a similar platform that was removed in 2009. The platform in SOCAL RC will be located just west of the Silver Strand Training Complex boat lanes, and the location in HRC is south of the entrance to Pearl Harbor within the Naval Defense Sea Area. Situated in the non-restrictive, flat, sandy seafloor training areas, the platforms will be permanently mounted, but removable for maintenance. The landing platforms will be approximately 40 ft by 20 ft and stand 15 ft high, with a weight of approximately 16 tons, situated at a depth between 60 and 100 ft. Prior to the installation, numerous pre-poured concrete blocks will be installed in a pre-surveyed area to create a positive anchor point to keep the platforms stationary (Navy 2024d).

The landing platform will require routine inspections, which will be accomplished by divers prior to each training evolution and during transponder installations. Each platform will be preserved in an anti-fouling coating that is similar to the bottom of a surface ship. Furthermore, a floating crane will be used approximately every five years to remove each platform from the ocean floor and then be taken to a ship repair facility to accomplish in-depth structural inspections, repairs, and preservation. Upon completion the platform will be returned and installed to their approved locations (Navy 2024d).

2.3 Foreign Military Activities

In furtherance of national security objectives, foreign militaries may participate in multinational training and testing events in the HCTT action area, including the biennial Rim of the Pacific Exercise (RIMPAC).

Foreign military vessel and aircraft activities within the HCTT action area that are planned by and under the substantial control and responsibility of the U.S. Navy are included in the proposed action (Navy 2025d). However, when foreign military vessels and aircraft operate independently within the action area as sovereign vessels and aircraft outside the planning, control, and responsibility of the Action Proponents, those activities are not considered part of the proposed action (Navy 2025d), nor would the resulting stressors be considered a consequence of the proposed action. There are many reasons why foreign military vessels may traverse U.S. waters or come into U.S. port, not all of which are at the request of any of the Action Proponents. Foreign military vessels and aircraft operate pursuant to their own national authorities and have independent rights under customary international law, embodied in the principle of sovereign immunity, to engage in various activities on the world's oceans and associated air space (Navy 2025d).

We evaluate the potential effects of foreign military vessel and aircraft activity and resulting stressors on ESA-listed species critical habitat only when such vessels and aircraft are actively participating in U.S. Navy-led training or testing exercises within the HCTT action area, and under the substantial control and responsibility of the U.S. Navy. For purposes of this opinion, the potential effects of foreign military vessel and aircraft activity within the HCTT action area when such vessels and aircraft are not actively participating in U.S. Navy-led training or testing exercises will either be considered part of the environmental baseline (i.e., for past or present activity) or part of our cumulative effects analysis (i.e., for future activity). Therefore, the effects of foreign military vessels and aircraft operating in any capacity on ESA-listed species or critical habitat in the action area are fully considered and evaluated in this opinion.

Other than scenarios when a foreign military vessel or aircraft actively participates in a HCTT U.S. Navy exercise, foreign military vessels and aircraft operating within the action area may adhere to their own standard operating procedures and environmental mitigation measures. When a foreign military vessel or aircraft participates in a U.S. Navy exercise within the U.S. territorial sea (i.e., 0 to 12 nm from shore), the U.S. Navy will request that the foreign vessel or aircraft follow the U.S. Navy's mitigation measures for that particular event. When a foreign military vessel or aircraft participates in a U.S. Navy exercise beyond the U.S. territorial sea but within the U.S. Exclusive Economic Zone, the U.S. Navy will encourage the foreign vessel or aircraft to follow the U.S. Navy's mitigation measures for that particular event (Navy 2022b; Navy 2022c). In either scenario (i.e., both within and beyond the territorial sea), U.S. Navy range operators will provide the foreign vessel or aircraft with the appropriate mitigation so that they can follow the mitigation measures (Navy 2021). Since the Action Proponents have indicated that

compliance with such measures cannot be legally mandated on a foreign entity, compliance by foreign vessels and aircraft isn't assumed to reach our effects determinations and conclusions. However, a high level of voluntary compliance is anticipated given the context (i.e., a foreign ally that is invited to participate with the U.S. Navy in a formal military readiness exercise) and past experience.

Foreign military vessels account for about 0.4% of total vessel activity (including non-military vessels) both in terms of numbers of ships and time spent in the action area (Starcovic and Mintz 2021). From 2014-2018, foreign military vessels averaged 4,200 hours per year spent within HSTT, compared to 26,800 hours average per year for U.S. Navy vessels. By range complex, about 55% of foreign military vessel hours occurred in the Hawaii Range Complex (HRC) and about 45% in SOCAL RC. The quantity of time foreign vessels are present within the action area is an over estimation of foreign participation in U.S. Navy-led training and testing activities because, as discussed above, a foreign military vessel may have other reasons for being in this area (e.g., transit through, independent steaming, port visit).

The participation level of foreign military vessels in U.S. Navy-led training or testing events within the HRC and within SOCAL RC differs greatly between RIMPAC and non-RIMPAC years. For example, in 2019 (a non-RIMPAC year), there was 0.1 foreign navy at-sea days (i.e., 1 day = 24 hours) within HRC and 20 foreign navy at-sea days within SOCAL RC in 2019 (Navy 2021). Out of 56 U.S.-led training events in 2019, four involved foreign navy vessels, with an average time per event of 8.7 hours. Foreign ships typically visit SOCAL RC for an extended period of time (i.e., up to 30 days) often associated with a Major Training Exercise (MTE) such as COMPTUEX, or multiple individual training events. For example, in 2019, one foreign ship participating in COMPTUEX accounted for 56% of the total foreign vessel training at sea days in SOCAL RC (Navy 2021).

In the months surrounding a scheduled RIMPAC event, the level of foreign military vessel activity in U.S. Navy-led events increases greatly in the HRC due to this multi-national training exercise. For example, in 2020, foreign vessels participating in U.S. Navy-led events accounted for 32 at-sea days in the HRC from August through September (some of this activity occurred after the RIMPAC exercise). RIMPAC 2022, which occurred from June 29th to August 4th, included 26 nations, 38 surface ships, three submarines, nine national land forces, more than 30 unmanned systems, approximately 170 aircraft, and over 25,000 participating personnel (Navy 2022a). During RIMPAC 2022, foreign vessels operated and/or transited through the HRC for 576 hours over 24 days.

By comparison, foreign military vessels did not participate in any U.S. Navy-led events in SOCAL RC during the 2020 RIMPAC year (Navy 2021). Participation by foreign nations in RIMPAC can vary from year to year, and 2020 participation was impacted by international COVID-19 concerns with fewer nations participating (Navy 2021). While most of the foreign surface ship and aviation assets were in the HRC, a subset of RIMPAC 2022 mine warfare

training events occurred in the SOCAL RC. There were a total of three foreign surface vessels within SOCAL RC waters between January and October of 2022 (Navy 2022a). Even in a RIMPAC year, the days at sea for foreign militaries engaged in a Navy-led training or testing activity accounts for a very small percentage compared to the U.S. Navy activities. For instance, the 2020 foreign military participation (a RIMPAC-year) was 1.5% of the U.S. Navy's average days at sea (i.e., 32 days out of an estimated 2,056 days at sea).

2.4 Standard Operating Procedures and Mitigation Measures

The primary purpose of the Action Proponents' standard operating procedures is to provide for safety and mission success. In some cases there are environmental benefits resulting from their implementation as well. Mitigation measures are designed specifically to avoid or reduce environmental impacts from the proposed activities. The standard operating procedures and mitigation measures the Action Proponents will incorporate in their training and testing activities in the action area are described below.

2.4.1 Standard Operating Procedures

Standard operating procedures have been developed by the Action Proponents over many years of experience conducting military readiness activities. Standard operating procedures are broadcast via numerous naval instructions and manuals to ensure compliance (Navy 2024d). For purposes of this consultation, the standard operating procedures that could result in conservation benefits to ESA-listed species or their habitats are described in Table 2 below.

Table 2. Standard Operating Procedures Proposed for HCTT Phase IV with Potential Conservation Benefits to Endangered Species and Their Habitats

Procedure Focus	Procedure Description
Laser safety	• Laser systems are approved for fielding by the Action Proponents' Laser Safety Review Board or equivalent. • Only properly trained and authorized personnel operate high-energy lasers within designated areas.
In-water explosive safety	• In-water explosive activities are scheduled to occur in areas located away from popular recreational dive sites, primarily for human safety. • Most explosive events are conducted during daylight hours. • Weapon firing activities that involve small boats deploying or retrieving targets are typically conducted in Beaufort Sea state number 4 conditions or better to ensure safe operating conditions for the small boat operators.
Cable installation	• Vessels engaged in installation would contain sorbent booms and pads for use in the unlikely event of a fuel spill, and would adhere to all Navy and Coast Guard requirements regarding the containment, cleanup, and reporting of spills. • To prevent any potential impacts to ESA-listed species during cable anchoring activities in SOCAL RC, divers would not place an anchor or the cable between the anchors within 3 ft of any abalone species. • Contractor vessels engaged in cable installation would follow Navy vessel movement mitigation measures.

Procedure Focus	Procedure Description
Invasive species	• All physical contact and disturbance to the benthos and any invasive species present shall be prevented whenever possible. • Any equipment, gear, or material used in water with known invasive species, shall be dried for 48 hours before moving to an uncontaminated area. • No movement/removal of benthos substrate, water, or invasive species itself from a known invasive species infested area to an uncontaminated area shall take place. Any removal of substrate or invasive species shall be properly disposed of so that it cannot spread to uncontaminated areas. • In-water equipment will be locally sourced thus reducing the risk of introducing non-native species. If any equipment must be brought to the project site from outside the Hawaiian Islands region, then the appropriate prevention measures (e.g., wash-down or hull cleaning triple flush procedures) will be included in the work plan. • Ballast water exchange during Military Readiness activities will comply with the Navy's Environmental Readiness Program Manual (OPNAV M-5090.1). • Military Readiness activities will be consistent with installation INRMPs designed to ensure, to the maximum extent possible, aquatic invasive species are not introduced into near-shore environments or bodies of water on or adjacent to the installation (OPNAV M-5090.1). • For the California Study Area, the Navy will comply, to the maximum extent possible, with the National Oceanic and Atmospheric Administration Caulerpa Control Protocol (National Marine Fisheries Service, 2021). • All Navy and USCG vessels undergo routine inspections and periodic hull cleanings. • Prior to entering port, Navy and USCG vessels undergo inspections as part of the ships' pest control program.
Visibility requirements during aircraft activities	• Aircrew are not authorized to deploy ordnance through extensive cloud cover where visual clearance for non-participants is not possible. The two exceptions to this requirement are (1) when operating in the open ocean, clearance for non-participating aircraft and vessels 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.
Safe vessel operation	• Vessels are required to operate in accordance with applicable navigation rules, including Inland Waters Navigation Rules (33 CFR section 83.01 et seq.) and International Regulations for Preventing Collisions at Sea (72 COLREGS). These rules and regulations were formalized in the Convention on the International Regulations for Preventing Collisions at Sea (1972) and implemented through the International Navigational Rules Act of 1977 (33 U.S.C. sections 1601–1608). Applicable navigation requirements specified in the Inland Navigation Rules include, but are not limited to, Rule 5 (Lookouts) and Rule 6 (Safe Speed). These rules require that vessels, at all times, proceed at a safe speed so proper and effective action can be taken to avoid collision and so vessels can be stopped within a distance appropriate to the prevailing circumstances and conditions. • Surface ships transit at speeds that are optimal for fuel conservation, to maintain ship schedules, and to meet mission requirements. Vessel captains use the totality of the circumstances to ensure the vessel is traveling at appropriate speeds in accordance with navigation rules. Depending on the circumstances, this may involve adjusting speeds during periods of reduced visibility or in certain locations. • The Action Proponents also avoid known navigation hazards that appear on nautical charts, such as submerged wrecks and obstructions. • With limited exceptions (e.g., amphibious vessels operating in designated locations, bottom-crawling vehicles), manned vessels and unmanned vehicles avoid contact with the seafloor as a standard collision avoidance procedure to prevent damage to the platforms.

Procedure Focus	Procedure Description
Lookouts	• Underway surface ships operated by or for the Action Proponents have personnel assigned to stand watch at all times (day and night) for safety of navigation, collision avoidance, range clearance, and man-overboard precautions. • Personnel on underway small boats (e.g., crewmembers responsible for navigation) fulfill similar watch standing responsibilities to those positioned on surface ships. Standard watch personnel, also referred to as "Lookouts," include officers, enlisted personnel, and civilians operating in similar capacities. • The Navy updated its standard operating procedures related to vessel safety to incorporate revised procedures regarding Lookouts for certain ship classes as per the 2021 Surface Ship Navigation Department Organization and Regulations Manual (NAVDORM; Navy 2021). For example, the 2021 NAVDORM requires the use of three Lookouts on Navy cruisers and destroyers as compared to the previous requirement of one Lookout when a vessel was underway and not engaged in sonar training or testing. Although the NAVDORM does not require three Lookouts on aircraft carriers for distances greater than 10 NM from shore, standard operating protocol mandates six Lookouts (i.e., fore, aft, port bridge, starboard bridge, flag bridge, and signal). • See Section 2.4.2 below for more details on the use of Lookouts for mitigation.
Pile driving	• Due to pile driving system design and operation, the Navy performs soft starts standard operating procedures during impact installation of each pile to ensure proper operation of the diesel impact hammer. During a soft start, the Navy performs an initial set of strikes (three, three-blow sets) from the impact hammer at reduced energy before it can be operated at full power and speed. Each three-blow set will be separated by at least 30 seconds.
Unmanned vehicle procedures	• Unmanned surface vehicles or unmanned underwater vehicles that operate autonomously may have embedded sensors designed for avoidance of large objects. For example, select unmanned vehicles have sensors, such as a forward-looking sonar (FLS), to perform obstacle avoidance. The FLS makes detections at a sufficient range for the onboard processor to determine if there is a need for an avoidance maneuver. If there is a need for an avoidance maneuver, the onboard vehicle control system would insert a new maneuver (in place of the currently executing activity) and continue to introduce new maneuvers if detections continue to be made. There are a number of possible maneuvers that could be implemented, from adjusting heading to stopping or hovering the vehicle. • As an additional standard collision avoidance procedure during specific stages of training or testing (e.g., during an initial training and testing phases), manned support vessels may escort unmanned surface vehicles and unmanned underwater vehicles. Lookouts on the support vessels may use radios to communicate with other vessels operating in the vicinity to coordinate safe maneuvering (e.g., communicating the positioning and safety distances for avoiding collisions with unmanned vehicles). • As a standard collision avoidance procedure for in-water devices towed by surface vessels (or by unmanned surface vehicles or unmanned underwater vehicles under positive control by manned support vessels), the Navy searches the intended path of the towed in-water device for floating debris, concentrations of floating vegetation, floating objects, or animals with potential to obstruct, tangle, or damage the device.

2.4.2 Mitigation Measures[4]

The Action Proponents propose to implement mitigation measures to avoid or reduce potential impacts from acoustic, explosive, and physical disturbance and strike stressors from military readiness activities on ESA-listed species in the action area. NMFS considers these measures as reasonably certain to be implemented and thus components of the proposed action. Mitigation measures fall into two main categories: activity-based mitigation and geographic mitigation.

The Action Proponents will publish, broadcast, disseminate, or distribute mitigation instructions through pre-event briefs, governing instructions, broadcast messages, the Protective Measures Assessment Protocol (PMAP), or other established internal processes. The PMAP is a software program accessed by appointed Action Proponent personnel during pre-event planning. The program provides operators with notification of the required mitigation measures applicable to a particular training or testing event, as well as a visual display of the planned event location overlaid with relevant environmental data. Its text and mapping data will be updated to align with best available science and the final mitigation that results from this consultation.

Environmental awareness and education training is also provided to personnel through the Afloat Environmental Compliance Training program or equivalent. Upon reporting aboard and annually thereafter, appointed personnel must complete training identified in their career path training plan, which may include the following training modules:

- *Introduction to Afloat Environmental Compliance* - provides information on at-sea environmental laws, regulations, and compliance roles.
- *Marine Species Awareness Training* - provides information on marine species sighting cues, visual observation tools and techniques, and sighting notification procedures.
- *PMAP Training* - provides information on how personnel should access and operate the PMAP.
- *Sonar Positional Reporting System and Marine Mammal Incident Reporting* - provides information on sonar reporting requirements and marine mammal incident reporting procedures.

Mitigation requirements are mandatory for the Action Proponents when conducting activities under the proposed action. As discussed in Section 2.3, foreign militaries may participate alongside the Action Proponents in multinational training and testing events in the action area. While compliance with HCTT mitigation measures cannot be legally mandated on a foreign entity, we anticipate a relatively high level of voluntary compliance. See Section 2.3 for more

[4] We consider these mitigation measures "conservation measures": actions that will be taken by the Action Proponents 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.

details on the implementation of HCTT mitigation measures by foreign military vessels and aircraft participating in Navy-led events.

The following sections summarize the mitigation measures that the Action Proponents propose to implement in association with the training and testing activities analyzed in this document. Most of these proposed mitigation measures are carried over from HSTT Phase III. A complete discussion of the mitigation measures, as well as measures considered but not proposed, and the evaluation process used by the Action Proponents to develop, assess, and select mitigation measures, can be found in the HCTT EIS/OEIS (Navy 2025d) for this action (see https://www.nepa.navy.mil/hctteis/).

Activity-based Mitigation

Activity-based mitigation measures have a primary objective of reducing overlap of individual marine mammals and sea turtles in real time with stressors that have the potential to cause injury or mortality. Activity-based mitigation generally involves: (1) the use of one or more trained Lookouts to observe for specific biological resources within a mitigation zone[5]; (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.

Lookouts are personnel who perform similar duties as the standard watch personnel but have an additional duty of helping meet the mitigation requirements by visually observing mitigation zones for marine mammals and sea turtles. For some activities, Lookouts may also be required to observe for additional biological resources, such as birds, fish, jellyfish aggregations, or floating vegetation, which can be indicators of potential marine mammal or sea turtle presence.

Depending on the activity, a Lookout may be positioned on a surface ship or small 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 one of the Lookouts.

The Action Proponents have added a requirement (new for HCTT Phase IV) for additional personnel on the platform conducting an event, or on additional participating platforms, to serve as part of the Lookout Team for all acoustic, explosive, and physical disturbance and strike

[5] Mitigation zones are areas at the surface of the water (measured as the radius from a stressor) within which training or testing activities would be halted, powered down, or modified to protect specific biological resources from an injurious impact (e.g., Permanent Threshold Shift [PTS], vessel strike).

stressor mitigation categories (Navy 2024d). During HSTT Phase III, additional personnel were required to assist Lookouts for explosive events only. The *U.S. Navy Lookout Training Handbook was* updated in 2022 to include a more robust chapter on environmental compliance, mitigation, and marine species observation tools and techniques (NAVEDTRA 12968-E). The Action Proponents have also clarified that for HCTT Phase IV aircraft operating at high altitudes (i.e., altitudes at which marine mammals on the surface cannot be distinguished) are exempt from requirements to conduct activity-based mitigation.

Mitigation Specific to Acoustic Stressors, Explosives, and Non-Explosive Ordnance

The mitigation measures described below will be implemented (as appropriate) in response to an applicable sighting within or entering the relevant mitigation zone for acoustic stressors, explosives, and non-explosive practice munitions:

- Prior to the initial start of an event (or stressor use), the Action Proponents will: (1) relocate the event to a location where applicable species are not observed, or (2) delay the initial start of the event (or stressor use) until one of the "Mitigation Zone All-Clear Conditions" (discussed below) has been met.
- During the event (i.e., during use of a stressor) the Action Proponents will (until one of the Mitigation Zone All-Clear Conditions has been met): (1) power down or shut down active acoustic transmissions, (2) cease air gun use, (3) cease pile driving or pile removal, (4) cease weapon firing or ordnance deployment, (5) or cease explosive detonations or fuse initiations (Navy 2024d).

Mitigation Zone All-Clear Conditions indicate that the mitigation zone is determined to be free of applicable species. The conditions include (1) a Lookout observes the applicable species exiting the mitigation zone, (2) a Lookout determines the applicable species has exited the mitigation zone based on its observed course and speed relative to the mitigation zone, (3) a Lookout affirms the mitigation zone has been clear from additional sightings for an applicable "wait period," or (4) for mobile events, the stressor has transited a distance equal to double the mitigation zone size beyond the location of the last sighting. Wait periods were established because events cannot be delayed or ceased indefinitely for the purpose of mitigation due to impacts on safety, sustainability, and the ability to meet mission requirements (Navy 2024d).

For HCTT Phase IV the Action Proponents added or modified a few mitigation requirements for activity-based mitigation. Mitigation zones for broadband sources, which were not included in Phase III, have been added for this consultation. A 200 yard (yd) shut down mitigation zone would apply to broadband and other active acoustic sources less than 200 decibel (dB), while the tiered 1,000-yd power down/500-yd power down/200-yd shut down mitigation zones would apply to those sources greater than or equal to 200 dB (Navy 2024d). This requirement is meant to encompass new acoustic sources (e.g., sources used for oceanographic and acoustic research) that use a range of frequencies.

In addition, the air gun mitigation zone size has been increased from 150 yd to 200 yd for consistency with other active acoustic sources (Navy 2024d). HCTT Phase IV also includes new activity-based mitigation requirements for applicable events that involve Unmanned Surface Vehicles (USVs) and Unmanned Underwater Vehicles (UUVs; and the sources they use, tow, or deploy) that are already being escorted and operated under positive control by a manned surface vessel (Navy 2024d). New requirements are in place for "research-based sub-surface explosives" to account for new explosive events with research applications (e.g., acoustic and oceanographic research) that would use 0.1 pound (lb.) to 5-lb. net explosive weight (NEW).

Other changes or additions to activity-based mitigation requirements for Phase IV include: 1) for pile driving activities, reduction of the mitigation zone from 100 yards to 5 yards and of the wait time from 30 minutes to 15 minutes (Table 3); 2) new activity-based mitigation requirements for nets deployed for UUVs obstacle avoidance exercises; 3) a new requirement to observe for manta rays during Explosive Mine Neutralization (With Divers) activities (Table 4).

Activity-based mitigation will not apply for the following acoustic stressor sources: not operated under positive control; used for navigation safety; deployed by aircraft operating at high altitudes; from unmanned platforms except when escort vessels are already participating in the event; used by submerged submarines; de minimis sources; unattended sources such as moored buoys used for acoustic and oceanographic research; when marine mammals (e.g., dolphins) are determined to be intentionally swimming at the bow or alongside or directly behind the vessel, vehicle, or device; and above two kilohertz (kHz) for sea turtles (Navy 2024d). Mitigation will also not apply to explosives or non-explosive ordnance that are: deployed by aircraft operating at high altitudes; deployed by submerged submarines, except for explosive torpedoes; deployed against aerial targets and land-based targets; during vessel or shore-launched missile or rocket events; used at or below the de minimis threshold; and deployed by unmanned platforms except when escort vessels are already participating in the event and have positive control over the explosive or ordnance deployment (Navy 2024d).

Additional details on the activity-based mitigation requirements for acoustic stressors, explosive stressors, and non-explosive ordnance are described in Table 3, Table 4, and Table 5, respectively.

Table 3. Activity-based Mitigation for Acoustic Stressors (Navy 2024d)

Mitigation Category	*Mitigation Zones*	*Mitigation Requirements*	*Mitigation Requirement Timing*	*Wait Period*
Active Acoustic Sources				
Sources with both power down and shut down capabilities: • Low-frequency active sonar ≥200 dB • Mid-frequency active sonar (surface ship hull mounted, including surfaced submarines) • Broadband and other active acoustic sources >200 dB	• 200 yd from active acoustic sources (shut down) • 500 yd from active acoustic sources (power down of 10 dB total) • 1,000 yd from active acoustic sources (power down of 6 dB total)	• One Lookout in/on one of the following: Aircraft Pierside, moored, or anchored vessel Underway vessel with space/crew restrictions (including small boats) Underway vessel already participating in the event that is escorting (and has positive control over sources used, deployed, or towed by) an unmanned platform • Two Lookouts on an underway vessel without space/crew restrictions • During the activity, Navy personnel conduct passive acoustic monitoring for marine mammals and use information from detections to assist visual observations.	• Immediately prior to the initial start of using active acoustic sources (e.g., while maneuvering on station), observe for: Marine mammals Sea turtles (for sources <2 kHz) Floating vegetation • During use of active acoustic sources, observe for: Marine mammals Sea turtles (for sources <2 kHz)	• 10 or 30 minutes (depending on fuel constraints of the platform)
Sources with shut down (but not power down) capabilities: • Low-frequency active sonar <200 dB • Mid-frequency active sonar (sources not hull mounted on a surface ship) • High-frequency active sonar • Air guns • Broadband and other active acoustic sources <200 dB	• 200 yd from active acoustic sources (shut down)			
Pile Driving and Pile Removal				
• Vibratory and impact pile driving and removal	• 5 yd from piles being driven or removed (cease pile driving or removal)	• One Lookout on one of the following: Shore Pier Small boat	• 15 minutes prior to the initial start of pile driving or pile removal, observe for: Marine mammals Sea turtles Floating vegetation • During pile driving or removal, observe for: Marine mammals Sea turtles	• 15 minutes
Weapon Firing Noise				
• Explosive and non-explosive large-caliber gunnery firing noise (surface-to-surface and surface-to-air)	• 30 degrees on either side of the firing line out to 70 yd from the gun muzzle (cease fire)	• One Lookout on a vessel	• Immediately prior to the initial start of large-caliber gun firing (e.g., during target deployment), observe for: Marine mammals Sea turtles	• 30 minutes

			Floating vegetation • During large-caliber gun firing, observe for: Marine mammals Sea turtles	

Table 4. Activity-based Mitigation for Explosive Stressors (Navy 2024d; Navy 2025e)

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
Explosive Bombs				
• Any NEW	• 2,500 yd from the intended target (cease fire)	• One Lookout in an aircraft	• Immediately prior to the initial start of bomb delivery (e.g., when arriving on station), observe for: Marine mammals and sea turtles Floating vegetation • During bomb delivery, observe for: Marine mammals and sea turtles • After the event, when practical, observe the detonation vicinity for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 10 minutes
Explosive Gunnery				
• Air-to-surface medium-caliber	• 200 yd from the intended impact location (cease fire)	• One Lookout on a vessel or in an aircraft	• Immediately prior to the initial start of gun firing (e.g., while maneuvering on station), observe for: Marine mammals and sea turtles Floating vegetation • During gunnery firing, observe for: Marine mammals and sea turtles • After the event, when practical, observe the detonation vicinity for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 10 or 30 minutes (depending on fuel constraints of the platform)
• Surface-to-surface medium-caliber	• 600 yd from the intended impact location (cease fire)			
• Surface-to-surface large-caliber	• 1,000 yd from the intended impact location (cease fire)			
Explosive Underwater Demolition Multiple Charge – Mat Weave and Obstacle Loading				

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
• Any NEW	• 700 yd from the detonation site (cease fire)	• Two Lookouts: one on a small boat and one on shore from an elevated platform	• For 30 min. prior to the first detonation, the Lookout positioned on a small boat will observe for: Marine mammals and sea turtles Floating vegetation • For 10 min. prior to the first detonation, the Lookout positioned on shore will use binoculars to observe for: Marine mammals and sea turtles • During detonations, both Lookouts will observe for: Marine mammals and sea turtles • After the event, observe the detonation vicinity for 30 minutes for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 10 minutes (determined by the shore observer)
Explosive Mine Countermeasure and Neutralization (No Divers)				
• 0.1–5 lb. NEW	• 600 yd from the detonation site (cease fire)	• One Lookout on a vessel or in an aircraft	• Immediately prior to the initial start of detonations (e.g., while maneuvering on station; typically, 10 or 30 minutes depending on fuel constraints), observe for: Marine mammals and sea turtles Floating vegetation • During detonations or fuse initiation, observe for: Marine mammals and sea turtles Concentrations of seabirds or individual foraging seabirds • After the event, observe the detonation vicinity for 10 or 30 minutes (depending on fuel constraints), for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 10 or 30 minutes (depending on fuel constraints of the platform)
• >5 lb. NEW	• 2,100 yd from the detonation site (cease fire)	• Two Lookouts: one in a small boat and one in an aircraft		
Explosive Mine Neutralization (With Divers)				
• 0.1–20 lb. NEW (positive control)	• 500 yd from the detonation site (cease fire)	• Two Lookouts in two small boats (one Lookout per boat), or one	• Time-delay devices will be set not to exceed 10 minutes • Immediately prior to the initial start of detonations or fuse initiation for positive control events (e.g., while maneuvering on station) or for 30 minutes prior for time-delay events, observe for: Marine mammals and sea turtles	• 10 or 30 minutes (depending on fuel constraints

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
		small boat and one rotary-wing aircraft (with one Lookout each), and one Lookout on shore for shallow-water events	Manta rays and hammerhead sharks Floating vegetation • During detonations or fuse initiation, observe for the following: Marine mammals Sea turtles Manta rays Hammerhead sharks: Divers will notify the support boat or Range Safety Officer of sightings (of any hammerhead, due to difficulty in differentiating species). Detonations will cease if divers sight a hammerhead when setting charges and will recommence when it is no longer observed. Concentrations of seabirds or individual foraging seabirds in the water during shallow-water events: A shore-based Lookout will survey the mitigation zone with binoculars before and after each detonation. If events involve multiple detonations, the second (or third, etc.) detonation will occur immediately after the preceding detonation (i.e., within 10 seconds), or after 30 min. • When practical based on mission, safety, and environmental conditions: Boats will observe from the mitigation zone radius mid-point When two are used, boats will observe from opposite sides of the mine location Platforms will travel a circular pattern around the mine location Boats will have one Lookout observe inward toward the mine location and one observe outward toward the mitigation zone perimeter Divers will be part of the Lookout Team • After the event, observe the detonation vicinity for 30 minutes for dead or injured: Marine mammals and sea turtles Manta rays and hammerhead sharks • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	of the platform)
• 0.1–29 lb. NEW (time-delay) • >20–60 lb. NEW (positive control)	• 1,000 yd from the detonation site (cease fire)	• Four Lookouts in two small boats (two Lookouts per boat), and one additional Lookout in an aircraft if used in the event		
Explosive Missiles and Rockets				
• 0.6–20 lb. NEW (air-to-surface)	• 900 yd from the intended impact location (cease fire)	• One Lookout in an aircraft	• Immediately prior to the initial start of missile or rocket delivery (e.g., during a fly-over of the mitigation zone), observe for: Marine mammals and sea turtles Floating vegetation • During missile or rocket delivery, observe for:	• 10 or 30 minutes (depending on fuel constraints

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
• >20–500 lb. NEW (air-to-surface)	• 2,000 yd from the intended impact location (cease fire)		Marine mammals and sea turtles • After the event, when practical, observe the detonation vicinity for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	of the platform)
Explosive Sonobuoys and Research-Based Sub-Surface Explosives				
• Any NEW of sonobuoys • 0.1–5 lb. NEW for other types of sub-surface explosives used in research applications	• 600 yd from the device or detonation site (cease fire)	• One Lookout on a small boat or in an aircraft • Lookouts would use information from passive acoustic detections to inform visual observations when passive acoustic devices are already being used prior to the initial start of detonations	• Immediately prior to the initial start of detonations (e.g., during sonobuoy deployment, which typically lasts 20 to 30 minutes), observe for: Marine mammals and sea turtles Floating vegetation • During detonations, observe for: Marine mammals and sea turtles • After the event, when practical, observe the detonation vicinity for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 10 or 30 minutes (depending on fuel constraints of the platform)
Explosive Torpedoes				
• Any NEW	• 2,100 yd from the intended impact location (cease fire)	• One Lookout in an aircraft • Conduct passive acoustic monitoring for marine mammals; use information from detections	• Immediately prior to the initial start of detonations (e.g., during target deployment), observe for: Marine mammals and sea turtles Floating vegetation Jellyfish aggregations • During torpedo launches, observe for: Marine mammals and sea turtles Jellyfish aggregations • After the event, when practical, observe the detonation vicinity for dead or injured:	10 or 30 minutes (depending on fuel constraints of the platform)

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
		to assist visual observations	Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	
Ship Shock Trials				
• Any NEW	• 3.5 nm from the target ship hull (cease fire)	• On the day of the event, 10 observers (Lookouts and third-party observers combined), spread between aircraft or multiple vessels as specified in the event-specific mitigation plan	• The Navy will develop a detailed event-specific monitoring and mitigation plan in the year prior to the event and provide it to NMFS for review • Beginning at first light on days of detonation until the moment of detonation (as allowed by safety measures), observe for: Marine mammals and sea turtles Floating vegetation Jellyfish aggregations Large schools of fish Flocks of seabirds • If an incident involving a marine mammal or sea turtle is observed after an individual detonation, the Navy will follow established incident reporting procedures and halt any remaining detonations until the Navy can consult with NMFS and review or adapt the event-specific mitigation plan, if necessary • During the 2 days following the event at a minimum and up to 7 days at a maximum, and as specified in the event-specific mitigation plan, observe the detonation vicinity for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	• 30 minutes
SINKEX				
• Any NEW	• 2.5 nm from the target ship hull (cease fire)	• Two Lookouts: one on a vessel and one in an aircraft • Conduct passive acoustic monitoring for marine mammals; use information	• During aerial observations for 90 minutes prior to the initial start of weapon firing, observe for: Marine mammals and sea turtles Floating vegetation Jellyfish aggregations • From the vessel during weapon firing, and from the aircraft and vessel immediately after planned or unplanned breaks in weapon firing of more than two hours, observe for: Marine mammals and sea turtles	• 30 minutes

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
		from detections to assist visual observations	• Observe the detonation vicinity for two hours after sinking the vessel or until sunset, whichever comes first, for dead or injured: Marine mammals and sea turtles • If a marine mammal is visibly injured or killed as a result of detonation, explosives use in the event shall be suspended immediately, and established incident reporting procedures shall be followed	

Table 5. Activity-based Mitigation for Non-Explosive Ordnance (Navy 2024d)

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
Non-Explosive Aerial-Deployed Mines and Bombs				
• Explosive aerial-deployed mines[6] • Non-explosive aerial-deployed mines • Non-explosive bombs	• 1,000 yd from the intended target (cease fire)	• One Lookout in an aircraft	• Immediately prior to the initial start of mine or bomb delivery (e.g., when arriving on station), observe for: Marine mammals Sea turtles Floating vegetation • During mine or bomb delivery, observe for: Marine mammals Sea turtles	• 10 minutes
Non-Explosive Gunnery				
• Non-explosive surface-to-surface large-caliber ordnance • Non-explosive surface-to-surface and air-to-surface	• 200 yd from the intended impact location (cease fire)	• One Lookout on a vessel or in an aircraft	• Immediately prior to the initial start of gun firing (e.g., while maneuvering on station), observe for: Marine mammals Sea turtles Floating vegetation	• 10 or 30 minutes (depending on fuel constraints of the platform)

[6] Explosive aerial-deployed mines do not detonate upon contact with the water surface and are therefore considered non-explosive when mitigating the potential for a mine shape to strike a marine mammal or sea turtle at the water surface. Mitigation for the explosive component of aerial-deployed mines is described in Table 5

Mitigation Category	Mitigation Zones	Mitigation Requirements	Mitigation Requirement Timing	Wait Period
medium-caliber ordnance • Non-explosive surface-to-surface and air-to-surface small-caliber ordnance			• During gunnery firing, observe for: Marine mammals Sea turtles	
Non-Explosive Missiles and Rockets				
• Non-explosives (air-to-surface)	• 900 yd from the intended impact location (cease fire)	• One Lookout in an aircraft	• Immediately prior to the start of missile or rocket delivery (e.g., during a fly-over of the mitigation zone), observe for: Marine mammals Sea turtles Floating vegetation • During missile or rocket delivery, observe for: Marine mammals Sea turtles	• 10 or 30 minutes (depending on fuel constraints of the platform)

Mitigation Specific to Vessels, Vehicles, and Towed In-Water Devices

For ship classes required to maintain more than one Lookout, the specific requirement is subject to change over time in accordance with the applicable navigation instruction, such as the Surface Ship Navigation Department Organization and Regulations Manual. The Action Proponents will notify NMFS should their Lookout policies change, including in the Surface Ship Navigation Department Organization and Regulations Manual.

Mitigation will be implemented to the maximum extent practical based on the prevailing circumstances, including consideration of safety of vessels, unmanned vehicles, towing platforms, and crews, as well as maneuverability restrictions (Navy 2024d). Mitigation will not be implemented (1) by submerged submarines, (2) by unmanned vehicles except when escort vessels are already participating in the event and have positive control over the unmanned vehicle movements, (3) when marine mammals (e.g., dolphins) are determined to be intentionally swimming at the bow, alongside the vessel or vehicle, or directly behind the vessel or vehicle (e.g., to bow-ride or wake-ride), (4) when pinnipeds are hauled out on man-made navigational structures, port structures, and vessels, (5) when impractical based on mission requirements (e.g., during certain aspects of amphibious exercises), and (6) by manned surface vessels and towed in-water devices actively participating in cable laying during modernization and sustainment of ranges activities. Additional details on the activity-based mitigation requirements for vessels, unmanned vehicles, and towed in-water devices are described in Table 6.

Navy Lookouts will monitor the mitigation zones around underway USVs as mission and circumstances allow. These circumstances include, but are not limited to, the Lookout's location on the deck, the support vessel's position relative to the USV, the size of the USV, and the other responsibilities of the Lookout, such as monitoring the support vessel's mitigation zone and safe operation of the support vessel (Navy 2024d).

On September 24, 2024, we issued a Concurrence Letter and Conference for the Endangered Species Act Section 7 Informal Consultation for the Training and Testing of the Extra Large Unmanned Undersea Vehicles and Unmanned Surface Vehicles at Naval Base Ventura County, Port Hueneme, California (NMFS 2024b). This consultation covers Navy training and testing activities involving the use of UUVs and USVs off of Port Hueneme, California, through 2025. Continuation of these activities past 2025 is included as part of this HCTT Phase IV consultation. The informal consultation includes mitigation requirements for the deployment of nets for testing UUVs, as shown in Table 7. These mitigation measures are included as part of the HCTT Phase IV action (Navy 2024d).

Table 6. Activity-based Mitigation for Vessels, Vehicles, and Towed In-Water Devices

Mitigation Category	*Mitigation Requirements*	*Mitigation Zones and Requirements*
Manned Surface Vessels		
Manned surface vessels, including surfaced submarines	One or more Lookouts on manned underway surface vessels in accordance with the most recent navigation safety instruction	Immediately prior to manned surface vessels getting underway and while underway, the Lookout(s) will observe for: Marine mammals Sea turtles Underway manned surface vessels will maneuver themselves (which may include reducing speed) to maintain the following distances as mission and circumstances allow: 500 yd from whales 200 yd from other marine mammals Vicinity of sea turtles
Unmanned Vehicles		
Unmanned Surface Vehicles and Unmanned Underwater Vehicles already being escorted (and operated under positive control) by a manned surface vessel	One Lookout on a support vessel that is already participating in the event, and has positive control over the unmanned vehicle	Immediately prior to unmanned vehicles getting underway and while underway, the Lookout will observe for: Marine mammals Sea turtles A support vessel that is already participating in the event, and has positive control over the unmanned vehicle, will maneuver the unmanned vehicle (which may include reducing its speed) to ensure it maintains the following distances as mission and circumstances allow: 500 yd from whales 200 yd from other marine mammals Vicinity of sea turtles
Towed In-Water Devices		
In-water devices towed by an aircraft, a surface vessel, or an Unmanned Surface Vehicle or Unmanned Underwater Vehicle already being escorted (and operated under positive control) by a manned surface vessel	One Lookout on the manned towing vessel or aircraft, or on a support vessel that is already participating in the event and has positive control over an unmanned vehicle that is towing an in-water device	Immediately prior to and while in-water devices are being towed, the Lookout will observe for: Marine mammals Sea turtles Manned towing platforms, or support vessels already participating in the event that have positive control over an unmanned vehicle that is towing an in-water device, will maneuver itself or the unmanned vehicle (which may include reducing speed) to ensure towed in-water devices maintain the following distances as mission and circumstances allow: 250 yd from marine mammals Vicinity of sea turtles

Table 7. Requirements for Activity-based Mitigation Involving Navy Lookouts to Reduce the Risk of Entanglement in Nets used for Obstacle Avoidance Exercises

Mitigation Category	*Mitigation Requirements*	*Mitigation Zone and Requirements*
Nets deployed for testing of an Unmanned Underwater Vehicle	One Lookout on the support vessel	• For 15 minutes prior to the deployment of nets and while nets are deployed, the Lookout will observe for: Marine Mammals Sea Turtles • If a marine mammal or sea turtle is sighted within 500 yards of the deployment location, the support vessel will: Delay deployment of nets until the mitigation zone has been clear for 15 minutes Recover nets if they are already deployed • Nets will be deployed during daylight hours only

Mitigation Specific to Amphibious Landing Events

The Navy will implement mitigation measures during amphibious warfare activities in Hawaii as part of their INRMP requirements (USMC 2017). Navy personnel will survey beaches and nearshore waters in the vicinity of planned amphibious warfare training activities one hour prior to the event and throughout the duration of the event. If Hawaiian monk seals or sea turtles are present prior to the event, it may be delayed, rerouted, or cancelled. If Hawaiian monk seals or other ESA-listed species appear during an event personnel are asked to move away from the area and regular protection zone protocols are followed. If a Hawaiian monk seal or sea turtle comes ashore, all persons must remain at least 100 feet away from them.

Geographic Mitigation Areas

The Action Proponents will implement mitigation measures within specified areas to avoid potential impacts to ESA-listed species and their habitats (Figure 1 and Figure 2). Mitigation areas are particular geographic locations within the action area where the Action Proponents will implement additional avoidance and minimization measures. The mitigation applies year-round unless specified otherwise. Should national security require the Action Proponents to exceed the limit of a mitigation area (e.g., sonar hours), personnel conducting the activity would be required to obtain approval through the chain of command prior to commencement of the applicable event (Navy 2024d). The Action Proponents would provide NMFS with advance notification of a national security requirement to exceed geographic mitigation area limits and include relevant information about the event (e.g., sonar hours, use of explosives) in their annual training and testing activity reports (Navy 2024d).

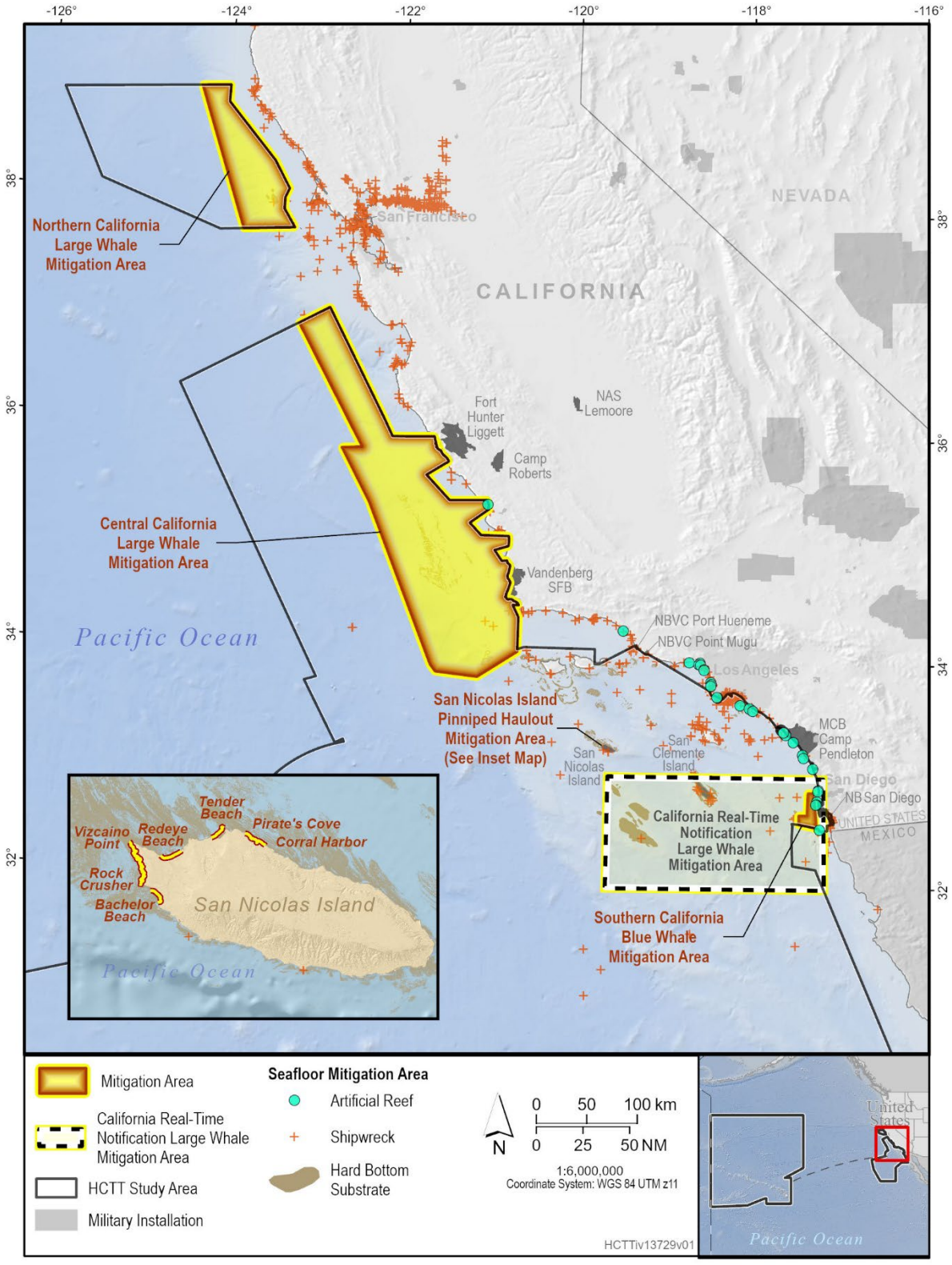


Figure 1. Mitigation Areas in the California Portion of the HCTT Action Area (Navy 2025d)

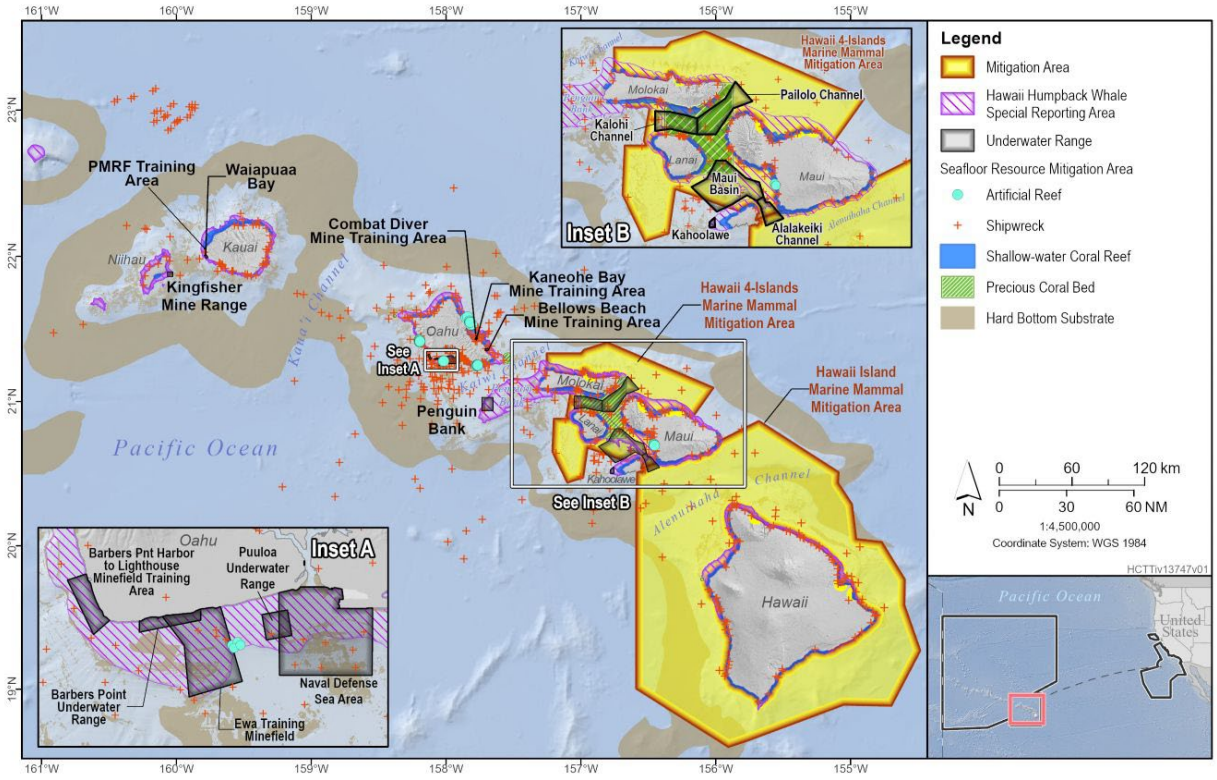


Figure 2. Mitigation Areas in the Hawaii Portion of the HCTT Action Area (Navy 2025d)

The Action Proponents considered several factors when determining the location of proposed geographic mitigation areas. First, they evaluated whether the mitigation area will be effective in reducing impacts to resources of biological or ecological importance. Next, they assessed how and to what degree implementation of mitigation measures will be compatible with planning, scheduling, and conducting proposed military readiness activities. A more thorough discussion of the factors used by the Action Proponents to determine which areas to propose for geographic mitigation is provided in Chapter 5 Mitigation of the HCTT FEIS/OEIS (Navy 2025d).

Shallow-Water Coral Reef and Precious Coral Bed Mitigation Areas

The Action Proponents propose to implement geographic mitigation designed to avoid potential impacts from explosives and physical disturbance and strike stressors on shallow-water coral reefs and precious coral beds. This mitigation will also help avoid potential impacts on organisms (e.g., invertebrates, fishes, sea turtles) that use these habitats for sheltering, resting, feeding, or other important life processes. A summary of the mitigation requirements within the Shallow-Water Coral Reef and Precious Coral Bed Mitigation Areas is shown in Table 8.

Table 8. Shallow-Water Coral Reef and Precious Coral Bed Mitigation Area Requirements (Navy 2024d)

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
Explosives	• The Action Proponents will not detonate any in-water explosives (including underwater explosives and explosives deployed against surface targets) within a horizontal distance of 350 yd from shallow-water coral reefs and precious coral beds (except in designated areas of the HRC, such as Barbers Point Underwater Range, Ewa Training Minefield, and Puuloa Underwater Range, where these features will be avoided to the maximum extent practical).	• Mitigation is designed to prevent direct impacts (and some level of indirect impacts) from explosives on shallow-water coral reefs and precious coral beds. The 350-yd mitigation area radius for in-water explosives is several times larger than the estimated impact footprint (e.g., crater and expelled material radius) of the largest bottom-laid explosive used in the HCTT action area.
Physical disturbance and strike	• The Action Proponents will not set vessel anchors within the anchor swing circle radius from shallow-water coral reefs and precious coral beds (except in designated anchorages). • The Action Proponents will not place non-explosive seafloor devices or deploy non-explosive ordnance against surface targets (including aerial-deployed mine shapes) within a horizontal distance of 350 yd from shallow-water coral reefs and precious coral beds (except in designated areas in the Hawaii and California OPAREAs, such as the nearshore areas of San Clemente Island and in the Silver Strand Training Complex, where these features will be avoided to the maximum extent practical).	• Mitigation is designed to avoid vessel anchors from coming into contact with shallow-water coral reefs and precious coral beds, when factoring in environmental conditions that could affect anchoring position, such as winds, currents, and water depth. • Mitigation is designed to prevent direct impacts (and some level of indirect impacts) from seafloor devices and non-explosive ordnance deployed against surface targets on shallow-water coral reefs and precious coral beds.

Artificial Reef, Hard Bottom Substrate, and Shipwreck Mitigation Areas

The Action Proponents propose to implement geographic mitigation for explosives and physical disturbance and strike stressors near artificial reefs, hard bottom substrate, and shipwrecks. For mitigation, the term "hard bottom substrate" is defined as substrate in the marine environment which could support a covering of biotic features (e.g., seaweed, sponges, and hard corals). To aid in the implementation of seafloor resource mitigation, the Action Proponents will include maps of the best available georeferenced data (i.e., where the available data accurately indicate the natural boundary of a seafloor resource and are not generalized within large geometric areas, such as large grid cells) in the PMAP for shallow-water coral reefs, artificial reefs, hard bottom substrate, and shipwrecks. A summary of the mitigation requirements within the Artificial Reef, Hard Bottom Substrate, and Shipwreck Mitigation Areas is shown in Table 9.

Table 9. Artificial Reef, Hard Bottom Substrate, and Shipwreck Mitigation Area Requirements (Navy 2024d)

Stressor	*Mitigation Requirements*	*Mitigation Details*
Explosives	• The Action Proponents will not detonate explosives on or near the seafloor (e.g., explosive bottom-laid or moored mines) within a horizontal distance of 350 yd. from artificial reefs, hard bottom substrate, and shipwrecks (except in designated areas in the Hawaii California OPAREAs, such as the nearshore areas of San Clemente Island and in the Silver Strand Training Complex, where these features will be avoided to the maximum extent practical).	• The 350-yd mitigation area radius is designed to prevent direct impacts (and some level of indirect impacts) from explosives on artificial reefs, hard bottom substrate, and shipwrecks.
Physical disturbance and strike	• The Action Proponents will not set vessel anchors within the anchor swing circle radius from artificial reefs, hard bottom substrate, and shipwrecks (except in designated anchorages). • The Action Proponents will not place non-explosive seafloor devices (that are not precisely placed) within a horizontal distance of 350 yd. from artificial reefs, hard bottom substrate, and shipwrecks (except as described in the bullet above for vessel anchors, the bullet below for precisely placed seafloor devices, and in designated areas of the Hawaii and California OPAREAs, such as the nearshore areas of San Clemente Island and in the Silver Strand Training Complex, where these features will be avoided to the maximum extent practical). • The Action Proponents will not position precisely placed non-explosive seafloor devices directly on artificial reefs, hard bottom substrate, or shipwrecks. • The Action Proponents will avoid positioning precisely placed non-explosive seafloor devices near these resources by the largest distance that is practical to implement based on mission requirements.	• Mitigation is designed to avoid vessel anchors from coming into contact with artificial reefs, hard bottom substrate, and shipwrecks, when factoring in environmental conditions that could affect anchoring position, such as winds, currents, and water depth. • Mitigation for seafloor devices that are either precisely placed or not precisely placed is designed to collectively prevent direct impacts (and some level of indirect impacts) from seafloor devices on artificial reefs, live hard bottom, and shipwrecks.

Hawaii Island Marine Mammal Mitigation Area

A summary of the mitigation requirements within the Hawaii Island Marine Mammal Mitigation Area is shown in Table 10.

Table 10. Hawaii Island Marine Mammal Mitigation Area Requirements (Navy 2024d)

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic	• The Action Proponents will not use more than 300 combined hours of MF1 and MF1C surface ship hull-mounted mid-frequency active sonar or 20 hours of helicopter dipping sonar (a mid-frequency active sonar source) annually within the mitigation area.	• Mitigation is designed to reduce exposure of numerous small and resident marine mammal populations (including Blainville's beaked whales, bottlenose dolphins, Cuvier's beaked whales, dwarf sperm whales, false killer whales, melon-headed whales, pantropical spotted dolphins, pygmy killer whales, rough-toothed dolphins, short-finned pilot whales, and spinner dolphins), humpback whales within important seasonal reproductive habitat, and Hawaiian monk seals within critical habitat, to levels of sound that have the potential to cause injurious or behavioral impacts.
Explosives	• The Action Proponents will not detonate in-water explosives (including underwater explosives and explosives deployed against surface targets) within the mitigation area.	• Mitigation is designed to prevent exposure of the species discussed above to explosives that have the potential to cause injury, mortality, or behavioral disturbance.

Hawaii 4-Islands Marine Mammal Mitigation Area

For HCTT Phase IV, the Action Proponents expanded the Hawaii 4-Islands Mitigation Area to include additional areas identified as high use by MHI IFKWs (see Section 4.2.7 for details). The 1,969.4 km^2 expansion of this mitigation area includes additional areas southwest of Lanai (897.6 km^2) and north of Molokai (1,071.8 km^2) to the westernmost extent of the island (Figure 2). A summary of the mitigation requirements within the Hawaii 4-Islands Marine Mammal Mitigation Area is shown in Table 11.

Table 11. Hawaii 4-Islands Marine Mammal Mitigation Area Requirements (Navy 2024d)

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic	• From November 15 to April 15, the Action Proponents will not use MF1 or MF1C surface ship hull-mounted mid-frequency active sonar surface within the mitigation area.	• Mitigation is designed to minimize exposure of humpback whales in high-density seasonal reproductive habitats (e.g., north of Maui and Molokai) and Main Hawaiian Islands insular false killer whales in high seasonal occurrence areas to levels of sound that have the potential to cause injurious or behavioral impacts.
Explosives	• The Action Proponents will not detonate in-water explosives (including underwater explosives and explosives deployed against surface targets) within the mitigation area (year-round).	• Mitigation is designed to prevent exposure of humpback whales in high-density seasonal reproductive habitats (e.g., north of Maui and Molokai), Main Hawaiian Islands insular false killer whales in high seasonal occurrence areas, and numerous small and resident marine mammal populations that occur year-round (including bottlenose dolphins, pantropical spotted dolphins, and spinner dolphins, and Hawaiian monk seals) to explosives that have the potential to cause injury, mortality, or behavioral disturbance.

Hawaii Humpback Whale Special Reporting Mitigation Areas

The Action Proponents will continue to implement special reporting requirements designed to monitor and minimize impacts to the non ESA-listed Hawaii DPS of humpback whales.

Northern California Large Whale Mitigation Area

A summary of the mitigation requirements within the Northern California Large Whale Mitigation Area is shown in Table 12. This mitigation area is new for HCTT Phase IV due to the northward expansion of the action area.

Table 12. Northern California Large Whale Mitigation Area Requirements (Navy 2025e)

Category	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic	• From June 1 – October 31, the Action Proponents will not use more than 300 combined hours of MF1 and MF1C surface ship hull-mounted mid-frequency active sonar (excluding normal maintenance and systems checks) total during training and testing within the combination of this mitigation area, the Central California Large Whale Mitigation Area, and the Southern California Blue Whale Mitigation Area.	• Mitigation to limit use of MF1 and MF1C active sonar is designed to reduce exposure of blue whales, fin whales, gray whales, and humpback whales in important seasonal foraging, migratory, and calving habitats to levels of sound that have the potential to cause injurious or behavioral impacts.

Central California Large Whale Mitigation Area

A summary of the mitigation requirements within the Central California Large Whale Mitigation Area is shown in Table 13. This mitigation area is also new for HCTT Phase IV.

Table 13. Central California Large Whale Mitigation Area Requirements (Navy 2024d)

Category	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic	• From June 1 to October 31, the Action Proponents will not use more than 300 combined hours of MF1 and MF1C surface ship hull-mounted mid-frequency active sonar (excluding normal maintenance and systems checks) total during training and testing within the combination of this mitigation area, the Northern California Mitigation Area, and the Southern California Blue Whale Mitigation Area.	• Mitigation to limit use of MF1 and MF1C active sonar is designed to reduce exposure of blue whales, fin whales, gray whales, and humpback whales in important seasonal foraging, migratory, and calving habitats to levels of sound that have the potential to cause injurious or behavioral impacts.

Southern California Blue Whale Mitigation Area

A summary of the mitigation requirements within the Southern California Blue Whale Mitigation Area is shown in Table 14. This mitigation is a continuation from HSTT Phase III, with a modified geographic extent based on best available science.

Table 14. Southern California Blue Whale Mitigation Area Requirements (Navy 2024d)

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic	• From June 1 to October 31, the Action Proponents will not use more than 300 combined hours of MF1 and MF1C surface ship hull-mounted mid-frequency active sonar (excluding normal maintenance and systems checks) total during training and testing within the combination of this mitigation area, the Northern California Mitigation Area, and the Central California Large Whale Mitigation Area.	• Mitigation to limit use of MF1 and MF1C active sonar is designed to reduce exposure of blue whales within important seasonal foraging habitats to levels of sound that have the potential to cause injurious or behavioral impacts.
Explosives	• From June 1 to October 31, the Action Proponents will not detonate in-water explosives (including underwater explosives and explosives deployed against surface targets) during large-caliber gunnery, torpedo, bombing, and missile (including 2.75-inch rockets) training and testing.	• Mitigation to limit the use of in-water explosives is designed to reduce exposure of blue whales within important seasonal foraging habitats to explosives that have the potential to cause injury, mortality, or behavioral disturbance.

California Large Whale Awareness Messages

A summary California Large Whale Awareness Message requirements is shown in Table 15.

Table 15. California Large Whale Awareness Requirements (Navy 2025e)

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
Acoustic, Explosives, Physical disturbance and strike	• The Action Proponents will broadcast awareness messages to alert applicable assets (and their Lookouts) transiting and training or testing off the U.S. West Coast to the possible presence of concentrations of large whales, including gray whales (November–June), fin whales (November–May), and mixed concentrations of blue, humpback, and fin whales that may occur based on predicted oceanographic conditions for a given year (e.g., May–November, April–November). - Awareness messages will describe that, while blue whales tend to be more transitory, some fin whales are year-round residents that can be expected in nearshore	• Mitigation to broadcast awareness messages to applicable assets, and to use that information to inform visual observations, is designed to minimize potential blue whale, gray whale, and fin whale vessel interactions and exposure to acoustic stressors, explosives, and physical disturbance and strike stressors that have the potential to cause mortality, injury, or behavioral disturbance during the foraging and migration seasons, and to resident whales.

Stressor	*Mitigation Requirements*	*Mitigation Benefits*
	waters within 10 nm of the California mainland and offshore operating areas at any time. - Awareness messages will describe that fin whales occur in groups of one to three individuals 90% of the time, and in groups of four or more individuals 10% of the time. - Awareness messages will describe that, unique to fin whales offshore Southern California (including the Santa Barbara Channel and Point Mugu Sea Range area), there could be multiple individuals and/or separate groups scattered within a relatively small area (1–2 nm) due to foraging or social interactions. - Awareness messages will emphasize that when a large whale is observed, this may be an indicator that additional marine mammals are present and nearby, and the vessel should take this into consideration when transiting. - Lookouts will use that knowledge to help inform their visual observations during military readiness activities that involve vessel movements, active sonar, in-water explosives (including underwater explosives and explosives deployed against surface targets), or the deployment of non-explosive ordnance against surface targets in the mitigation area.	

California Real-Time Notification Large Whale Mitigation Area

A summary of the mitigation requirements within the California Real-Time Notification Large Whale Mitigation Area is shown in Table 16.

Table 16. California Real-Time Notification Large Whale Mitigation Area Requirements (Navy 2025e)

Category	*Mitigation Requirements*	*Mitigation Benefits*
Physical disturbance and strike	• For each instance an aggregation of large whales (three or more whales within 1 nm (1.9 km) is sighted in the area between 32-33 degrees North and 117.2-119.5 degrees West, Action Proponent surface ships will report the sightings to other Action Proponent ships in the vicinity. Reported sightings will be made as soon as operationally and safely feasible. - The three whales that make up a defined "aggregation" would not all need to be from the same species, and the aggregation could consist either of a single group of three (or more) whales, or any combination of smaller groups totaling three (e.g., a group of two whales and a solitary whale) within the 1 nm zone.	• The real-time notification area encompasses the locations of recent (i.e., 2021 through 2025) military vessel strikes, and historic strikes where precise latitude and longitude were known. Timely information regarding an aggregation of whales in a particular location may result in an increased awareness of vessel strike risk by Lookouts and vessel operators.

Category	*Mitigation Requirements*	*Mitigation Benefits*
	- Lookouts will use the information from the real-time notifications to inform their visual observations of applicable mitigation zones. If ship watchstanders observe a large whale aggregation within 1 nm of the event vicinity within the area between 32–33 degrees North and 117.2–119.5 degrees West, the ship's crew will ensure the notification gets promulgated to other Navy vessels in the area. • The following information will be provided by the Navy in annual reporting to NMFS: Date, time and general location of the whales when the aggregation was first sighted, and the total number of whales in the aggregation. If the whales are identified by species, that information will be provided as well.	

San Nicolas Island Pinniped Haulout Mitigation Area

A summary of the mitigation requirements within the San Nicolas Island Pinniped Haulout Mitigation Area is shown in Table 17.

Table 17. San Nicolas Island Pinniped Haulout Mitigation Area Requirements (Navy 2024d)

Category	*Mitigation Requirements*	*Mitigation Benefits*
In-air vehicle launch noise	• Navy personnel shall not enter pinniped haulout or rookery areas. Personnel may be adjacent to pinniped haulouts and rookery prior to and following a launch for monitoring purposes. • Missiles shall not cross over pinniped haulout areas at altitudes less than 305 meters (m; 1,000 ft). • The Navy may not conduct more than 10 launch events at night annually. • Launch events shall be scheduled to avoid the peak pinniped pupping seasons from January through July, to the maximum extent practicable. • The Navy shall implement a monitoring plan using video and acoustic monitoring of up to three pinniped haulout areas and rookeries during launch events that include missiles or targets that have not been previously monitored for at least three launch events.	• Mitigation is designed to minimize in-air launch noise and physical disturbance to pinnipeds hauled out on beaches, as well as to continue assessing baseline pinniped distribution/abundance and potential changes in pinniped use of these beaches after launch events.

2.5 Stressors Resulting from the Components of the Proposed Action

In this section, the direct or indirect modifications to the land, water, or air caused by an action are identified as potential stressors. This section identifies all of the stressors that may affect[7] ESA-listed species or critical habitat, as well as the sources of those stressors. Some stressors may have multiple sources. Likewise, multiple sources may combine to create a stressor that would not exist if only one of the sources were present. The following is a summarization of stressors that are reasonably certain to be caused by this action.

The stressors we expect to result from the proposed action that may affect ESA-listed species or critical habitat fall into one of the following six categories:

- **Acoustic** (sonar and other transducers; vessel noise; aircraft noise; weapons noise; and air guns)
- **Explosives** (explosions in water, at the surface, and in air)
- **Energy** (in-water electromagnetic devices; high-energy lasers)
- **Physical disturbance and strikes** (vessels; in-water devices; military expended materials; and seafloor devices)
- **Entanglement** (wires and cables; decelerators/parachutes)
- **Ingestion** (military expended materials - munitions; military expended materials other than munitions)

2.5.1 Acoustic Stressors

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; and use of weapons or other deployed objects. Explosives also produce broadband sound but are characterized separately (Section 2.5.2) from other acoustic sources due to their unique energetic characteristics.

Vessel Noise

Radiated noise from vessels varies depending on the nature, size, and speed of the ship. Military vessels (including large ships and small craft) typically produce low-frequency, broadband underwater sound, though the exact level of noise produced varies by vessel type. Many military vessels incorporate quieting technology that other vessels (e.g., commercial ships) do not (Mintz and Filadelfo 2011; Mintz 2012). Anti-submarine warfare platforms (such as guided missile destroyers and cruisers) and submarines make up a large part of Navy traffic but contribute little noise to the overall sound budget of the oceans as these vessels are designed to be quiet to minimize detection. 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

[7] Pile driving is not discussed here because, due to the location of this activity in relation to ESA-listed species or critical habitat occurrence, we determined that this is not a stressor that "may affect" ESA-listed species or critical habitat in the action area (i.e., "no effect" determination).

(Mintz and Filadelfo 2011). The average acoustic signature for a Navy vessel is 163 dB re: 1 µPa, while the average acoustic signature for a commercial vessel is 175 dB re: 1 µPa (Mintz and Filadelfo 2011). A variety of smaller craft that vary in size and speed, such as service vessels for routine operations and opposition forces used during training and testing events, would be operating within the action area. During training and testing, speeds of most large naval vessels (greater than 60 ft) generally range from 10 to 17 knots for fuel consumption; however, ships will, on occasion, operate at higher speeds within their specific operational capabilities. During range modernization and sustainment activities, vessels would move slowly (0 to 3 knots) which would limit ship-radiated noise from propeller cavitation and water flow across the hull.

Exposure to HCTT vessel noise would be greatest in the areas of highest vessel traffic along the coast of California and near the main Hawaiian Islands. As of February 2025, there are 55 Navy surface vessels homeported in San Diego with another seven that could be part of training in SOCAL RC but are homeported in the Pacific Northwest (Navy 2025b). Nine Navy ships are homeported in Hawaii although some of these would also travel to SOCAL RC for various integrated training activities. Due to the high number of homeported ships in San Diego, the SOCAL RC accounts for about two-thirds (67%) of Navy large (i.e., 387 ft or greater) surface vessel days in the HCTT action area (Navy 2025k). The remaining one-third of Navy large (i.e., 387 ft or greater) surface vessel days in the HCTT action area are divided as follows: HRC and Temporary OPAREA (TOA) 20%; Transit Corridor 6%; PMSR 4%; and NOCAL RC 3% (Navy 2025k).

Action Proponent vessel activity can vary from year to year depending on a number of factors including ship building schedules, delays or acceleration in shifting additional ships to San Diego or Pearl Harbor, out of area forward deployments, budget constraints, training schedule changes, maintenance schedule changes, major training exercises, or other operational considerations. However, we anticipate that HCTT vessel activity levels associated with the action will, on average, be similar to historical activity levels. The estimated average annual at-sea days from 2009 through 2025 for Navy manned vessels greater than 118 m (387 ft or Littoral Combat Ship size and above) is 2,178 days (Navy 2025b). In addition, new for HCTT Phase IV, USVs greater than 61 m (200 ft) will be added as a new category of surface vehicle for testing activities with an estimated annual average of 728 at-sea days per year. The estimated average annual at-sea days from 2009 through 2025 for Coast Guard manned vessels greater than 100 m (328 ft) is 262 days (Navy 2025b). To account for limitations in data availability, future new vessel or repositioning home port assignments, and the history of medium sized (i.e., < 100 m) Coast Guard vessels striking large whales, the Coast Guard anticipates an additional 100 at-sea days per year during HCTT Phase IV activities (Navy 2025e).

Military vessels represent a relatively small proportion of the overall vessel traffic within the HCTT action area. Based on positional vessel traffic data from 2016-2023, Action Proponent vessels accounted for an estimated 6.7% (Navy 5.9%; Coast Guard 0.8%) of all vessel traffic (measured as vessel at-sea days) in the HCTT action area by vessels greater than 65 ft in length

(Johnson 2025; Navy 2025k). Action Proponent vessels make up a much larger proportion of overall vessel traffic in the SOCAL RC (22.0%) as compared to the HRC (2.4%).

While commercial vessel traffic (and, therefore, broadband noise generated by it) is relatively steady throughout the year, Action Proponent traffic is more episodic in space and time. Vessels engaged in training and testing may consist of a single vessel involved in a unit-level activity for a few hours or multiple vessels involved in a MTE that could last a few weeks within a given area. Vessel noise would also be produced during range modernization and sustainment activities including SOAR refurbishment, the installation, testing, maintenance of Shallow Water Training Ranges, Maritime Test Bed Expansion, Hawaii Cable Project, Installation and Maintenance of Underwater Platforms, and Kelp Removal (Navy 2024d). Activities involving vessel movements occur intermittently and are variable in duration. While Action Proponent vessels do contribute to the overall increased ambient noise in inshore waters near military ports, their contribution to the overall noise in these environments is typically a small percentage compared to the large volume of commercial and recreational vessel traffic in these areas (Mintz and Filadelfo 2011).

Aircraft Overflight Noise

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. The majority of aircraft noise would be generated at air stations, which are located at a sufficient distance from any ESA-listed species or critical habitat such that acoustic impacts would have no effect. Military activities involving aircraft generally are dispersed over large expanses of open ocean, but can be highly concentrated in time and location. Table 18 provides source levels for 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 during takeoff.

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 (Navy 2024d). 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 1983). 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 18. Sound Characteristics of Representative Aircraft Proposed for HCTT (Navy 2024d)

Noise Source	*Sound Pressure Level*
In-Water Noise Level	
F/A-18 Subsonic at 1,000 ft (300 m) Altitude	152 dB re: 1 µPa at 2 m below water surface[1]
F/A-18 Subsonic at 10,000 ft (3,000 m) Altitude	128 dB re: 1 µPa at 2 m below water surface[1]
H-60 Helicopter Hovering at 82 ft (25 m) 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 ft (25 m) Altitude	113 dBA re 20 µPa at 25 m from source[2]
F-35A Takeoff Through 1,000 ft (300 m) Altitude	119 dBA re 20 μPa^2s[4**] (per second of duration)
EA-18G Takeoff Through 1,622 ft (500 m) Altitude	115 dBA re 20 μPa^2s [5**] (per second of duration)

Sources: [1]Eller and Cavanagh (2000), [2]Bousman and Kufeld (2005), [3]U.S. Naval Research Advisory Committee (2009), [4]U.S. Department of the Air Force (2016), [5]U.S. Department of the Navy (2012)

[*] estimate based on in-air level

**average sound exposure 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), ft = feet

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 ft (Navy 2024d). 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) (Navy 2024d). Sound exposure levels at the sea surface from most air combat maneuver overflights are expected to be less than 85 A-weighted dBs (based on an F/A-18 aircraft flying at an altitude of 5,000 ft and at a subsonic airspeed (400 knots) (Navy 2024d). Kuehne et al. (2020) measured underwater noise levels at the end of the runway produced by Growler aircraft (EA-18G) immediately after take-off at Naval Air Station (NAS) Whidbey Island, WA. The average of the underwater received levels detected across all 10 overflights in the strongest 1-second window was 134 ± 3dB re: 1 micropascal (µPa) at 30 meters (m) below the sea surface. The frequency of the sound from these overflights ranged from 20 Hz to 30 kHz, with a peak between 200 Hz and one kHz. While sound levels between the hydrophone and the surface may have been stronger than those measured at 30 m (Kuehne et al. 2020), this study only examined received levels from one focal point in the water column and did not model sound propagation to determine how noise from Growler overflights spreads throughout the water column.

Rotary-wing Aircraft

Noise generated from rotary-wing aircraft (e.g., helicopters) is transient in nature and extremely variable in intensity (Navy 2024d). The underwater noise produced by helicopters is estimated to be 125 dB re: 1 µPa at 1 m below water surface for an UH-60 hovering at 82 ft (25 m) altitude (Bousman and Kufeld 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 (Navy 2024d). 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 to 100 ft. 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 (Navy 2024d).

Sonic Booms

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 the intensity of sonic booms include weight, size, and shape of aircraft or vehicle; altitude; flight paths; and atmospheric conditions. 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 ft of altitude (Navy 2024d). For example, an aircraft flying supersonic, straight, and level at 50,000 ft can produce a sonic boom carpet about 50 miles wide. 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 (Navy 2024d).

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 2024d). 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 2024d). Atmospheric conditions such as wind speed and direction, and air temperature and pressure can also influence the sound propagation of a sonic boom.

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 2002), and frequencies greater than 20 Hz are difficult to observe at depths greater than 33 ft (10 m) (Sohn et al. 2000). F/A-18 Hornet supersonic flight

was modeled to obtain peak sound pressure levels and energy flux density at the water surface and at depth (Laney and Cavanagh 2000). These results are shown in Table 19.

Table 19. Sonic Boom Underwater Sound Levels Modeled for F/A-18 Hornet Supersonic Flight (Navy 2024d)

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

[1] *Equivalent to SEL for a plane wave.*

* *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 µPa²-s = decibel(s) referenced to 1 micropascal squared seconds, m = meter(s), km = kilometers

For the proposed action, supersonic aircraft flights are not intentionally generated below 30,000 ft (9.1 km) unless over water and more than 30 nm from inhabited coastal areas or islands (Navy 2024d). 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 (Navy 2024d).

Sonar and other Transducers

Sonars and other transducers are non-impulsive sources and include the following types of devices: submarine sonars, surface ship sonars, rotary-wing aircraft dipping sonars, torpedo sonars, active sonobuoys, acoustic countermeasures, mine hunting sonars, navigation sonars, underwater communications, tracking pingers, and UUVs and their associated sonars. Active sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. The proposed action includes the use of a variety of sonars and other transducers 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 kHz) Doppler sonars used for navigation, like those used on commercial and private vessels (Navy 2024d).

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. Anti-submarine warfare activities often use mid-frequency (1 to 10 kHz) active sonar, though low-frequency and high-frequency active sonar systems are also used for specialized purposes. Mine warfare military readiness activities use a variety of different sonar systems that are typically high-frequency (greater than 10 kHz) and very high-frequency (greater than 100 kHz).

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.

In the BA for this consultation, non-impulsive narrowband bins were grouped by their frequency category (low – LF [<1 kHz], medium – MF [1-10 kHz], high – HF [10-100 kHz], or very high – VHF [>100 kHz]) and their source sound pressure level category (low, medium, or high), resulting in 12 possible (3x4) source category combinations (e.g., LFL, LFM, etc.). Broadband sources spanning more than one frequency category were grouped by the frequency categories they span (e.g., LF to MF, LF to HF, etc.). The sonar and transducer source categories and activity levels (i.e., hours or counts) for the proposed action are shown in Table 20. An additional category was added to represent the hull-mounted surface ship sonar (i.e., MF1) with variations of this category (i.e., MF1C and MF1K) for particular source characteristics as shown below.

Table 20. Sonar and Transducer Source Categories and Activity Levels (Navy 2024d)

Source Class Category	*Description*	*Unit*	*Training*		*Testing*	
			Annual	*7-year total*	*Annual*	*7-year total*
Broadband Sources						
LF	<205 dB	H	-	-	430 – 570	3,430
LF to HF		C	804 – 818	5,668	686 – 859	4,413
		H	-	-	1,662 – 2,077	11,371
LF to MF		H	-	-	2,801 – 2,833	19,737
MF to HF		H	8,082 – 11,585	67,085	1,451 – 1,779	10,483
Low-Frequency Acoustic Sources						
LFL	160 dB to 185 dB	H	-	-	12	70
LFM	185 dB to 205 dB	C	-	-	1,160 – 1,384	8,792
		H	468 – 536	3,480	7,531 – 8,984	56,955
LFH	>205 dB	C	1,493 – 2,120	12,353	6,046 – 6,704	44,296
		H	14	98	4,050 – 6,050	34,350
Mid-Frequency Acoustic Sources Other Than Hull-Mounted						
MFL	160 dB to 185 dB	H	-	-	12,632 – 14,982	92,794
MFM	185 dB to 205 dB	C	4,890 – 6,552	39,328	15,080 – 16,928	110,737
		H	30	210	14,381 – 16,081	101,064
MFH	>205 dB	C	41,060 – 41,191	287,927	-	-
		H	1,942 – 3,003	16,972	8,115 – 10,424	63,221

Source Class Category	Description	Unit	Training		Testing	
			Annual	7-year total	Annual	7-year total
Hull-Mounted Surface Ship Sonar						
MF1C	Hull-mounted surface ship sonar (previously MF11) with duty cycle >80%	H	796 – 1,406	7,400	45	314
MF1K	Hull-mounted surface ship sonar (previously MF1K) in Kingfisher mode	H	455	3,183	14	91
MF1	Hull-mounted surface ship sonar (previously MF1)	H	5,084 – 8,758	46,780	413 – 917	4,275
High-Frequency Acoustic Sources						
HFL	160 dB to 185 dB	H	60	420	21,326 – 22,076	151,532
HFM	185 dB to 205 dB	C	9	63	1,800 – 2,346	14,238
		H	3,907 – 5,290	31,498	12,409 – 13,259	89,322
HFH	>205 dB	C	801 – 899	5,902	835 – 1,137	6,351
		H	2,419 – 2,498	17,170	1,367 – 1,920	10,735
Very High-Frequency Acoustic Sources						
VHFL	160 dB to 185 dB	H	30	210	9,160	64,120
VHFM	185 dB to 205 dB	H	-	-	96	672
VHFH	>205 dB	C	-	-	72 – 106	580
		H	5,458 – 7,862	45,418	12,544 – 16,824	100,648

H = hours, C = count, L = Low, M = Medium, H = High, F = Frequency, K = Kingfisher Mode

In addition to the sources described above, the proposed action involves sonar and transducer sources described by the Action Proponents as *de minimis*. De minimis sources include those with low source level, narrow beamwidth, downward-directed transmission, short pulse lengths, frequencies above known hearing ranges of marine mammals, or some combination of these factors, as well as sources used for safety of navigation (Navy 2024d). Some examples include Doppler sonar/speed logs, fathometers, hand-held sonar, imaging sonar, high-frequency acoustic modems, tracking pingers, acoustic releases, and side-scan sonar.

Noise from Weapons

The Action Proponents train and test with a variety of weapons. Depending on the weapon, noise may be produced at launch or firing, while in flight, or upon impact. Other devices intentionally produce noise to serve as a non-lethal deterrent. 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, are discussed in Section 2.5.2.

Noise associated with large caliber weapons firing and the impact of non-explosive practice munitions would typically occur at locations greater than 12 nm from shore for safety reasons,

with the exception of areas near San Clemente Island in the Southern California Range Complex and near Kaula Island and the Pacific Missile Range Facility (PMRF) in the Hawaii portion of the action area (Navy 2024d). PMRF is a Navy training and testing facility located on the western side of Kauai that is used to launch missiles and aerial targets for military training and testing. PMRF is also used as a location for Army and Marine Corps artillery firing. Small and medium caliber weapons firing could occur throughout the action area. Examples of some types of weapons noise are shown in Table 21.

Table 21. Examples of Noise from Weapons (Navy 2024d)

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
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]

Sources: [1]Yagla and Stiegler (2003); [2]U.S. Department of the Army (1999); [3]U.S. Department of the Navy (2013).

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)

Firing a gun produces a muzzle blast in air that propagates away from the gun with the strongest directivity in the direction of fire. 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. 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 (Navy 2024d).

Supersonic projectiles, such as a fired gun shell, 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 (Navy 2024d). The bow shock wave itself travels at the speed of sound in air. Sound waves from a projectile in flight enters 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 (Navy 2024d).

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 (Navy 2024d).

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 (Navy 2024d).

Air Guns

Air guns are used with the following HCTT testing activities: Acoustic and Oceanographic Research; Intelligence, Surveillance, Reconnaissance; and Semi-Stationary Equipment Testing. Air guns are stainless steel tubes charged with high-pressure air via a compressor. An impulsive sound is generated when the air is almost instantaneously released into the surrounding water. Small air guns with capacities up to 60 cubic inches (in^3) would be used during testing activities in various offshore areas in the action area (Navy 2024d). Table 22 shows the number of air guns shots proposed in the action area. Air guns could be used up to 57 days per year in the HRC, up to 57 days per years in the NOCAL RC, and up to 44 days per years in the SOCAL RC (Navy 2025f).

Table 22. Number of Shots from Air Guns Proposed for use in the Action Area (Navy 2024d)

Source Class Category	*Description*	*Unit*	*Training*		*Testing*	
			Annual	*7-Year Total*	*Annual*	*7-Year Total*
AG	Air gun	C	0	0	30,432 – 36,780	228,720

Notes: C = Count

Generated impulses would have short durations, typically a few hundred milliseconds, with dominant frequencies below 1 kHz. The root mean square (RMS) sound pressure level[8] (SPL) and peak pressure (SPL peak) at a distance 1 m from the air gun would be approximately 215 dB rms re 1 µPa and 227 dB_{peak} re 1 µPa, respectively, if operated at the full capacity of 60 in^3 (Navy

[8] Sound pressure level (SPL) is a measure of the root-mean square, or "effective" sound pressure, converted to dB.

2024d). The size of the air gun chamber can be adjusted smaller, which would result in a lower SPL and sound exposure level[9] (SEL) per shot.

In general, the types of air guns proposed for HCTT are relatively low power compared to those used for seismic surveys (Navy 2025k). The Navy air guns are hand-held systems designed to produce a broadband signal to study effects in the water column. Unlike air guns used for oil and gas exploration, Navy air guns are not designed to penetrate into the seabed. In addition, Navy air guns use single shots, as compared to multiple shot arrays that are typically used for oil and gas exploration (Navy 2025k).

2.5.2 Explosive Stressors

Explosive detonations used during HCTT military readiness activities include bombs, missiles, rockets, naval gun shells, torpedoes, mines, demolition charges, and explosive sonobuoys. Explosive detonations occur in the air or near the water's surface (e.g., bombs, missiles, and naval gun shells), in the water column (e.g., torpedoes, explosive sonobuoys, mines, demolition charges), and on the ocean bottom (e.g., mines and demolition charges). Most explosive detonations would occur in waters greater than 200 ft in depth and greater than 3 nm from shore, except for nearshore Mine Warfare ranges (Navy 2024d).

The weight of the explosive warhead and the type of explosive material greatly influence the effect of an explosive. In their BA, the Action Proponents categorize explosives that could detonate in water or at the water surface into bins based on their NEW. The NEW is the explosive power of a charge expressed as the equivalent weight of trinitrotoluene (Navy 2024d). Table 23 shows the number of explosives within each bin that could be used in any year (and over seven years) during HCTT military readiness activities.

In addition to the sources described in Table 23, the proposed action involves very small impulsive sources (less than 0.1 lbs NEW) in bin E0. Quantitative modeling in multiple locations has validated that these sources have a very small zone of influence. Since bin E0 explosives are not anticipated to result in take of ESA-listed species, these sources are categorized as de minimis and are qualitatively analyzed.

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 (Navy 2024d). Explosions in air produce an initial blast front that propagates away from the detonation. When pressure waves from an explosion in air meet the water surface, the pressure wave can be transmitted across the

[9] Sound exposure level (SEL) SEL is the total noise energy produced from a single noise event and is the integration of all the acoustic energy contained within the event. Therefore, SEL takes into account both the intensity and the duration of a noise event.

air-water boundary and reflected off the water surface. When pressure waves in air meet the water at a perpendicular angle (e.g., straight down from an in-air source to a flat-water surface), the sound waves are both transmitted directly across the water surface in the same direction of travel and reflected 180° back toward the original direction of travel (Navy 2024d).

Explosions in air include detonations of projectiles and missiles during surface-to-air gunnery and air-to-air missile exercises conducted during air warfare. Most explosions in air for the proposed action would occur far above the water surface at sufficient altitude that a large portion of the sound would refract upward due to cooling temperatures with increased altitude, and would not reach the water's surface (Navy 2024d). Typical air explosive munitions proposed for HCTT, along with their NEW and typical detonation altitude, are shown in Table 24.

Table 23. Number of Explosives within Each Bin that Would Occur Annually (and over seven years) for HCTT Military Readiness Activities (Navy 2024d)

Bin	*Net Explosive Weight*	*Example Explosive Source*	*Training*		*Testing*	
			Annual	*7-year total*	*Annual*	*7-year total*
E1	0.1 – 0.25	Medium-caliber projectile	1,750 – 4,303	19,911	7,305 – 7,430	51,510
E2	> 0.25 – 0.5		2,950 – 3,000	20,800	-	-
E3	> 0.5 – 2.5	Large-caliber projectile	5,588 – 5,870	39,962	4,744 – 6,568	36,704
E4	> 2.5 – 5	Mine neutralization charge	179 – 190	1,286	1,324 – 2,624	18,352
E5	> 5 – 10	5 in. projectile	5,059 – 5,984	37,996	2,024 – 2,676	16,732
E6	> 10 – 20	Hellfire missile	2,293 – 2,357	16,243	144 – 148	1,020
E7	> 20 – 60	Demo block/shaped charge	115 – 190	1,030	549 – 622	2,322
E8	> 60 – 100	Lightweight torpedo	3 – 5	27	213 – 234	1,552
E9	> 100 – 250	500 lb. bomb	386 – 408	2,771	111 – 115	789
E10	> 250 – 500	Harpoon missile	89	620	13	91
E11	> 500 – 675	650 lb. mine	7 – 11	61	1 – 2	8
E12	> 675 – 1,000	2,000 lb. bomb	17 – 19	125	-	-
E13	> 1,000 – 1,740	Underwater demolitions – large area clearance	6	42	-	-
E16	10,000	Ship shock detonation	-	-	0-3	0-3

Notes: > = greater than, in. = inch, lb. = pound

Table 24. Typical Air Explosive Munitions Proposed for HCTT (Navy 2024g)

Weapon Type[1]	*Net Explosive Weight (lb.)*	*Typical Altitude of Detonation (ft)*
Surface-to-Air Missile		
RIM-66 SM-2 Standard Missile	80	> 15,000
RIM-116 Rolling Airframe Missile	39	< 3,000
RIM-7 Sea Sparrow	36	> 15,000 (can be used on low targets)
FIM-92 Stinger	7	< 3,000
Air-to-Air Missile		
AIM-9 Sidewinder	38	> 15,000
AIM-7 Sparrow	36	> 15,000
AIM-120 AMRAAM	17	> 15,000
Air-to-Surface Missile		
AGM-88 HARM	45	< 100
Projectile - Large Caliber[2]		
5"54 HE-ET	7	< 100
5"54 Other	8	< 3,000

[1] Mission Design Series and popular name shown for missiles.
[2] 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; HARM = High-Speed Anti-Radiation; HE-ET = High Explosive- Electronic Time

2.5.3 Energy Stressors

Energy stressors include electromagnetic devices, high-energy lasers, and high-power microwave systems, each of which is described further in the sections below.

In-Water Electromagnetic Devices and Cables

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 (Navy 2024d). The sound and electromagnetic signature cause nearby mines to detonate. Generally, voltage used to power these systems is around 30 volts (Navy 2024d). The static magnetic field generated by the mine neutralization devices is of relatively minute strength (i.e., somewhere between a refrigerator magnet and a can opener) (Navy 2024d). Mine neutralization systems use would be concentrated in areas where practice mines are deployed, typically in water depths less than 200 ft and at established training minefields or temporary minefields close to strategic ports and harbors (Navy 2024d).

The number and general location of activities using in-water electromagnetic devices for the proposed action is provided in Table 25.

Table 25. Proposed Number of HCTT Events with In-Water Electromagnetic Devices (Navy 2024d)

Activity Area	*Annual # of Events*	*7-Year # of Events*
Training		
California Study Area	30	210
Total	**30**	**210**
Testing		
Hawaii Study Area	8–15	52
California Study Area	14–33	145
Total	**22–48**	**197**

Cables deployed on the seafloor during SOAR modernization, the installation of two SWTRs, and the Maritime Test Bed Expansion all generate an electromagnetic force. However, these forces are less than that of the natural background magnetic force of the earth at distances beyond 0.6 centimeters (cm; 0.25 inches) from the cable (Navy 2024d).

High-energy Lasers

High-energy laser weapons training and testing involves the use of up to 30 kilowatts of directed energy as a weapon against small surface vessels and airborne targets (Navy 2024d). High-energy lasers would be employed from surface ships and are designed to create small but critical failures in potential targets. The high-energy laser is expected to be used at short ranges and the laser shuts off when it loses contact with the target (Navy 2024d).

The number and general location of activities using high-energy lasers for the proposed action is provided in Table 26.

Table 26. Number of Proposed HCTT Events Including High-Energy Lasers (Navy 2024d)

Activity Area	*Annual # of Events*	*7-Year # of Events*
Training		
Hawaii Study Area	8	56
California Study Area	8	56
Total	**16**	**112**
Testing		
Hawaii Study Area	55-63	421
California Study Area	555-565	3,930
Total	**610-628**	**4,351**

High-Power Microwave Systems

Pulsed-wave high-power microwave systems convert electrical or chemical energy into radiated energy and deliver high-power, short bursts of radiofrequency energy to neutralize a target (Navy 2024d). High-power microwave systems operate within a wide range of frequencies, from one megahertz to 100 gigahertz, and transmit energy to a target to degrade or destroy electrical components in the target. High-power microwave systems would be used only during testing activities off California and can be based on land, ships, or aircraft and directed to engage air, land, or surface targets. The Action Proponents have proposed to conduct up to 100 events with high-power microwave systems per year (Navy 2024d).

2.5.4 Physical Disturbance and Strike Stressors

Physical disturbance and strike stressors include vessels, in-water devices, military expended materials, and seafloor devices, each of which is described further in the sections below.

Vessels

Vessels used by the Action Proponents during training and testing activities include ships (e.g., aircraft carriers, surface combatants), support craft, and submarines ranging in size from 15 ft to over 1,000 ft. Typical speed of Navy vessels in HSTT core high use areas from 2014-2018 were between 10 and 15 knots (Starcovic and Mintz 2021). Vessel speed profiles (i.e., mean speed by area from 2014-2018) for U.S. military vessels operating within the HSTT action area are shown in Figure 3 (Hawaiian Islands) and Figure 4 (SOCAL RC). Large military ships (greater than 18 m in length) generally operate at average speeds of between 10 and 15 knots, and submarines generally operate at speeds in the range of 8 to 13 knots. At times, Navy ships may go faster or slower due to operational, training, and testing requirements, as well as weather or traffic conditions. Small craft (for purposes of this discussion, less than 50 ft in length), which are all support craft, have much more variable speeds (0 to 50+ knots, dependent on the mission).

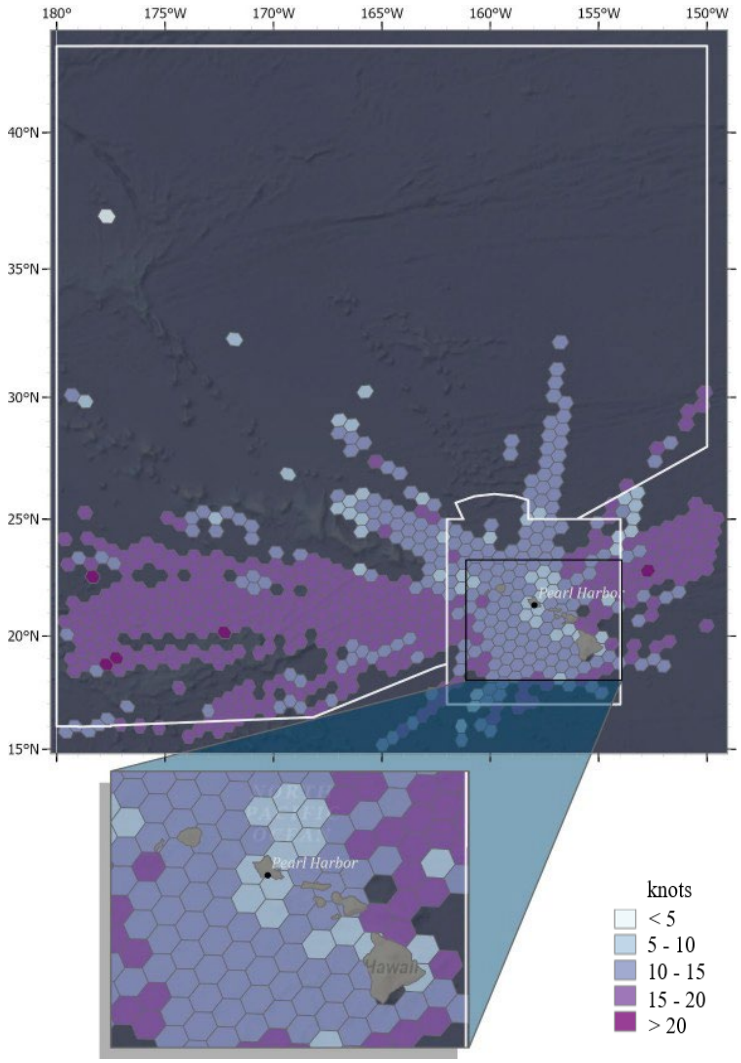


Figure 3. Mean Vessel Speed of U.S. Military Vessels Operating within the Hawaiian Islands Portion of the HCTT Action Area from 2014-2018 (Starcovic and Mintz 2021)

The ability of any vessel to detect a marine mammal or sea turtle 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. Table 27 provides examples of the types of vessels, lengths, and typical operating speeds for the HCTT proposed military readiness activities. The relative risk of a vessel strike is often assessed as a function of animal density and vessel traffic density (e.g., Fonnesbeck et al. 2008; Vanderlaan et al. 2008; Silber et al. 2021). Additional information on military vessel traffic density throughout the action area can be found above in Section 2.5.1.

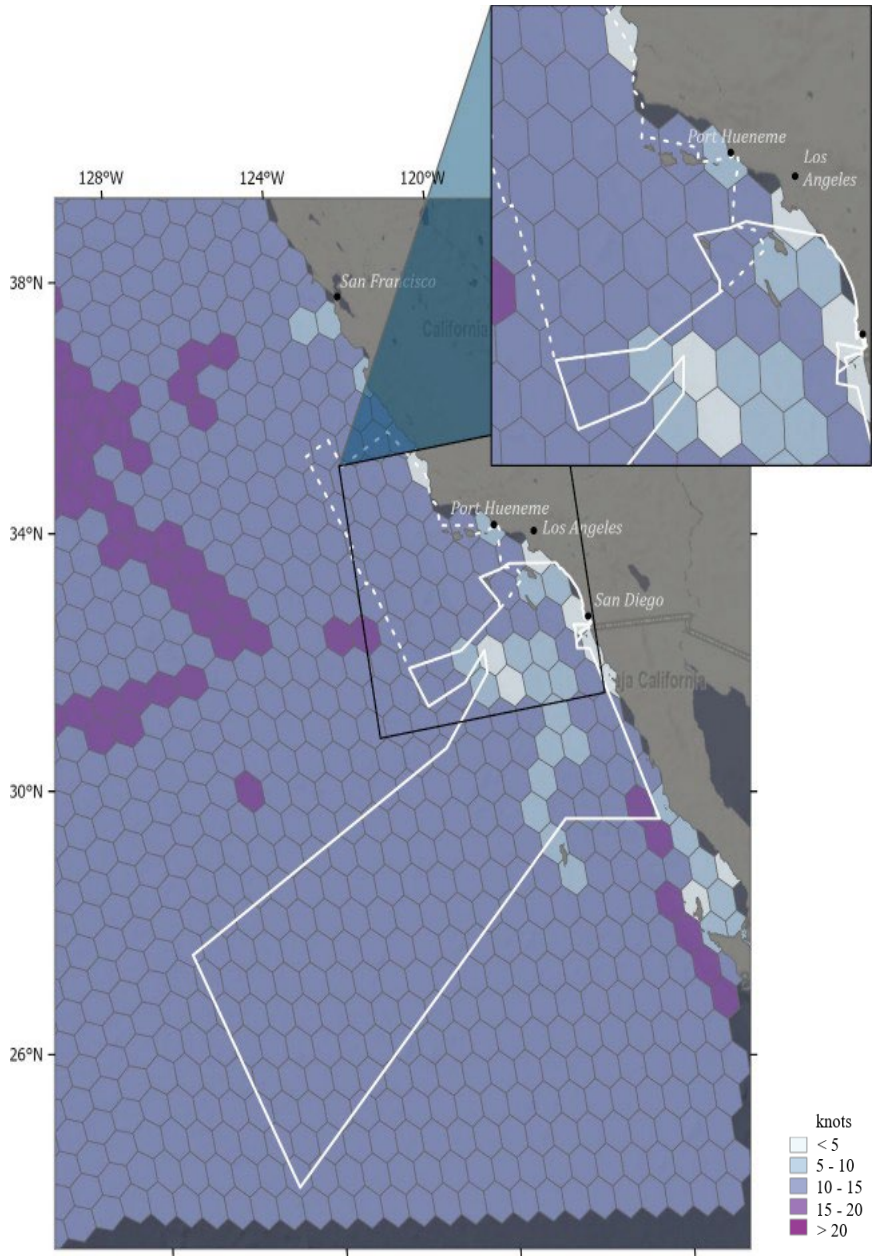


Figure 4. Mean Vessel Speed of U.S. Military Vessels Operating within the SOCAL RC Portion of the HCTT Action Area from 2014-2018 (Starcovic and Mintz 2021)

Table 27. Representative Vessel Types, Lengths, and Speeds (Navy 2024d)

Navy and Marine Vessel Types	*Example(s)*	*Length*	*Typical Operating Speed*
Aircraft Carrier	Aircraft Carrier (CVN)	>1000 ft	10–15 knots
Surface Combatant	Cruisers (CG), Destroyers (DDG), Frigates (FF) 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 vehicles	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
Coast Guard Vessel Types			
Large cutters	Legend Class, Heritage Class, Famous Class, Juniper Class, Reliance Class	181–418 ft	0 to 30 knots
Small cutters	Keeper Class, Sentinel Class, Bay Class, Island Class, Marine Protector Class, Small Harbor Tug	66–180 ft	0 to 30 knots
Boats	Aid to Navigation Boats, Screening Vessels, Lifeboats, Response Boats, Training Boats, Long Range Interceptors, Law Enforcement Boats, Cutterboat Over the Horizon, Transportable Security Boats	13–65 ft	0 to 40 knots

In-Water Devices

In-water devices include unmanned vehicles, such as remotely operated vehicles, USVs, UUVs, motorized autonomous targets, and towed devices. These devices are self-propelled and unmanned or towed through the water from a variety of platforms, including rotary-wing aircraft, UUVs, and surface ships. In-water devices are generally smaller than most Navy vessels, ranging from several inches to about 50 ft, although some USVs are up to 328 ft and some UUVs are up to 98 ft (Navy 2024d). These devices can operate anywhere from the water surface to the benthic zone. Representative types, sizes and speeds of in-water devices for the proposed action are shown in Table 28. For details regarding the number and location of different types of in-water devices used as part of the proposed action see Table 5-16 (Section 5.1.3.1.4.1, pp. 5-30 to 5-31) of the HCTT BA (Navy 2024d), which we incorporate here by reference.

Table 28. Representative Types, Sizes, and Speeds of In-Water Devices (Navy 2024d).

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	< 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	< 100 ft	Variable, up to 50+ knots
Large Unmanned Surface Vehicle	Research and Development Surface Vessels, Patrol Boats	> 100 ft	Typical 1-15 knots, sprint 25-50 knots
Unmanned Underwater Vehicle (UUV)	Acoustic Mine Targeting System, Airborne Mine Neutralization System, 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 UUV, Extra-Large UUV	< 100 ft	1–15 knots
Torpedoes	Light-weight and Heavy-weight Torpedoes	< 33 ft	20-30 knots

Military Expended Materials

Military expended materials that may cause physical disturbance or strike include: (1) all sizes of non-explosive practice munitions; (2) fragments from high explosive munitions; (3) expendable targets; and (4) expended materials other than munitions, such as sonobuoys or torpedo accessories (Navy 2024d). For details regarding the number and location of different types of military expended materials used as part of the proposed action see Tables 5-17 through 5-24 (Section 5.1.3.1.4.2, pp. 5-31 to 5-42) of the HCTT BA (Navy 2024d), which we incorporate here by reference.

Seafloor Devices

Seafloor devices represent items used during training or testing activities that are deployed onto the seafloor and typically recovered. These items include moored mine shapes, recoverable anchors, bottom-placed instruments, and robotic vehicles referred to as “crawlers” (Navy 2024d). Permanent bottom cable arrays and mine anchors associated with modernization and sustainment of ranges are not recovered. Seafloor device use occurs primarily in shallow waters and are either stationary or move very slowly along the bottom.

For details regarding the number and location of different types of seafloor devices used as part of the proposed action see Tables 5-25 (Section 5.1.3.1.4.3, p. 5-42) of the HCTT BA (Navy 2024d), which we incorporate here by reference.

2.5.5 Entanglement Stressors

The Action Proponents propose to utilize a variety of materials, including wires and cables, nets, decelerators and parachutes that could pose an entanglement risk to ESA-listed species.

Wires and Cables

Fiber optic cables are expended during training and testing associated with remotely operated mine neutralization activities. Although a portion may be recovered, some fiber optic cables used during training and testing associated with remotely operated mine neutralization activities would be expended (Navy 2024d). The Navy also proposes to permanently deploy fiber optic cables along the seafloor for activities related to modernization and sustainment of ranges.

The length of the expended tactical fiber would vary (up to about 3,000 m) depending on the activity. Tactical fiber has an 8 micrometer (0.008 millimeter [mm]) silica core and acylate coating, and looks and feels like thin monofilament fishing line. Tactical fiber is relatively brittle; it readily breaks if knotted, kinked, or abraded against a sharp object. The tactical fiber would be suspended within the water column during the activity, and then be expended and sink to the seafloor.

Guidance wires are used during heavy-weight torpedo firings to help the firing platform control and steer the torpedo (Navy 2024d). The torpedo guidance wire is a single-strand, thin gauge, coated copper alloy with a maximum tensile breaking strength of 40.4 pounds (lbs.) (Swope and McDonald 2013). The wire trails behind the torpedo as it moves through the water and sinks to the ocean floor after it is released. Torpedo guidance wire sinks at a rate of 0.24 m per second (Swope and McDonald 2013).

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. (Navy 2024d). The length of wire that extends out is no more than 1,500 ft 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. 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 (Navy 2024d).

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 (Navy 2024d).

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 29). Aircraft-launched sonobuoys and lightweight torpedoes (such as the MK 46 and MK 54) 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 cruciform shaped and associated with sonobuoys (Navy 2024d). 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.

Aerial targets (drones) use large and extra-large decelerators/parachutes. Large and extra-large decelerators/parachutes are also made of cloth and nylon, with suspension lines of varying lengths (Navy 2024d). 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 (Navy 2024d). The Action Proponents will attempt to retrieve large and extra-large decelerators/parachutes on the surface; however, they are not always successful due to logistics and safety concerns.

Table 29. Size Categories for Decelerators and Parachutes Expended During Proposed HCTT Activities (Navy 2024d).

Size Category	Diameter (ft)	Associated Activity
Small	1.5 to 6	Air-launched sonobuoys, lightweight torpedoes, and drones (drag parachute)
Medium	19	Illumination flares
Large	30 to 50	Drones (main parachute)
Extra-large	82	Drones (main parachute)

The number and general location of small and medium decelerators and parachutes used for the proposed action are provided in the HCTT BA (Section 5.1.3.1.6.3, Tables 5-32 and 5-33, p. 5-52), and are incorporated here by reference (Navy 2024d).

Nets

As part of extra-large UUV testing, scenarios would be developed to create subsurface obstacle avoidance interactions and would be recovered at the end of the test (Navy 2024d). Nets are anticipated to be a maximum size of 300 ft wide and 100 ft deep, with a 1-inch mesh. Nets would be tethered to one or two support vessel(s) and monitored at all times when in the water. Net

deployment and retrieval are estimated to take approximately 30 minutes. Nets would only be used during daylight hours and individual net deployment scenarios would occur over the course of a 48 hour window, but in discrete deployments for up to four hours at a time (i.e., no more than 16 hours over the 48 hour activity window). Nets would be deployed up to 80 days per year, over up to 40 individual deployment scenarios annually. Nets would be connected to and constantly monitored by the support vessels, which would hold static nets in place and move nets depending on the testing activity (Navy 2024d).

The number and general location of nets deployed as part of the proposed action are provided in the HCTT BA (Section 5.1.3.1.5.1, Tables 5-27 and 5-28, pp. 5-44 to 5-45), and are incorporated here by reference (Navy 2024d).

2.5.6 Ingestion Stressors

The Action Proponents expend the following types of materials that could become ingestion stressors during training and testing: 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 decelerators/parachutes (Navy 2024d).

Small- and medium-caliber projectiles range in size up to 2.25 inches diameter. Flechettes from some non-explosive rockets are approximately two inches long. Each non-explosive flechette rocket contains approximately 1,180 individual metal flechettes that are released. Many different types of high-explosive munitions and targets can result in fragments that are expended at sea during training and testing activities. Fragments would result from fractures in the munitions casing and would vary in size depending on the net explosive weight and munition type. Expendable targets that may result in small fragments that represent an ingestion risk include air-launched decoys, ship hulks, surface targets (e.g., marine markers, cardboard boxes, and 10 ft diameter red balloons), and mine shapes (Navy 2024d). Other military expended materials such as targets, large-caliber projectiles, intact training and testing bombs, guidance wires, 55-gallon drums, sonobuoy tubes, and marine markers would likely be too large for marine organisms to consume (Navy 2024d). At-sea targets that are used during activities involving high-explosives may result in fragments and ultimate loss of the target. The number and general location of targets used during proposed military readiness activities that may result in fragments are provided in the HCTT BA (Section 5.1.3.1.6.3, Tables 5-30 and 5-31, pp. 5-49 to 5-50), and are incorporated here by reference (Navy 2024d).

Solid metal materials, such as small-caliber projectiles or fragments from high-explosive munitions, are expected to sink rapidly to the seafloor. Lighter plastic items (e.g., plastic end caps from chaff cartridges or plastic pistons from flare cartridges) 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 (Navy 2024d).

Chaff consists of reflective, aluminum-coated glass fibers used to obscure ships and aircraft from radar-guided systems. Chaff is released or dispensed from cartridges that contain millions of

fibers. 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). 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). 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 (Navy 2024d).

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 (Navy 2024d). 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.

Decelerators/parachutes are expended with the use of sonobuoys, lightweight torpedoes, and illumination flares. Only the small and medium size decelerators/parachutes expended with sonobuoys and lightweight torpedoes pose an ingestion risk to marine life (Navy 2024d). See Section 2.5.3 for a more detailed discussion of this stressor.

2.6 MMPA Regulations and Issuance of a Letter of Authorization

On September 16, 2024, the Permits Division notified us that they had received an application from the Action Proponents requesting LOAs for the incidental taking of marine mammals during the conduct of training and testing military readiness activities in the North Pacific Ocean in areas off of Hawaii and the California coast. The application request was submitted for four LOAs: 1) a 7-year LOA for Navy training activities, 2) a 7-year LOA for Navy testing activities, 3) a 7-year LOA for Coast Guard training activities, and 4) a 7-year LOA for Army training activities. All four LOAs were requested for the period from December 21, 2025, through December 20, 2032.

The Permits Division proposes to issue regulations pursuant to the MMPA, as amended (16 U.S.C. 1361 et seq.) for the Action Proponents to "take" marine mammals incidental to HCTT military readiness activities. The regulations propose to authorize the issuance of LOAs that will allow the Action Proponents to "take" marine mammals incidental to their training and testing activities. Issuance of the LOAs by the Permits Division is 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 CFR §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, which also include proposed mitigation, monitoring, and reporting requirements, 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 Action Proponents to "take" marine mammals incidental to HCTT activities, as modified during ESA consultation. The final MMPA rule, upon publication, will also be available at the website shown above. Note that this opinion was completed prior to the publication of the final MMPA rule in the *Federal Register*. We anticipate that, upon publication, the MMPA rule will reflect the mitigation and monitoring measures proposed by the Action Proponents and/or agreed to during ESA consultation. A description of the mitigation measures for this consultation is in Section 2.4.2 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 (Section 10). We also anticipate that the levels of take of ESA-listed marine mammals authorized under the final MMPA rule and LOAs will be consistent with those analyzed in this opinion. Upon publication, we will review the MMPA rule to ensure these conditions are met. If administrative changes are needed following publication of the MMPA rule, we will update this 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 12 may apply.

3 ACTION AREA

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 CFR §402.02). The action area is defined by the extent of the environmental changes the stressors cause on the physical environment (e.g., land, air or water, detailed in the previous section). The HCTT action area includes areas of the North Pacific Ocean off of Hawaii and California, including a transit corridor connecting these two areas.

Navy range complexes consists of geographic areas that encompass a water component (above and below the surface), airspace, and may include land components, where training and testing of military platforms, tactics, munitions, explosives, and electronic warfare systems occur. Range complexes include established operating areas and special use airspace, which may be further divided to provide better control of the area for safety reasons. Range complexes provide controlled environments where military ship, submarine, and aircraft crews can train in realistic conditions while safely deconflicting with non-military activities, such as civilian shipping and air traffic.

The Hawaii portion of the action area is comprised of the Hawaii Range Complex (HRC) and the TOA. Nearly all the military readiness activities in the Hawaii portion of the action area take place within the Hawaii Range Complex (Figure 5), the area that immediately surrounds the island chain from Hawaii to Kauai. The Hawaii Range Complex (HRC) consists of 115,000 nm^2

of special use airspace and 235,000 nm^2 of sea and undersea space, including 1,100 nm^2 of instrumented underwater ranges at the PMRF (Navy 2024d). Within the Hawaii Range Complex are areas where specific military readiness activities occur, generally centered around the islands of Kauai, Oahu, and Maui. The TOA is used minimally for HCTT activities; primarily for vessel transits between Hawaii and destinations in the western Pacific, and some ballistic missile defense exercises where debris may fall within the broad ocean area of the TOA boundaries (Navy 2024d).

Descriptions of the specific training and testing areas within the HRC, and more detailed island maps, can be found in the HCTT Phase IV BA (Section 2.1.1.1, pp. 2-4 through 2-11, Tables 2-1 through 2-4, and Figures 2-3 through 2-6), which we incorporate here by reference.

The California portion of the action area (Figure 6) is comprised of the SOCAL RC, PMSR, NOCAL RC, and Silver Strand Training Complex (SSTC). The proposed SOCAL RC (Figure 7) for HCTT Phase IV encompasses 217,000 nm^2 of sea space and 210,000 nm^2 of special use airspace. This includes newly proposed air space (W-293 and W-294) expanding the range complex from Phase III, as shown in Figure 6. The SOCAL RC includes instrumented underwater training ranges, mine training ranges, laser training ranges, and access to the seaside of Naval Base Point Loma, as well as pierside locations at Naval Base Point Loma, Naval Base Coronado, and Naval Base San Diego within San Diego Bay (Navy 2024d). The SSTC is an integrated set of training areas located on and adjacent to the Silver Strand, a narrow, sandy isthmus separating the San Diego Bay from the Pacific Ocean. PMSR is located along the Pacific Coast of Central and Southern California and includes 27,000 nm^2 of air and sea space (Figure 8). PMSR supports training, testing, and evaluation of a wide variety of weapons, ships, aircraft, and specialized systems, including in-water minefields and associated training and testing (Navy 2024d). The NOCAL RC is an existing range complex being incorporated into HCTT for the first time. The NOCAL RC consists of two separate areas located offshore of central and northern California, one northwest of San Francisco and the other southwest of Monterey Bay (Figure 9). Each area includes special use airspace and the underlying sea space. The southern area includes approximately 10,000 nm^2 of airspace and the northern area includes approximately 6,000 nm^2 of airspace. These areas' proximity to Naval Air Station Lemoore is particularly important for the support of critical Strike Fighter Wing training. These areas also provide air and sea space for Carrier Strike Groups and Amphibious Ready Groups to conduct training, certifications, and testing.

Descriptions of the specific training and testing areas within the California portion of the action area, including more detailed maps, can be found in the HCTT Phase IV BA (Section 2.1.2, pp. 2-16 through 2-27, Tables 2-5 through 2-9, and Figures 2-10 through 2-12), which we incorporate here by reference (Navy 2024d).

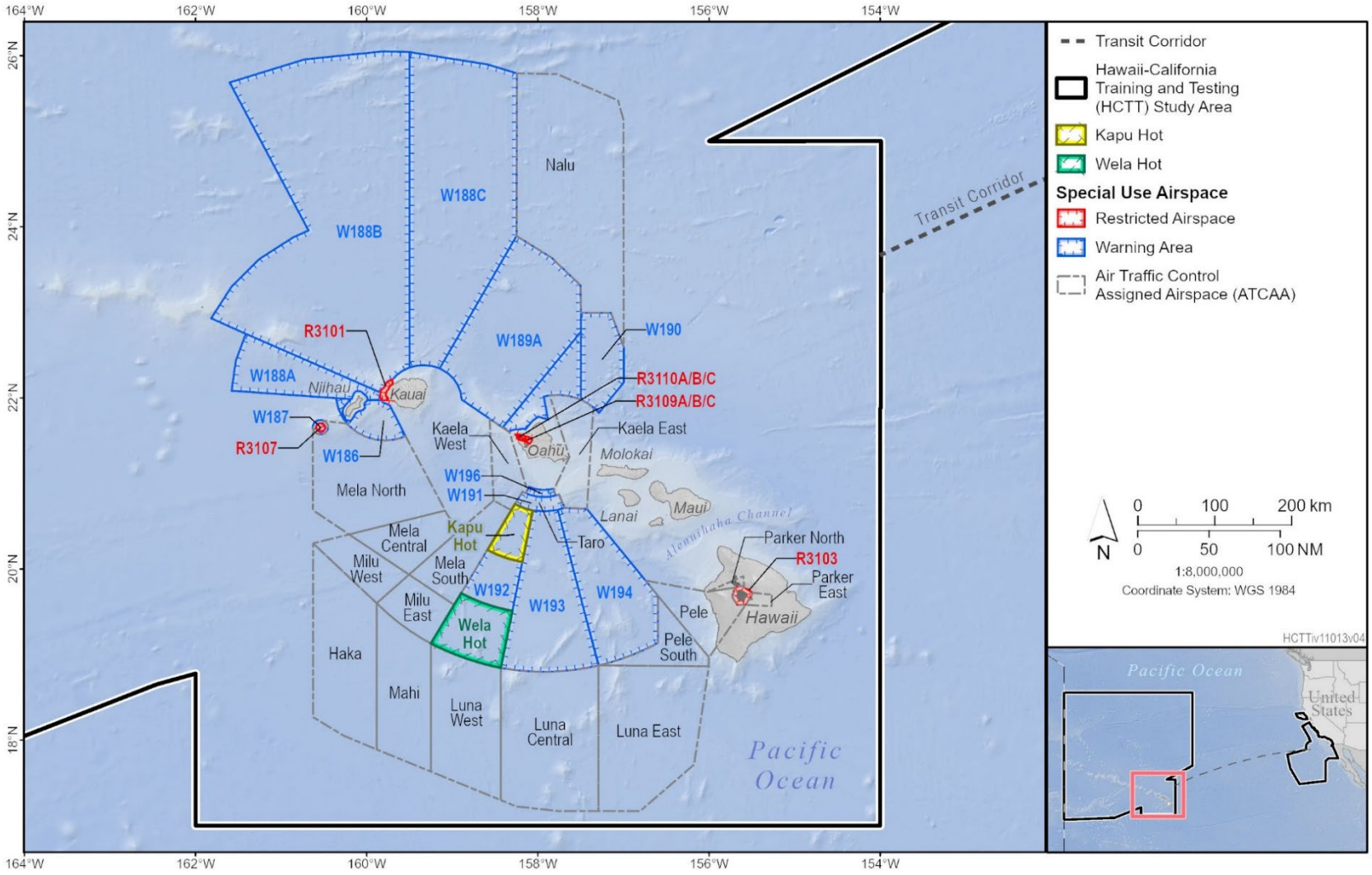


Figure 5. Hawaii Range Complex (Navy 2024d)

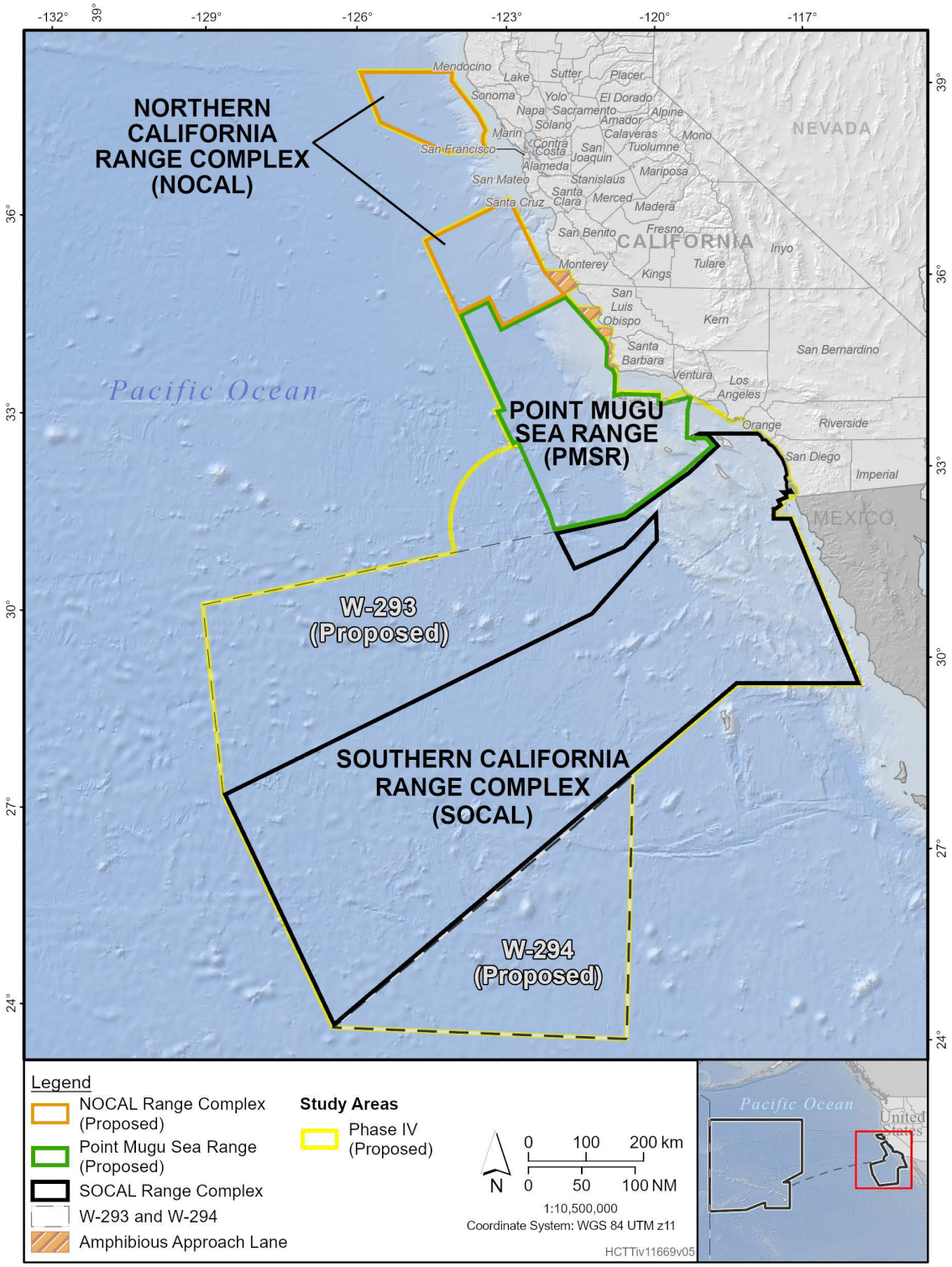


Figure 6. California Portion of the HCTT Action Area (Navy 2024d)

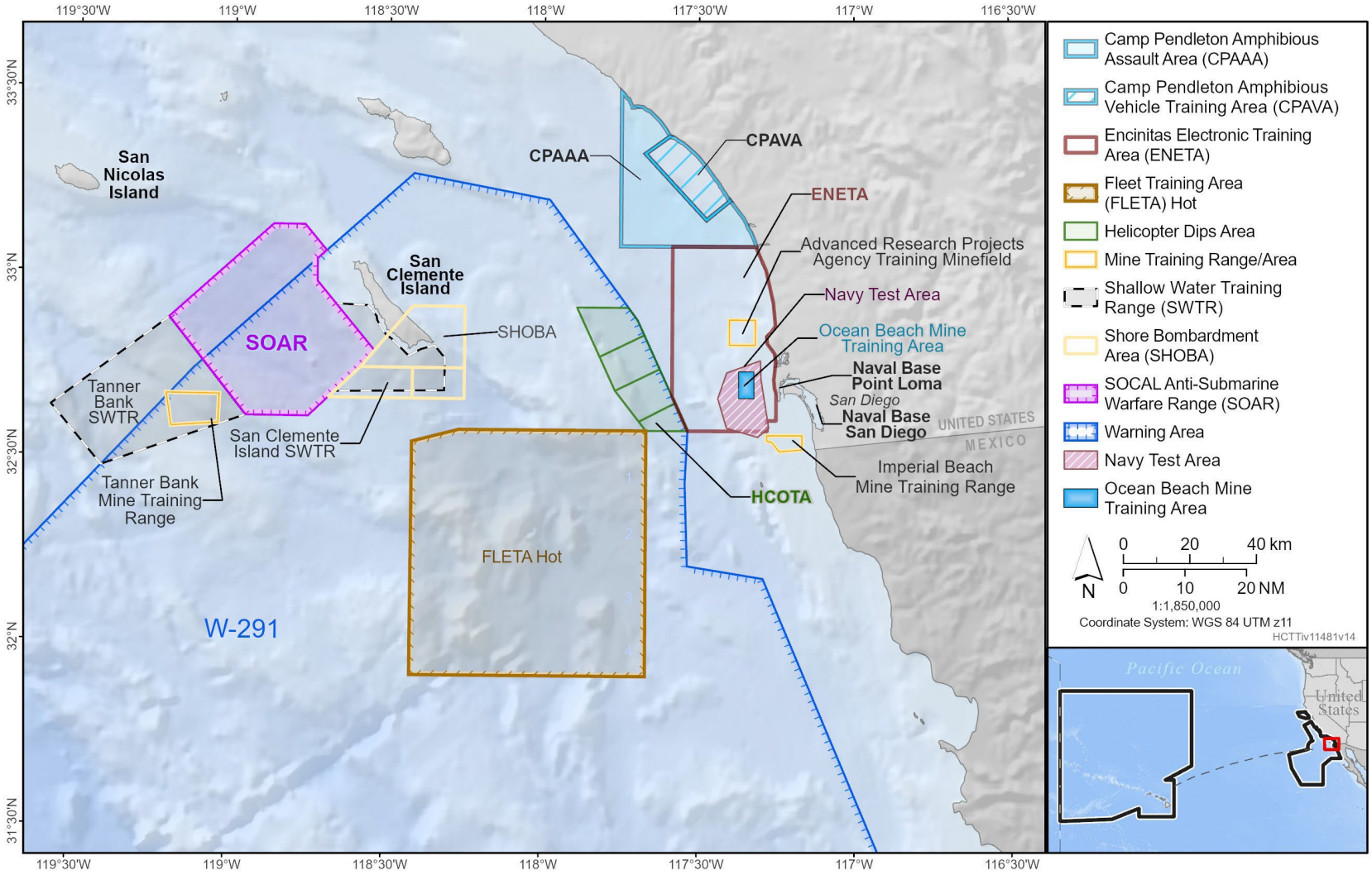


Figure 7. Southern California Range Complex (Navy 2024d)

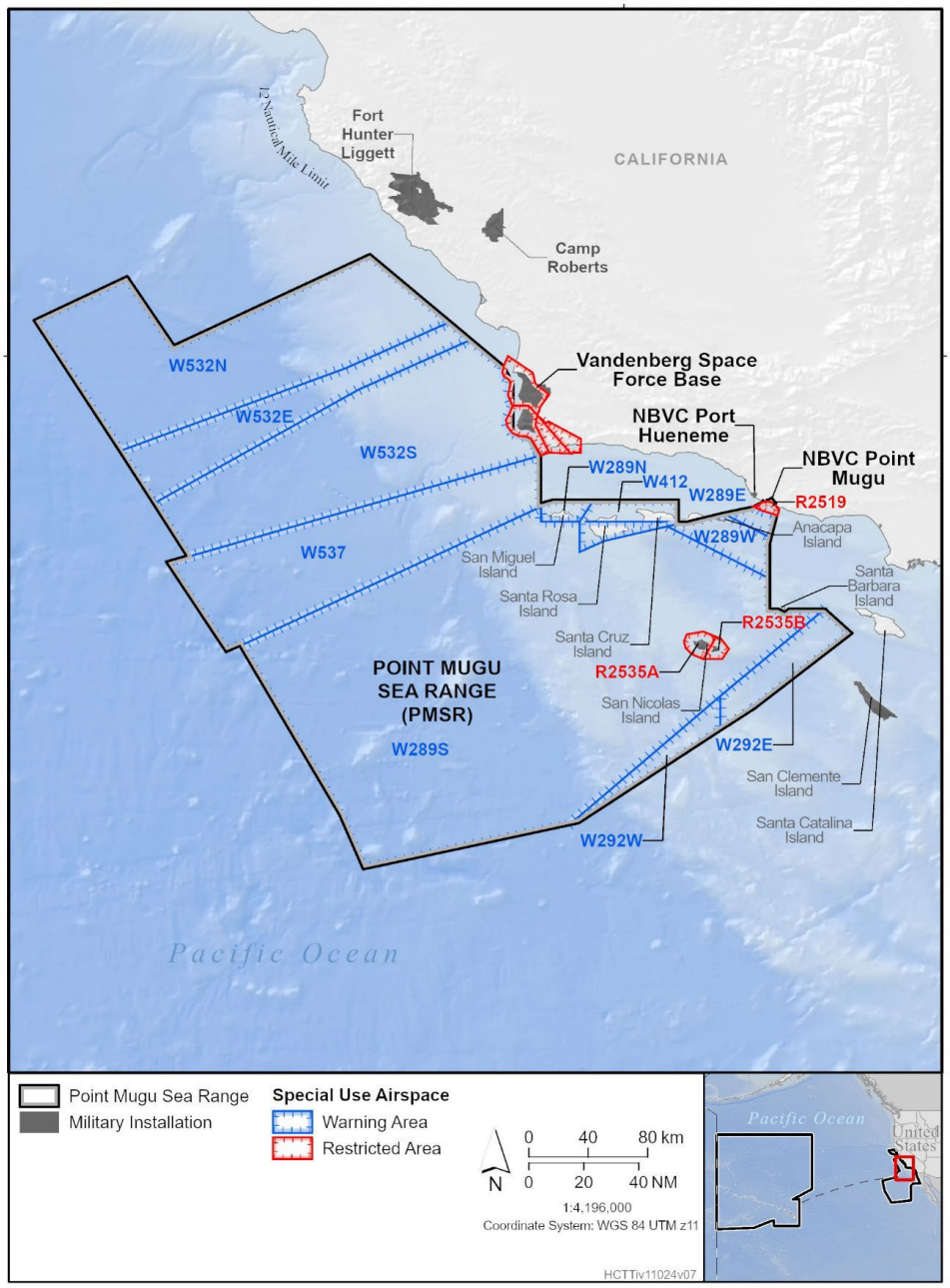


Figure 8. Point Mugu Sea Range (Navy 2024d)

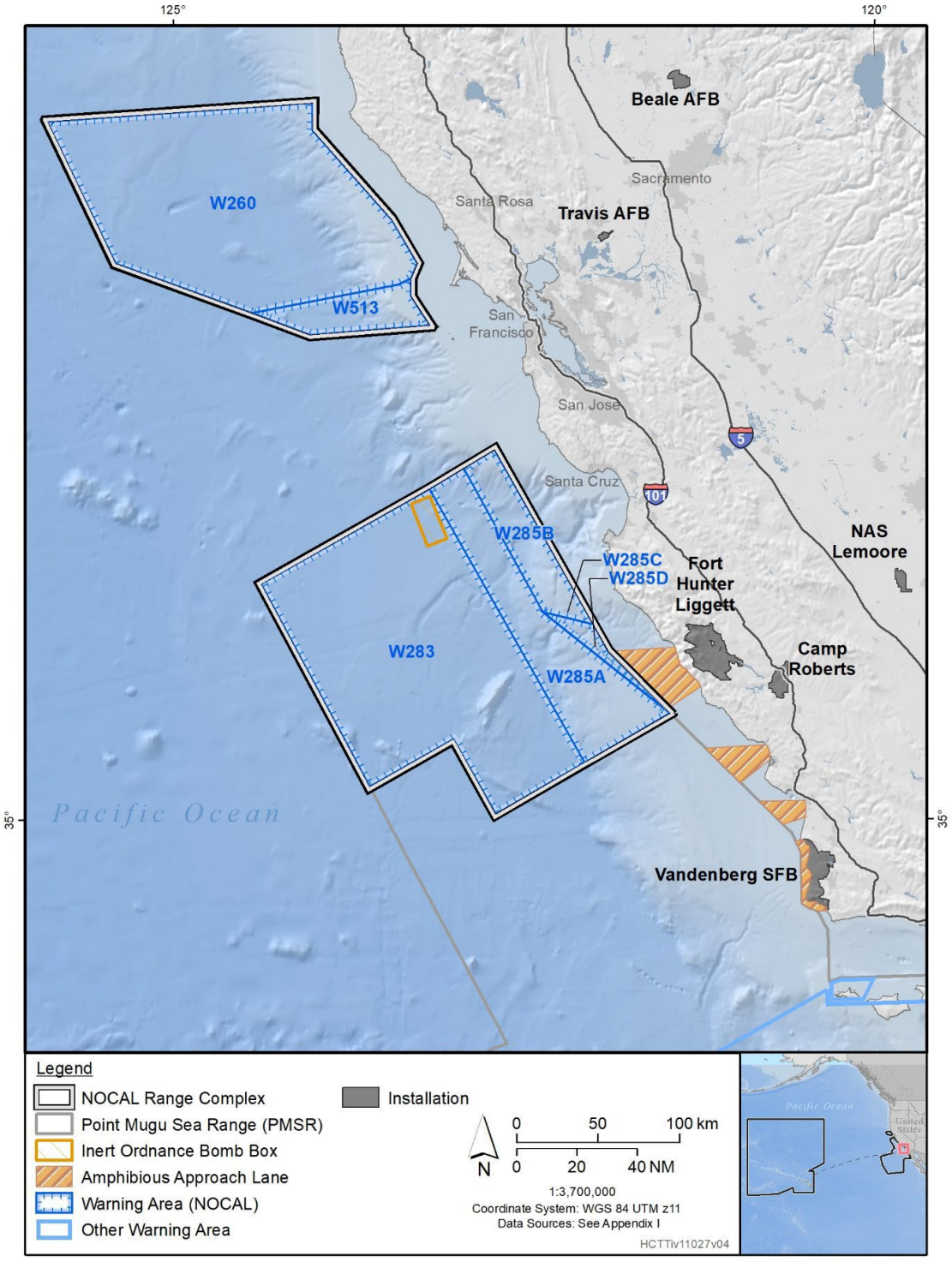


Figure 9. Northern California Range Complex (Navy 2024d)

4 SPECIES AND DESIGNATED CRITICAL HABITAT THAT MAY BE AFFECTED

The ESA allows for three general determinations for listed species and critical habitat: 1) no effect, 2) may affect, not likely to adversely affect (NLAA), and 3) may affect, likely to adversely affect (LAA). Action agencies, prior to requesting ESA consultation, determine whether their proposed action may affect ESA-listed or proposed species or their designated critical habitat. Generally, a "no effect" determination means there is no plausible route of effects between the proposed action and any ESA-listed or proposed species or designated critical habitat and does not require consultation. Any scenario where there is a plausible route of effects, no matter how unlikely, is considered "may affect." For any action that "may affect" an ESA-listed or proposed species or its designated critical habitat, the action agency shall consult with the Services under section 7(a)(2) of the ESA.

An action agency is also required to confer with the Services on any effects to proposed species or proposed critical habitat if those effects are likely to jeopardize the continued existence of the species or destroy or adversely modify the proposed critical habitat. However, action agencies may voluntarily confer with the Services for all proposed species or proposed critical habitat in the action area when the action may affect those proposed entities without rising to a level requiring us to confer.

This section of the opinion considers: 1) the stressors generated by the proposed action (Section 2.5), and 2) the action area (Section 3) to identify the ESA-listed or proposed species and designated critical habitat that may be affected by those stressors in the action area. The ESA-listed species and designated critical habitats that may be affected by this action, along with their regulatory status, are shown in Table 30.

Table 30. ESA-Listed or Proposed Species and Designated Critical Habitat That May be Affected by the Proposed Action

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals			
Blue Whale (*Balaenoptera musculus)*	E – 35Fed. Reg. 18319	-- --	11/2020
Fin Whale (*Balaenoptera physalus)*	E – 35Fed. Reg. 18319	-- --	75Fed. Reg. 47538 07/2010
Gray Whale (*Eschrichtius robustus*) Western North Pacific DPS	E – 35Fed. Reg. 18319	-- --	-- --
Humpback Whale (*Megaptera novaeangliae*) – Central America DPS	E – 81Fed. Reg. 62259	84Fed. Reg. 54354	11/1991
Humpback Whale (*Megaptera novaeangliae*) – Mexico DPS	T – 81Fed. Reg. 62259	84Fed. Reg. 54354	11/1991
Sei Whale (*Balaenoptera borealis*)	E – 35Fed. Reg. 18319	-- --	12/2011
Sperm Whale (*Physeter macrocephalus*)	E – 35Fed. Reg. 18319	-- --	75Fed. Reg. 81584
North Pacific Right Whale (*Eubalaena japonica*)	E – 73Fed. Reg. 12024	73Fed. Reg. 19000*	78Fed. Reg. 34347 06/2013

Species	ESA Status	Critical Habitat	Recovery Plan
False Killer Whale (*Pseudorca crassidens*) – Main Hawaiian Islands Insular DPS	E – 77Fed. Reg. 70915	83Fed. Reg. 35062	11/2021
Killer Whale (*Orcinus orca*) – Southern Resident killer whales DPS	E – 70Fed. Reg. 69903 Amendment 80Fed. Reg. 7380	71Fed. Reg. 69054 86Fed. Reg. 41668	73Fed. Reg. 4176 01/2008
Guadalupe Fur Seal (*Arctocephalus townsendi*)	T – 50Fed. Reg. 51252	-- --	-- --
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	E – 41Fed. Reg. 51611	80Fed. Reg. 50925	72Fed. Reg. 46966 2007
Sea Turtles			
Green Turtle (*Chelonia mydas*) – Central North Pacific DPS	T – 81Fed. Reg. 20057	88Fed. Reg. 46572 (Proposed)	63Fed. Reg. 28359
Green Turtle (*Chelonia mydas*) – East Pacific DPS	T – 81Fed. Reg. 20057	88Fed. Reg. 46572 (Proposed)	63Fed. Reg. 28359
Hawksbill Turtle (*Eretmochelys imbricata)*	E – 35Fed. Reg. 8491	-- --	63Fed. Reg. 28359 and 05/1998 – U.S. Pacific
Leatherback Turtle (*Dermochelys coriacea*)	E – 35Fed. Reg. 8491	44Fed. Reg. 17710 and 77Fed. Reg. 4170	05/1998 – U.S. Pacific
Loggerhead Turtle (*Caretta caretta*) – North Pacific Ocean DPS	E – 76Fed. Reg. 58868	-- --	63Fed. Reg. 28359
Olive Ridley Turtle (*Lepidochelys olivacea*) All Other Areas	T – 43Fed. Reg. 32800	-- --	-- --
Olive Ridley Turtle (*Lepidochelys olivacea*) Mexico's Pacific Coast Breeding Colonies	E – 43Fed. Reg. 32800	-- --	63Fed. Reg. 28359
Fishes			
Giant Manta Ray (*Mobula birostris*)	T – 83Fed. Reg. 2916	-- --	-- --
Oceanic Whitetip Shark (*Carcharhinus longimanus*)	T – 83Fed. Reg. 4153	-- --	-- --
Scalloped Hammerhead Shark (*Sphyrna lewini*) – Eastern Pacific DPS	E – 79Fed. Reg. 38213	-- --	-- --
Green Sturgeon (*Acipenser medirostris*) – Southern DPS	T – 71Fed. Reg. 17757	74Fed. Reg. 52300	8/2018- Final
Eulachon (*Thaleichthys pacificus*) – Southern DPS	T – 75Fed. Reg. 13012	76Fed. Reg. 65323*	9/2017
Steelhead Trout (*Oncorhynchus mykiss*) – Southern California DPS	E – 71Fed. Reg. 834	70Fed. Reg. 52487*	77Fed. Reg. 1669
Steelhead (*Oncorhynchus mykiss*) – California Central Valley DPS	T – 71Fed. Reg. 834	70Fed. Reg. 52487*	79Fed. Reg. 42504
Steelhead (*Oncorhynchus mykiss*) – Central California Coast DPS	T – 71Fed. Reg. 834	70Fed. Reg. 52487*	81Fed. Reg. 70666
Steelhead (*Oncorhynchus mykiss*) – Northern California DPS	T – 71Fed. Reg. 834	70Fed. Reg. 52487*	81Fed. Reg. 70666
Steelhead (*Oncorhynchus mykiss*) – South-Central California Coast DPS	T – 71Fed. Reg. 834	70Fed. Reg. 52487*	78Fed. Reg. 77430
Chinook salmon (*Oncorhynchus tshawytscha)* – California Coastal ESU	T – 70Fed. Reg. 37160	70Fed. Reg. 52488*	81Fed. Reg. 70666

Species	ESA Status	Critical Habitat	Recovery Plan
Chinook salmon (*Oncorhynchus tshawytscha)* – Central Valley Spring-Run ESU	T – 70Fed. Reg. 37160	70Fed. Reg. 52488*	79Fed. Reg. 42504
Chinook salmon (*Oncorhynchus tshawytscha)* – Sacramento River Winter-Run ESU	E – 70Fed. Reg. 37160	58Fed. Reg. 33212*	79Fed. Reg. 42504
Coho salmon (*Oncorhynchus kisutch*) – Central California Coast ESU	E – 70Fed. Reg. 37160	64Fed. Reg. 24049*	77Fed. Reg. 54565
Coho salmon (*Oncorhynchus kisutch*) – Oregon Coast ESU	T – 73Fed. Reg. 7816	73Fed. Reg. 7816*	81Fed. Reg. 90780
Coho salmon (*Oncorhynchus kisutch*) – Southern Oregon and Northern California Coasts ESU	T – 70Fed. Reg. 37160	64Fed. Reg. 24049*	79Fed. Reg. 58750
Marine Invertebrates			
Black Abalone (*Haliotis cracherodii*)	E – 74Fed. Reg. 1937	76Fed. Reg. 66805	11/2020
White Abalone (*Haliotis sorenseni*)	E – 66Fed. Reg. 29046	-- --	73Fed. Reg. 62257
Sunflower Sea Star (*Pycnopodia helianthoides*)	T – 88Fed. Reg. 16212 (proposed)	-- --	-- --

Abbreviations: E – endangered; T – threatened; Fed. Reg. – Federal Register

* 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.

-- -- Indicates that critical habitat has not been designated and/or recovery plan has not been finalized for this species.

4.1 May Affect, Not Likely to be Adversely Affected

Once we have determined the action may affect ESA-listed or proposed species and designated critical habitat, the next step is differentiating between stressors that are NLAA and LAA for each listed species and critical habitat in the action area. An action warrants a NLAA finding when its effects are completely beneficial, discountable, or insignificant. Completely beneficial effects have an immediate positive effect without any adverse effects to the species or habitat. Completely 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, albeit positively. Discountable effects are those that could occur while an ESA-listed species is in the action area but, because of the intensity, magnitude, frequency, duration, or timing of the stressor, exposure to the stressor is extremely unlikely to occur. Insignificant effects relate to the response of exposed individuals where the response, in terms of an individual's growth, survival, or reproduction, would be immeasurable or undetectable, or an impact to the conservation value of a PBF of critical habitat would be immeasurable or undetectable. For stressors that meet these criteria for either completely beneficial, discountable, or insignificant, the appropriate conclusion is NLAA.

To assist us in reaching a determination, we perform a two-step assessment that considers all of the stressors identified in Section 2.5 of this opinion and all of the species and critical habitats within the action area (see Table 30) to understand the likelihood of the stressors having an effect

on the ESA-listed species or their critical habitat. First, we consider whether it is likely that a listed species or critical habitat is exposed to a stressor or there is a reasonable expectation of the stressor and an individual or habitat co-occurring. If we conclude that exposure of a species or critical habitat to a stressor caused by the proposed action or activity is discountable, we must also conclude it is NLAA. However, if exposure is probable, the second step is to evaluate the probability of a response to the stressor. When all stressors of an action are found to be NLAA for a listed species or a critical habitat, we conclude informal consultation for that species or critical habitat. Likewise, if a stressor associated with an action is found to be NLAA for all listed species and all critical habitats, there is no need to continue analyzing the consequences of that stressor in our Analysis of Effects (Section 6). Where the negative effects to any species or critical habitat or from any stressor to those species or critical habitat are found to exceed the standards of insignificant or discountable, we must analyze those consequences in our Analysis of Effects section.

4.1.1 Stressors Not Likely to Adversely Affect ESA-listed Species

In our 2024 HSTT Phase III reinitiated biological opinion (NMFS 2024f) Analysis of Effects section (Section 8.1, Table 4, pp. 102-104) we determined that some stressors associated with the action were not likely to adversely affect any ESA-listed species because any effects of that stressor would be either insignificant or discountable. Based on a review of newly available information (since completion of this opinion), we found no new information that would change our NLAA determination for any of these stressors. That is, there is no new information that reveals effects of the proposed action that may affect ESA-listed species in the action area in a manner or to an extent not previously considered for these NLAA stressors. Although changes have been made to the proposed action (i.e., from Phase III to Phase IV), including expansion of the action area and inclusion of modernization and sustainment activities, we have determined that, for the NLAA stressors discussed below, the action has not been modified in a manner that causes an effect to ESA-listed species or their habitat that was not previously considered in the 2024 reinitiated biological opinion. Therefore, we reaffirm our 2024 NLAA determinations for these stressors.

In the sections below, we summarize our findings regarding the likelihood of exposure, and how ESA-listed species and habitats are expected to respond, if exposed to these stressors. Any new (i.e., since the 2024 opinion) and relevant information for our analysis, either related to the effects of these stressors on species/habitats or changes in the proposed action, are discussed here as well.

Our analyses of the effects of NLAA stressors to species is organized by species group (i.e., marine mammals, sea turtles, fish, invertebrates) because, in general, we expect species within each group to respond in similar ways to these stressors. Where different responses by a species or subgroup (e.g., mysticetes versus odontocetes) within a species group have been documented in the literature, we consider those differences in our analysis. We then summarize our findings for each species group, and provide the rationale for our determination that the identified NLAA stressors resulting from the proposed action are not likely to adversely affect any of the ESA-

listed species within each species group. For some stressors, we determined there is no pathway to effects (i.e., no effect) to certain species (or species groups) and, therefore, no need to analyze those stressors in relation to those species.

Our analyses of the effects of NLAA stressors on critical habitat is provided in Section 4.1.3. For many NLAA stressors, we determined that there is no pathway to effects (i.e., no effect) to the PBFs of critical habitat and, therefore, no need to analyze those stressors here.

Acoustic Stressors

Vessel Noise

Action Proponent 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. Activities involving vessel movements occur intermittently and are variable in duration, ranging from a few hours up to two weeks.

The ESA-listed marine mammals, sea turtles and fish considered in this opinion will be exposed to noise from Action Proponent vessels during training, testing, range modernization and sustainment activities in the action area. Vessel source sound levels are below those that would cause hearing loss or auditory injury in ESA-listed species. Therefore, the focus of our vessel noise analysis is on the behavioral, physiological and potential masking effects of this stressor. Behavioral and physiological responses to vessels can be caused by multiple factors. Since it is difficult to separate the effects of vessel noise and vessel presence, this analysis will consider both aspects. Additional discussion of vessel noise as a potential stressor is included in Section 2.5.1.

<u>Marine Mammals</u>

Mysticetes demonstrate a variety of responses to vessel traffic and noise, from not responding at all to both horizontal (swimming away) and vertical (increased diving) avoidance (Watkins 1981a; Baker et al. 1983a; Reeves et al. 1998; Nowacek et al. 2004; Gende et al. 2011). Other common responses include changes in vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au and Green 2000; Williams et al. 2002; Richter et al. 2003). The likelihood of response may be driven by the distance or speed of the vessel, the animal's behavioral state, or by the prior experience of the individual or population (Watkins 1981a; Leatherwood et al. 1982; Bernasconi et al. 2012)

Mysticetes have been shown to both increase (Doyle et al. 2008; Melcon et al. 2012) and decrease (Sousa-Lima and Clark 2008; Castellote et al. 2012; Dearden 2025) calling behavior in the presence of vessel noise depending on the context (i.e., types/purpose of calls produced and type of activity engaged in). Changes in the amplitude or frequency of vocalizations in the presence of increased vessel noise have also been observed (Parks et al. 2007; Parks 2011).

Overall baleen whale responses to vessel noise and traffic are varied but are generally minor, and habituation or disinterest seems to be the predominant long-term response. If baleen whales do avoid ships, they do so by altering their swim and dive patterns to move away from the vessel, but no strong reactions have been observed. In many cases, whales do not appear to change their behavior at all. This may result from habituation by the whales, but may also result from reduced received levels near the surface due to propagation, or due to acoustic shadowing of the propeller cavitation noise by the ship's hull.

Some previous studies have documented odontocetes reacting neutrally to vessels, although both avoidance and attraction behavior have also been observed (Hewitt 1985; Wursig et al. 1998; Bejder et al. 2006). Sperm whales generally react only to vessels approaching within several hundred meters. However, some individuals may display avoidance behavior, such as quick diving (Wursig et al. 1998; Magalhaes et al. 2002) or a decrease in time spent at the surface (Isojunno and Miller 2015). One study showed that after diving, sperm whales showed a reduced time frame before they emitted the first click than prior to a vessel interaction (Richter et al. 2006). Azzara et al. (2013) also found a reduction in sperm whale clicks while a vessel was passing, as well as up to a half hour after the vessel had passed.

Vessel traffic exposes Southern Resident killer whales to several threats that have consequences for the species' likelihood of survival and recovery. 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. In addition to the disturbance associated with the presence of vessels, vessel traffic affects the acoustic ecology of Southern Resident killer whales, which can affect their social ecology and disrupt feeding behavior.

Pinniped reactions to vessels are variable and reports include a wide spectrum from avoidance and alert, to cases where animals in the water are attracted, and cases on land where there is a lack of significant reaction suggesting habituation to or tolerance of vessels (Richardson et al. 1995c). 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. (2007), pinniped responses to vessels are affected by the context of the situation and by the animal's experience.

Marine mammals may also experience masking due to vessel noises. Vessel noise can potentially mask vocalizations and other biologically important sounds (e.g., sounds of prey or predators) that marine mammals may rely on. Potential masking can vary depending on the ambient noise level within the environment, the received level and frequency of the vessel noise, and the received level and frequency of the sound of biological interest. In the open ocean, ambient noise levels are between about 60 and 80 dB re: 1 µPa in the band between 10 Hz and 10 kHz due to a combination of natural (e.g., wind) and anthropogenic sources (Urick 1983), while inshore noise levels, especially around busy ports, can exceed 120 dB re: 1 µPa. When the noise level is above the sound of interest, and in a similar frequency band, masking could occur. This analysis

assumes that any sound that is above ambient noise levels and within an animal's hearing range may potentially cause masking. However, the degree of masking increases with increasing noise levels; a noise that is just detectable over ambient levels is unlikely to cause any substantial masking.

The ESA-listed marine mammals considered in this opinion will be exposed to noise from Action Proponent vessels during training, testing and other military readiness activities in the action area. As documented above, vessel noise has the potential to disturb marine mammals and elicit an alerting, avoidance, or other behavioral reaction. These reactions are anticipated to be short-term, likely lasting the amount of time the vessel and the whale are in close proximity (e.g., Watkins 1981a; Richardson et al. 1995c; Magalhaes et al. 2002), and not consequential to the animals. Additionally, the effects of any temporary masking specifically from Action Proponent vessels are expected to be of a short duration and not result in meaningful changes to an animal's ability to communicate or detect biologically relevant cues given: 1) the intermittent and sporadic nature of military vessel activity, 2) the fact that the sound (i.e. vessel) and the receiver (whale) are both moving continuously and likely in different directions, 3) the background noise levels in the action area independent of military vessels, and 4) the small percentage of vessel traffic that Action Proponent vessels represent in the action area (as discussed in Section 2.5.1). While vessel noise and disturbance, in general, has been identified as a threat to some ESA-listed marine mammals (e.g., Southern Resident killer whales), this is due to the large volume and continuous or chronic nature of commercial and recreational vessel noise in particular areas occupied by these species (e.g., Puget Sound). By contrast, HCTT vessel noise is intermittent, dispersed over a very large area, and represents an extremely small proportion of the total vessel activity throughout the action area.

In summary, ESA-listed marine mammals are either not likely to respond to vessel noise and disturbance 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. Therefore, the effects of vessel noise on ESA-listed marine mammals are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Additionally, the effects of any temporary masking specifically from Action Proponent vessels is insignificant given the background noise levels in the action area independent of these vessels, and the small percentage of vessel traffic these vessels represent in the action area. Thus, we conclude that the effects of vessel noise and disturbance resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

Action Proponent vessels used during training and testing activities will produce low-frequency, broadband underwater sound below 1 kHz (for larger vessels), and higher-frequency sound between 1 kHz to 50 kHz (for smaller vessels), although the exact level of sound produced varies

by vessel type. ESA-listed turtles could be exposed to a range of vessel noises within their hearing abilities. Depending on the context of exposure, potential responses of sea turtles to vessel noise and disturbance include startle responses, avoidance or other behavioral reactions, and physiological stress responses.

Very little research exists on sea turtle responses to vessel noise disturbance. Currently, there is nothing in the available literature specifically aimed at studying and quantifying sea turtle response to vessel noise. However, a study examining vessel strike risk to green sea turtles suggested that turtles may habituate to vessel sound and may be more likely to respond to the sight of a vessel rather than the sound of a vessel, although both may play a role in prompting reactions (Hazel et al. 2007). Regardless of the specific stressor associated with vessels to which turtles are responding, they only appear to show responses (avoidance behavior) at approximately 10 m or closer (Hazel et al. 2007). Therefore, the noise from vessels is not likely to affect sea turtles from further distances, and disturbance may only occur if a sea turtle hears a vessel nearby or sees it as it approaches. These responses appear limited to non-injurious, minor changes in behavior based on the limited information available on sea turtle response to vessel noise. Díaz et al. (2024) found that while travelling green sea turtles in the Galapagos Archipelago increased their vigilance with increasing vessel noise. However, increased vigilance with increasing noise levels was not observed when the turtles were on the seabed. Their findings illustrate that vessel noise has the potential to alter overall time budgets of sea turtles.

Based on the available information, vessel noise is expected to cause minimal behavioral disturbance to sea turtles. If a sea turtle detects a vessel and avoids it, or has a stress response from the noise disturbance, these responses are expected to be temporary and only endure while the vessel transits through the area where the sea turtle encountered it. Therefore, sea turtle responses to vessel noise disturbance are considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated), and a sea turtle would be expected to return to normal behaviors and stress levels shortly after the vessel passes by. Thus, we conclude that the effects of vessel noise and disturbance resulting from the proposed action may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

ESA-listed fish considered in this opinion may be exposed to sound from vessel movement during Action Proponent training, testing, and other military readiness activities. In general, information regarding the effects of vessel noise on fish hearing and behaviors is limited. Vessel noise may 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. 1995; Engas et al. 1998; Mitson and Knudsen 2003). Physiological responses have also been documented for fish exposed to increased boat noise. 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 these were controlled studies. In contrast, wild fish such as those considered in this opinion are not hindered from movement away from an irritating sound source, if detected, so 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. Some temporary threshold shift (TTS) hearing loss has been observed in fish exposed to elevated background noise and other white noise, a continuous sound source similar to noise produced from vessels. However, TTS from vessel noise has not been documented for fish in the wild under normal vessel traffic conditions.

All fish species can detect vessel noise due to its low-frequency content and their hearing capabilities. Action Proponent vessels produce moderate to low-level passive sound sources (larger ships would produce low-frequency, broadband underwater sound below one kHz; and smaller vessels emit higher-frequency sound between 1 kHz to 50 kHz). Therefore, ESA-listed fish 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 vessels is unlikely to result in direct injury, hearing impairment, or other trauma to fish. Plus, in the near field, fish are able to detect water motion as well as visually locate an oncoming vessel. In these cases, most fish 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. However, impacts from Action Proponent 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. Instead, the only impacts expected from exposure to Action Proponent vessel noise for fish may include temporary auditory masking, physiological stress, or minor changes in behavior.

Therefore, similar to marine mammals and sea turtles, exposure to vessel noise for fish could result in short-term behavioral or physiological responses (e.g., avoidance, stress). Vessel noise would only result in brief periods of exposure for fish 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 from exposure to vessel noise (i.e., short-term physiological stress, masking, or behavioral reactions) resulting from the proposed action are considered insignificant, and thus are not likely to cause adverse effects to the following ESA-listed fish: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Invertebrates

Compared to some other taxa of marine animals (e.g., fishes, marine mammals), little information is available on the potential impacts on invertebrates from exposure to sound-producing activities (Hawkins et al. 2015). Since we could not find any studies directly addressing the impacts of sound on the ESA-listed invertebrates considered in this opinion (i.e., abalone and sea stars), we use information from invertebrate studies as a proxy for the possible range of responses these species may experience when exposed to vessel noise and other sound sources.

Most marine invertebrates, including corals and mollusks, possess specialized mechanisms for detecting acoustic stimuli that differ fundamentally from those of vertebrates. Rather than responding primarily to sound pressure, invertebrates are mainly sensitive to particle motion, particularly when they live close to the seabed where substrate vibration plays a significant role (Solé et al. 2023). This detection occurs through various specialized organs, including ciliated hair cells and statocysts, which allow these organisms to sense local water movements and maintain equilibrium (Solé et al. 2023). The detection range is typically limited, as particle motion diminishes rapidly with distance from the sound source, making the impact of noise highly dependent on proximity to its source. Abalone and sea star hearing is expected to be similar to other marine invertebrates, which are generally thought to perceive sound via either external sensory hairs or internal statocysts. Many aquatic invertebrates have ciliated “hair” cells that may be sensitive to water movements, such as those caused by currents or water particle motion very close to a sound source (Budelmann 1992b; Budelmann 1992a; Mackie and Singla 2003). This may allow sensing of nearby prey or predators, or help with local navigation. Detection of particle motion is thought to occur in mechanical receptors found on various body parts (Roberts et al. 2016). Aquatic invertebrates that are able to sense local water movements with ciliated cells include molluscs such as abalone (Budelmann 1992b; Budelmann 1992a; Popper et al. 2001). Like other invertebrates, abalone and sunflower sea stars are likely only sensitive to water particle motion caused by nearby low-frequency sources, and likely do not sense distant or mid- and high-frequency sounds(Navy 2018)Navy 2018a).

Different invertebrate species exhibit distinct behavioral modifications when exposed to anthropogenic noise. Recent research has documented varied responses across different taxonomic groups. For instance, razor clams respond to white noise and sine waves by initiating defensive digging behavior at sound levels around 100 dB re: 1 µPa, while showing no response at lower levels. In contrast, Mediterranean mussels exposed to acoustic sweeps demonstrated no significant behavioral changes, even at relatively high sound pressure levels. These divergent responses highlight the species-specific nature of behavioral adaptations to acoustic disturbance. ESA-listed corals demonstrate limited sensitivity to acoustic stressors from military sonar operations. These invertebrate species primarily detect sound through particle motion rather than pressure waves, with sensory capabilities largely restricted to detecting water movement using receptors on their tentacles. The exterior cilia of coral larvae may help them detect nearby water movements, but their ability to sense acoustic signals is confined to very short distances from the source. This biological limitation naturally reduces the potential impact of sonar operations on coral populations.

In general, the effects of acoustic stressors (including vessel noise) on the ESA-listed invertebrates considered in this opinion are largely unknown. We anticipate that exposure to vessel noise for abalone and sea stars could result in short-term behavioral or physiological responses (e.g., avoidance, stress). Vessel noise would only result in brief periods of exposure for these species 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 from exposure to vessel noise (i.e., short-term physiological stress, masking, or behavioral reactions) resulting from the proposed action are considered insignificant, and thus are not likely to cause adverse effects to white abalone, black abalone or sunflower sea stars.

Aircraft Overflight Noise and Disturbance

Aircraft overflights will usually occur near Navy airfields, installations, and in special use airspace within Navy Range Complexes. Additional discussion of aircraft overflight noise as a potential stressor is included in Section 2.5.1.

Marine Mammals

Marine mammals may respond to both the physical presence and 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 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. Thorough reviews of the subject and available information is presented in Richardson et al. (1995c) and elsewhere (e.g., Efroymson et al. 2001; Luksenburg and Parsons 2009; Holst et al. 2011; Smith et al. 2016). 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 (Manci et al. 1988; Holst et al. 2011).

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Koski et al. 1998). Richardson et al. (1985) and Richardson et al. (1995c) found no evidence that single or occasional aircraft flying above mysticetes causes long-term displacement of these mammals. Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns (Richardson et al. 1985). Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 150 m or higher. The bowheads exhibited fewer behavioral changes than did the odontocetes in the same area (Patenaude et al. 2002).

Several studies have corroborated the variability in sperm whales' reactions to fixed-wing aircraft or helicopters (Wursig et al. 1998; Richter et al. 2003; Richter et al. 2006; Smultea et al. 2008). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al. 1995c). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 ft) 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).

Smaller delphinids generally react to overflights either neutrally or with a startle response (Wursig et al. 1998). Beluga whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns to a greater extent than mysticetes in the same area (Patenaude et al. 2002). These reactions increased in frequency as the altitude of the helicopter dropped below 150 m. A change in travel direction was noted in a group of pilot whales as the aircraft circled while conducting monitoring (HDR 2011).

Richardson et al. (1995b) noted that pinniped 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 (Gjertz and Børset 1992; Blackwell et al. 2004). Pinniped 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 they are 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 m 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 m (Richardson et al. 1995b).

Helicopters are used in studies of several species of seals hauled out and are considered an effective means of observation (Gjertz and Børset 1992; Bester et al. 2002), 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 Nicholas 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 haul-out 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 Nicholas Island (Holst et al. 2011).

Pinnipeds may be 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).

In most cases, exposure of a marine mammal to fixed-wing aircraft presence and noise resulting from the proposed action would last for only seconds as the aircraft quickly passes overhead, typically at altitudes above 3,000 ft, which would make marine mammals unlikely to respond. 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 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. For example, the average underwater received level detected by Kuehne et al. (2020) produced by Growler aircraft (EA-18G) in the strongest 1-second window was 134 ± 3dB re: 1µPa at 30 meters below the sea

surface. However, since these take-off levels would occur for a small amount of time (seconds), it is extremely unlikely that ESA-listed marine mammals would co-occur in time and space with aircraft sound at levels that could result in adverse effects. Any exposures of cetaceans to aircraft noise that may occur this close to the surface would likely be short-term and infrequent, resulting in either no measurable response or a brief, inconsequential behavioral response.

Acoustic energy in the air does not effectively cross the air-water interface and noise can be reflected off the water surface (Richardson et al. 1995b). Sound waves in air can be reduced by a factor of more than a thousand when they cross the air-sea interface (Hildebrand 2005), constituting a substantial sound barrier. Some sonic booms from aircraft could startle marine mammals, but these events are transient and happen infrequently at any given location within the action area. Research conducted by the U.S. Air Force reported significant reductions in noise underwater related to in air sonic booms[10], indicating a lack of harassment risk for protected marine species under water (USAF 2000). Due to the short term and infrequent nature of any exposures to fixed-wing aircraft, and the brief responses that would likely follow such exposure, the effects of fixed-wing aircraft overflight on ESA-listed marine mammals is insignificant.

Low flight altitudes of rotary-wing aircraft during some anti-submarine warfare and mine warfare activities, often under 100 ft, may elicit a somewhat stronger behavioral response due to the proximity to marine mammals, the slower airspeed and therefore longer exposure duration, and the downdraft created by the helicopter's rotor. Marine mammals 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 these aircraft typically transit open ocean areas within the action area. The literature cited above indicates that aircraft noise would cause only small temporary changes in the behavior of marine mammals. Specifically, marine mammals at or near the surface when a helicopter flies overhead at low altitude may startle, divert their attention to the aircraft, or avoid the immediate area by swimming away or diving.

In summary, due to the short-term nature of any exposures to aircraft and the brief responses that could follow such exposure, the effects of aircraft overflight noise on ESA-listed marine mammals are insignificant (i.e., so minor that the effects cannot be meaningfully evaluated). Thus, we conclude that the effects of aircraft overflight noise and disturbance resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

[10] When the aircraft exceeds the speed of sound, pressure waves around the aircraft form shock waves, the sound heard on the ground as a "sonic boom" is an impulsive noise, similar to thunder, resulting from the sudden onset and release of pressure after the buildup by the shock wave. https://www.af.mil/About-Us/Fact-Sheets/Display/Article/104540/sonic-boom/

Sea Turtles

Noise produced from aircrafts have the potential to propagate underwater at levels that could be detectable to sea turtles. As with vessel disturbance above, little information is available on how ESA-listed sea turtles respond to aircraft. For the purposes of this consultation, we assume all sea turtles may exhibit similar short-term behavioral responses such as diving, changes in swimming, etc., which is also consistent with those behaviors observed during aerial research surveys of sea turtles. While we are unaware of any data on the physiological responses sea turtles exhibit to aircraft, we assume that a low-level, short-term stress response is also possible.

The working group that developed the 2014 *ANSI Guidelines* for sea turtles did not consider this specific acoustic stressor for sea turtles, in part because aircraft noise is not considered to pose a great risk (Popper et al. 2014a). Although the aircraft used by the Action Proponents can produce extensive airborne noise from either turbofan or turbojet engines, via sonic booms, and from rotary wing aircraft low-frequency sound and vibration, depending on the altitude, some sound would not be transmitted into the water, and any low-flying altitude air craft would only likely transmit low levels of sound within one meter into the water column. As mentioned previously, we assume that sea turtles located at or near the water surface may exhibit startle reactions to certain aircraft overflights if the aircraft is flying at a low altitude and the turtle can see it or detect it through sound or water motion generated from wind currents on the surface. This would most likely occur when helicopters are hovering (other aircraft are mostly flying at higher altitudes) and might be visually detected by a sea turtle. The currents and waves the helicopter produces on the water's surface may also cause sea turtles to respond to the disturbance along with the sound. Longer duration activities (such as a couple of hours) involving periods of time where helicopters hover may increase the chance that a sea turtle will startle, change swimming patterns, or have a physiological stress response. 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 extremely low.

A sea turtle could leave the immediate 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 ESA-listed sea turtles is considered temporary and insignificant (i.e., so minor that the effects cannot be meaningfully evaluated). Thus, we conclude that the effects of aircraft overflight noise and disturbance resulting from the proposed action may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle –

all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

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. Thus, we conclude that the effects of aircraft overflight noise and disturbance resulting from the proposed action are considered insignificant, and are not likely to cause adverse effects to the following ESA-listed fish species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Weapons Firing Noise

All of the noise produced during the use of the weapons described here is expected to be brief (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. Gunfire activities could produce multiple shots fired over a brief period of time. The Action Proponents will implement mitigation to avoid or reduce potential impacts from weapon firing noise during large-caliber gunnery activities, as discussed in Section 2.4.2. Additional discussion of weapons noise as a potential stressor is included in Section 2.5.1.

Marine Mammals: In-water Effects

This section addresses the effects of weapons firing noise to ESA-listed marine mammals in the water. In 6.1.2 we address the effects of weapons firing noise to ESA-listed pinnipeds when they are hauled out on land.

Marine mammals may be exposed to sounds caused by the firing of weapons, objects in flight, and inert impact of non-explosive munitions on the water's surface. A gun fired from a ship on the surface of the water propagates a blast wave away from the gun muzzle into the water. Most in-air sound would be reflected at the air-water interface (Navy 2024d). Underwater sounds would be strongest just below the surface and directly under the firing point. Any sound that enters the water only does so within a narrow cone below the firing point or path of the projectile (Navy 2024d). Yagla and Stiegler (2003) 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.

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. Many missiles are launched from aircraft, which would 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 marine mammals 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.[11] Animals 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.

For exposed marine mammals, 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 marine mammals is also extremely unlikely.

Marine mammals at the surface of the water, in the narrow footprint under a weapons trajectory, could be exposed to gunfire noise and may exhibit brief startle reactions, avoidance, diving, or no reaction at all. Behavioral reactions would likely be short-term (seconds to minutes) and due to the short-duration, transient nature of launch noise, animals are unlikely to be exposed multiple times within a short period. Masking of biologically relevant sounds is also not expected to occur because weapons noise will not persist for a long enough duration.

For the reasons above, the effects of weapons noise on ESA-listed marine mammals in the water are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Thus, we conclude that the effects of noise from weapons firing resulting from the proposed action may affect, but is not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

Individual sea turtles from all of the ESA-listed species may be exposed to sounds caused by the weapons firing (guns, missile, torpedoes), objects dropping in the water, and inert impact of non-explosive munitions on the water's surface. For exposed sea turtles, most of the weapons noise

[11] Note that the potential for objects to physically strike an ESA-listed marine mammal is discussed in section 9.1.1.4.

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 sea turtles is also extremely unlikely.

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. Masking of biologically relevant sounds is also not expected to occur because weapons noise will not persist for a long enough duration. Most activities producing weapons noise will typically occur more than 12 nm from the coast; therefore, sea turtles would still be able to detect biologically relevant sounds near the coastal areas they inhabit or need to detect near nesting beaches.

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 sea turtles from weapons noise exposure are anticipated to be minor, temporary and will not lead to a significant disruption of normal behavioral patterns. As such the effects from weapons noise on sea turtles is considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Thus, we conclude that the effects of noise from weapons firing resulting from the proposed action may affect, but is not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

Weapons noise could affect fish located at or near the surface of the water, albeit in a narrow footprint under a weapons trajectory, as described previously. 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). With the exception of green sturgeon that typically occur near the seafloor, all of the ESA-listed fish species in the action area may be affected by weapons noise at or near the surface.

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. Naval gunfire could elicit a brief behavioral reaction such as startle reactions or avoidance and could expose fishes to multiple shots within a few seconds. 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.

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.

In summary, any effects on fishes from weapons noise are anticipated to be minor, temporary, and are not expected to lead to a significant disruption of normal behavioral patterns. As such, the effects from weapons noise on fishes are considered insignificant. Thus, we conclude that the effects of noise from weapons firing resulting from the proposed action may affect, but is not likely to adversely affect the following species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Energy Stressors

The only energy stressors that may affect ESA-listed species are in-water electromagnetic devices and cables deployed for range modernization and sustainment. For the HSTT Phase III consultation, we also analyzed the effects of high-energy lasers, which we determined may affect but were not likely to adversely affect ESA-listed species. However, for HCTT Phase IV the Action Proponents have provided new information indicating that when contact with the target (i.e., surface vessel) is lost, the lasers automatically turn off, thereby removing any potential to impact marine resources. Therefore, based on the new information regarding how Navy high-energy lasers operate, we no longer consider this to be a "may affect" stressor and do not analyze it here.

In-water Electromagnetic Devices and Cables

Offshore training and testing activities that use in-water electromagnetic devices would occur in both the Hawaii and California portions of the action area. The Navy also proposes to deploy fiber optic cables, which generate an electromagnetic force, in the SOCAL RC. Additional discussion of in-water electromagnetic devices and cables and as potential stressors is included in Section 2.5.3.

Marine Mammals

Normandeau et al. (2011) concluded there was behavioral, anatomical, and theoretical evidence indicating cetaceans sense magnetic fields. Fin, humpback, and sperm whales have shown positive correlations with 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. 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 (Klinowska 1985; Kirschvink 1990; Walker et al. 1992). 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 field allowing animals to monitor their progress on this map (Klinowska 1990).

If a marine mammal was in proximity of an electromagnetic field source associated with training or testing, emitting a field strong enough to be detected, and that animal is sensitive to the exposure, it is conceivable that the electromagnetic field could have an effect on a marine mammal, primarily impacting that animal's navigation. 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.

The static magnetic fields generated by electromagnetic devices used in HCTT activities are of relatively minute strength. The maximum strength of the magnetic field is approximately 2,300 microteslas, with the strength of the field decreasing further from the device. At a distance of four meters from the source of a 2,300 microtesla magnetic field, the strength of the field is approximately 50 microteslas, which is within the range of the Earth's magnetic field (25 to 65 microteslas). At 8 m, the strength of the field is approximately 40% of the Earth's magnetic field, and only 10% at 24 m away from a 2,300 microtesla magnetic field (Navy 2024d). Therefore, at a distance of 200 m (the maximum predicted distance of the magnetic field proposed for use by the Navy) the strength of the magnetic field would be approximately 0.2 microteslas (Navy 2024d), which is less than 1% of the strength of the Earth's magnetic field. A marine mammal would have to be present within the electromagnetic field (approximately 200 m from the source) during the activity in order to detect it, though detection does not necessarily signify a significant biological response. Once the source is turned off or moves from the location, the electromagnetic field is gone. 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), and the anticipated density of marine mammals in these areas, we consider it extremely unlikely that an ESA-listed marine mammal would be exposed to electromagnetic energy at sufficient intensities to create an adverse effect through behavioral disruption or otherwise.

Cables deployed on the seafloor during range modernization and sustainment activities generate an electromagnetic force that is less than that of the natural background magnetic force of the earth at distances beyond 0.6 cm (0.25 in) from the cable (Navy 2024d). As electromagnetic energy dissipates exponentially by distance from the energy source, the magnetic field from the cable would be equal to 0.1% of the Earth's at a distance of 6 m (20 ft). Given the location of the cables on the seafloor and the low intensity of the electromagnetic force, we consider it extremely unlikely that an ESA-listed marine mammal would be exposed at sufficient intensities to result in an adverse effect through behavioral disruption, injury or other effects.

Therefore, potential effects from electromagnetic devices used for training and electromagnetic forces from cables are discountable. Thus, we conclude that the effects of energy stressors resulting from the use of in-water electromagnetic devices or deployment of cables may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

Magnetic fields and other cues (e.g., visual cues), are known to be important for sea turtle orientation and navigation (Lohmann et al. 2000; Putman et al. 2015). Sea turtles use geomagnetic fields to navigate at sea, and therefore changes in those fields could impact their movement patterns (Lohmann and Lohmann 1996; Lohmann and Lohmann 1998). Turtles in all life stages orient to the earth's magnetic field to position themselves in oceanic currents and directional swimming presumably aided by magnetic orientation has been shown to occur in some sea turtles (Christiansen et al. 2016). This life strategy enables them to locate seasonal feeding and breeding grounds and return to their nesting sites (Lohmann and Lohmann 1996; Lohmann and Lohmann 1998). Sea turtles have been shown able to detect changes in magnetic fields, which may cause them to deviate from their original direction (Lohmann and Lohmann 1996). For example, Liboff (2016) determined that freshly hatched sea turtles are able to detect and use the local geomagnetic field as a reference point before embarking on a post-hatchling migration. This study suggests that the information is transferred from the mother to the egg through some undetermined geomagnetic imprinting process.

Sea turtles may also use nonmagnetic cues for navigation and migration, and these additional cues may compensate for variations in magnetic fields. Putman et al. (2015) conducted experiments on loggerhead hatchlings and determined that electromagnetic fields may be more important for sea turtle navigation in areas that may constrain a turtle's ability to navigate by other means (cold temperatures or displacement from a migration route). The findings of this study suggest that the magnetic orientation behavior of sea turtles is closely associated with ocean ecology and geomagnetic environment (Putman et al. 2015). Other studies on behavioral responses of sea turtles to magnetic fields have been conducted on green and loggerhead sea

turtles (Normandeau et al. 2011). Loggerhead sea turtles have been shown to be sensitive to electromagnetic field intensities ranging from 0.005 to 4,000 microteslas. Green turtles were found to be sensitive to field intensities from 29.3 to 200 microteslas. Although data is lacking for other sea turtle species, the sensitivities of loggerhead and green turtles is assumed similar to other species.

As discussed above for marine mammals, the static magnetic fields generated by electromagnetic devices used in HCTT activities are of relatively minute strength. At a distance of 200 m the strength of the magnetic field would be approximately 0.2 microteslas (Navy 2024d), which is less than 1% of the strength of the Earth's magnetic field. This is likely within the range of detection for sea turtle species, but at the lower end of their sensitivity to the field. For any sea turtles located in the immediate area (within about 200 m) where in-water electromagnetic devices are being used, adult, sub-adult, juveniles, and hatchling sea turtles could be temporarily disoriented and could deviate from their original movements, but the extent of this disturbance is likely to be inconsequential given the brief duration of the potential disorientation (seconds or minutes). These brief behavioral disruptions are expected to be limited and minor, and not anticipated to result in any significant effect, beyond what would be similar to natural stressors regularly occurring in the animal's life cycle. Therefore, the effects from exposure to in-water electromagnetic devices for sea turtles are considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Cables deployed on the seafloor during range modernization and sustainment activities generate a very small electromagnetic force. As we concluded for marine mammals, given the location of the cables on the seafloor and the low intensity of the electromagnetic force, we consider it extremely unlikely that an ESA-listed sea turtle would be exposed at sufficient intensities to result in an adverse effect through behavioral disruption, injury or other effects.

Thus, we conclude that the effects of energy stressors resulting from the use of in-water electromagnetic devices or deployment of cables may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

ESA-listed fish in the action area may encounter in-water electromagnetic devices during training and testing activities. Some fish species have been shown to be sensitive to electromagnetic fields, based upon their orientation in the water column and prey location (Kirschvink 1997; Tricas and New 1997). The species most commonly studied for electromagnetic field sensitivity include elasmobranchs (sharks, skates, and rays), agnatha (lampreys), and bony fish (teleosts and chondrosteans). Elasmobranchs (rays,

skates, sharks) are particularly sensitive to electromagnetic fields as they use electromagnetic-receptive sensory systems for orientation, navigation, and locating conspecifics or buried prey.

Exposures to relatively high strength magnetic fields for days to weeks can have physiological effects on organisms. Formicki et al. (2019) reviewed effects on spermatozoa movement, fertilization rates, and egg incubation period in a range of fish species, and Juutilainen (2005) reports developmental effects in fish and sea urchin embryos from exposure to magnetic fields in the range of 0.1–10 microteslas. In addition, natural diurnal weak magnetic field variation could play a role in organisms' internal clocks, and magnetic disturbances may hence be able to cause chronobiologic disruptions, with potential health consequences for the organism (Liboff 2014) and effects of anthropogenic magnetic fields on homeostatic and metabolic functions have been suggested (Begall et al. 2013). Also, distorted magnetic fields during developmental phases have resulted in failed magnetic orientation later in life, perhaps by effects on an internal magnetic map, in loggerhead sea turtles and rainbow trout (Fuxjager et al. 2014; Putman et al. 2015).

However, such long-term exposure effects are likely not relevant in the context of the HCTT electromagnetic energy emissions which only disturb animals for a short period (minutes to hours). As discussed above for sea turtles, the static magnetic fields generated by electromagnetic devices used in HCTT activities are of relatively minute strength. Any behavioral or physiological response would likely be short-term, relatively minor, and would not result in a significant disruption of normal behavior patterns that include, but are not limited to, breeding, feeding or sheltering

In summary, while ESA-listed fishes may be exposed to in-water electromagnetic devices during training and testing activities the effects would be insignificant because of: (1) the relatively low intensity of the magnetic fields generated (0.2 microteslas at 200 m from the source), (2) the highly localized potential impact area, and (3) the limited and temporally distinct duration of the activities (hours). Therefore, the effects of in-water electromagnetic devices on ESA-listed fishes are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Thus, we conclude that the effects of energy stressors resulting from the use of in-water electromagnetic devices or deployment of cables may affect, but are not likely to adversely affect the following species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Physical Disturbance and Strike Stressors

This section analyzes the potential impacts of the various types of physical disturbance, including the potential for strike, during military readiness training and testing activities within

the action area from: in-water devices; military expended materials (including non-explosive practice munitions and fragments from high-explosive munitions); and seafloor devices.

In-water Devices (excluding Unmanned Surface Vehicles)

In-water devices are used in both offshore and inshore portions of the action area and could pose a collision risk to ESA-listed species. Additional discussion of in-water devices as a potential stressor is included in Section 2.5.4.

Given their size and potential operating speed, USVs are the in-water devices with the most potential to strike living marine resources. The risk of an USV striking an ESA-listed species will be analyzed, along with the risk of a manned vessel strike, in our Analysis of Effects (Section 6) below.

Marine Mammals

Most in-water devices do not have a realistic potential to strike living marine resources because they either move very 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 (Navy 2024d).

Despite thousands of military exercises in which in-water devices have been used, there have been no recorded instances of a marine mammal strike from these devices. In addition, the Action Proponents will implement mitigation, as described in Section 2.4.2, to further minimize the potential risk of an in-water device striking a marine mammal. For these reasons, we consider it extremely unlikely for any ESA-listed marine mammal to be struck by an in-water device. It is possible that marine mammals 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 the effects to the individual exposed would be insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Therefore, potential effects on ESA-listed marine mammals from in-water devices are discountable, in the case of strike risk, and insignificant in the case of behavioral disturbance. Thus, we conclude that the effects of in-water devices (excluding USVs) resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

As discussed above for marine mammals, with the exception of USVs (which are discussed below in Section 6 Analysis of Effects), it is extremely unlikely that an in-water device will strike a sea turtle because the proposed devices either move too slowly through the water column, or are closely monitored by an observer to ensure the towed device does not run into objects in the water. The Action Proponents will also implement mitigation, as described in Section 2.4.2, to further minimize the potential risk of an in-water device striking a sea turtle. It

is possible that sea turtles occurring in areas that overlap areas where in-water devices are used 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 the effects to the individual exposed would be insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Therefore, potential effects on ESA-listed sea turtles from in-water devices are discountable, in the case of strike risk, and insignificant in the case of behavioral disturbance. Thus, we conclude that the effects of in-water devices (excluding USVs) resulting from the proposed action may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

ESA-listed fish may overlap spatially with HCTT in-water devices in some portions of the action area. As discussed above for sea turtles, with the exception of USVs (which are discussed below in Section 6 Analysis of Effects), it is extremely unlikely that an in-water device will strike an ESA-listed fish because the proposed devices either move too slowly through the water column, or are closely monitored by an observer to ensure the towed device does not run into objects in the water (Navy 2024d). In addition, there has never been a recorded incidence of a Navy in-water device striking an ESA-listed fish. It is possible that ESA-listed fish occurring in areas that overlap areas where in-water devices are used 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 the effects to the individual exposed would be insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Therefore, potential effects on ESA-listed fish from in-water devices are discountable, in the case of strike risk, and insignificant in the case of behavioral disturbance. Thus, we conclude that the effects of in-water devices (excluding USVs) resulting from the proposed action may affect, but are not likely to adversely affect the following species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Seafloor Devices

Seafloor devices associated with the proposed action include items that are placed on, dropped on, or moved along the seafloor, such as mine shapes, anchor blocks, anchors, bottom-placed instruments, bottom-crawling unmanned underwater vehicles, and bottom-placed targets that are not expended. Additional discussion of seafloor devices as a potential stressor is included in Section 2.5.4.

Marine Mammals

Seafloor devices are either stationary or move very slowly along the bottom and do not pose a threat to highly mobile animals. Armored cables and fiber optic cables will be installed on the seafloor in the SOCAL RC as part of Range Modernization and Sustainment activities at specified locations within the action area. Both types of cable will be deployed from a slow-moving (0–3 knots) cable laying vessel, which operates continuously (day and night) until all cables are deployed and installed on the seafloor. Mitigation to reduce the probability of physical disturbance or strike during cable laying activities would be implemented as part of the activity.

The only seafloor device used during training and testing activities that has the potential to strike an ESA-listed marine mammal at or near the surface is an aircraft deployed mine shape, which is used during aerial mine laying activities. These devices are identical to non-explosive practice bombs, and the analysis of the potential impacts from those devices are covered below in *Military Expended Materials*. We consider it extremely unlikely for any ESA-listed marine mammals to 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 are likely to behaviorally avoid them. 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 the slow movement and relatively small size of seafloor devices, the effort expended by individuals to avoid them will be minimal and temporary, and will not have fitness consequences. Any avoidance behavior would be of short duration and intensity such that the effects to the individual exposed would be insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Thus, we conclude that the effects of seafloor devices resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

Sea turtles could potentially be exposed to seafloor devices when foraging in benthic habitats (e.g., loggerhead, olive ridley, hawksbill, and green sea turtles) or throughout the water column in deep waters (e.g., leatherback sea turtle). Additionally, some sea turtle species in coastal habitats can occur near the bottom when resting. For activities that occur in offshore waters, the species and age classes that may be impacted by seafloor devices are juvenile, sub-adult, and adult loggerhead, olive ridley, green, and hawksbill turtles, especially if seafloor devices are placed in waters where the depths are within benthic foraging ability dive depths. Juvenile and adult leatherback turtles are more likely to co-occur where seafloor devices are used in offshore areas given their preference for open-ocean habitats and feeding throughout the water column, and may dive to depths greater than 1,000 m. All sea turtle species are expected to be greatly dispersed throughout offshore waters of the action area.

For nearshore activities, juvenile, sub-adult, and adult leatherback and loggerhead sea turtles, as well as green, olive ridley, and hawksbill sea turtles that are located in benthic foraging habitats may overlap with areas where seafloor devices are used. However, the likelihood of a sea turtle encountering and being disturbed by a seafloor device in benthic foraging habitats is low because these devices are either stationary or move very slowly along the bottom, making them easily avoidable by a sea turtle. Sea turtles would be expected to ignore or avoid any slowly moving or stationary seafloor device. Given the slow movement of seafloor devices, the effort expended by sea turtles to avoid them will be minimal, and behavioral impacts will be temporary, and insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Thus, we conclude that the effects of seafloor devices resulting from the proposed action may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

The deployment of various devices on the seafloor during military operations introduces potential interaction risks with marine fish populations. These interactions merit careful examination to understand their ecological implications and assess the actual level of risk posed to different fish species. Fish demonstrate natural behavioral tendencies to aggregate around structures in the water column. Static fish aggregating devices, which are anchored to the seafloor, serve as primary examples of how fish interact with artificial structures in marine environments. These devices effectively attract various fish species seeking food sources or refuge, as they create microhabitats where smaller organisms settle and attract larger predatory fish. This natural attraction behavior suggests that fish might similarly be drawn to other deployed devices, such as non-explosive mine assemblies or monitoring equipment.

Fish possess sophisticated sensory mechanisms that enable them to navigate their environment and avoid collisions with stationary objects. The lateral line system enables fish to identify and avoid fixed objects in their path, significantly reducing the likelihood of accidental collisions with deployed devices. This natural avoidance capability is particularly relevant when considering the potential risks associated with anchored or tethered objects in the water column. Aircraft- and vessel-deployed mine shapes, anchor blocks, anchors, bottom-placed instruments, and targets all have the potential to strike fish upon deployment as they are sinking through the water column and settling on the seafloor.

The analysis of potential strike risks from seafloor devices reveals that while fish may be attracted to these structures, their natural sensory capabilities and avoidance behaviors effectively minimize the risk of harmful interactions. The use of seafloor devices in inshore waters would be temporary (days to weeks) and the amount of soft bottom habitat impacted during that short timeframe would be negligible compared to the remaining amount of habitat that would be unaffected by the activities. These findings support the conclusion that deployed

seafloor devices pose a discountable strike risk to ESA-listed fish, negating the need for extensive additional protective measures beyond standard operational protocols.

Given the slow movement of seafloor devices, the effort expended by fish to avoid them will be minimal, and behavioral impacts will be temporary, and insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). Thus, we conclude that the effects of seafloor devices resulting from the proposed action may affect, but are not likely to adversely affect the following species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Invertebrates

HCTT seafloor devices are typically placed in soft bottom areas that are generally not preferred habitat for ESA-listed invertebrates in the action area. Most abalone rocky habitat at San Clemente and along the California coast is too shallow to meet training and testing requirements that would use seafloor devices. Some shallow water seafloor devices are used by the Navy along the coast at Silver Strand, but only at designated sandy, soft bottom areas not associated with ESA-listed abalone habitat (Navy 2018). Inert (non-explosive) mine laying would not affect abalone since the Navy has committed to conducting this activity at depths greater than 60 m and in sandy areas (away from kelp and rocky bottom) where ESA-listed abalone are not found. Therefore, direct impacts from seafloor devices on black abalone and white abalone habitat are extremely unlikely (i.e., discountable). Disturbance of soft substrate by seafloor devices may cause temporary and localized episodes in turbidity, but the impact on ESA-listed invertebrates nearby would likely be insignificant. Although sunflower sea stars may occur in sandy, soft bottom areas where seafloor devices are deployed, given the low density of sunflower sea stars in the action area, we find it extremely unlikely that such a device would physically disturb or strike a sunflower sea star. Sunflower sea stars are mobile benthic animals. Given the slow movement of seafloor devices, the effort expended by sunflower sea stars to avoid them will be minimal, and behavioral impacts will be temporary, and insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Precision anchoring could affect abalone or sunflower sea star habitat if conducted in shallow water habitats where these species occurs. However, under the proposed action, precision anchoring would only be conducted at U.S. Coast Guard designated anchorages at the mouth of San Diego where ESA-listed abalone do not occur. The Action Proponents have committed to conducting precision anchoring at depths from 18-60 ft and over sandy or loose fragmented shell bottom where abalone are not likely to be found (Navy 2024d). Precision anchoring would not be conducted around San Clemente Island or other parts of the HCTT action area where black, white abalone or sunflower sea stars are known to occur (Navy 2024d).

Thus, we conclude that the effects of seafloor devices resulting from the proposed action may affect, but are not likely to adversely affect the following species: black abalone, white abalone, and sunflower sea star.

Military Expended Materials

Training and testing activities in the action area include firing a variety of weapons and employing a variety of explosive and non-explosive rounds including bombs; small-, medium-, and large-caliber projectiles; or sinking exercises and testing events with ship hulks (Navy 2024d). During these training and testing activities, various types of munitions and non-munitions may be introduced and expended into the marine environment, which collectively are referred to as military expended materials. Military readiness activities that involve military expended materials would occur throughout the action area, with the exception of shallow nearshore areas, San Diego Bay, and Pearl Harbor. Additional discussion of military expended materials as a potential stressor is included in Section 2.5.4.

This section analyzes the strike potential to ESA-listed species from the following categories 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. The primary concern is the potential for an animal to be hit with a military expended material at or near the water's surface, which could result in injury or death. While disturbance or strike from an item falling through the water column is possible, it is considered extremely unlikely because the objects generally sink slowly through the water based on the weights of expended materials and can be avoided by ESA-listed marine mammals, sea turtles, and fish. Therefore, the discussion of military expended materials strikes focuses on the potential of a strike at or near the surface of the water.

Marine Mammals

Although a strike of a marine mammal from military expended materials has never been reported or recorded by the Action Proponents, the possibility of a strike still exists (Navy 2024d). The Action Proponents estimated potential direct strike exposures based on a modeled scenario using the marine mammal species with the highest average seasonal density in areas with the highest military expended materials throughout the action (i.e., highest likelihood approach). Input values include materials data (frequency, footprint, and type), size of the training or testing area, marine mammal density data, and size of the animal. To estimate the potential of military expended materials to strike a marine mammal, the impact area of each category of military expended materials analyzed for marine mammals was totaled over one year in the area with highest combined amounts of military expended materials. The results indicate that it is extremely unlikely that an ESA-listed marine mammal would be struck by military expended materials as a result of the proposed action. For details on this approach, we refer the reader to Section 5.5.1.3.2 (pp. 5-168 to 5-170) of the HCTT BA, which we incorporate here by reference (Navy 2024d). Due to the low density and highly dispersed nature of ESA-listed marine mammals in the pelagic environment, the potential of any individual animal occurring within the

small area where projectiles are fired is extremely unlikely. The Action Proponents will also implement mitigation, as described in Section 2.4.2, to further minimize the potential risk of military expended materials striking a marine mammal.

Thus, we consider it extremely unlikely for any ESA-listed marine mammal to be directly struck by HCTT military expended materials. An individual marine mammal could be exposed to observing or hearing the projectile hit the water and travel through the water column. 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 (i.e., so minor that the effect cannot be meaningfully evaluated). For these reasons, the effects on ESA-listed marine mammals from physical disturbance and strike with military expended materials are insignificant, in the case of behavioral disturbance, or discountable, in the case of strike. Therefore, we conclude that the effects of military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

While proposed activities involving military expended materials occur both in nearshore and offshore areas, the majority of materials would be expended in offshore areas. During activities that produce military expended materials, the potential for a strike of ESA-listed sea turtles exists from all sizes of non-explosive practice munitions, fragments from high-explosive munitions, expendable targets, and expended materials other than munitions; such as sonobuoys, expended bathythermographs, and torpedo accessories. The chance of a turtle being hit is related to the sea turtle's life history and likelihood of presence in the action area when materials are expended. The primary concern with these materials is for a sea turtle located at or near the water surface to be struck. If this occurs, a turtle could potentially be injured or killed.

For military readiness activities occurring in offshore waters, the species and age classes that could overlap with activities involving military expended materials are hatchlings and pre-recruitment juveniles of all sea turtle species, adult loggerhead turtles, and leatherback turtles of all age classes. Adult sea turtles in these areas could be located at the surface, but generally spend most of their time submerged. Thus, adult sea turtles are expected to be at the surface for brief periods of time compared to hatchlings and juveniles, as these early life stages spend more time at the surface while in ocean currents. However, all life stages do spend some time at the surface basking.

Using the same approach as described for marine mammals (above), the Action Proponents estimated potential direct strike exposures to sea turtles based on a modeled scenario. Based on estimated sea turtle densities and volume of military expended materials, the highest likelihood

of a strike would be to green sea turtles within the Hawaii Range Complex. The modeling results indicate that it is extremely unlikely that an ESA-listed sea turtle would be struck by military expended materials as a result of the proposed action. For details on this approach, we refer the reader to Section 5.4.1.4.2 (pp. 5-120 to 5-121) of the HCTT BA, which we incorporate here by reference (Navy 2024d). Due to the low density and highly dispersed nature of ESA-listed sea turtles in the pelagic environment, the potential of any individual animal occurring within the small area where projectiles are fired is extremely unlikely. The Action Proponents will also implement mitigation, as described in Section 2.4.2, to further minimize the potential risk of military expended materials striking a marine mammal.

Thus, we consider it extremely unlikely for any ESA-listed sea turtle to be struck by military expended materials. Any individual sea turtles 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 (i.e., so minor that the effect cannot be meaningfully evaluated). For these reasons, the effects on ESA-listed sea turtles from physical disturbance and strike with military expended materials are insignificant, in the case of behavioral disturbance, or discountable, in the case of strike. Thus, we conclude that the effects of military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following sea turtle species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

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. The probability of direct strikes must be considered within the context of species abundance and distribution patterns. The analysis of military expended material strike risks reveals a complex interaction between physical factors and biological characteristics. Different fish species exhibit varying levels of vulnerability to fragment strikes based on their physical characteristics and position in the water column. Factors such as body size, scale thickness, and swimming capabilities influence their susceptibility to injury from explosive fragments. Fish with greater mobility may have better chances of avoiding fragments as they disperse through the water column. Fragments produced by exploding munitions at or near the surface may present a high-speed strike hazard for an animal at or near the surface. Marine environments where military readiness activities occur contain multiple fish species that frequent surface waters, making them potentially vulnerable to strikes from expended materials. Oceanic whitetip sharks, scalloped hammerhead sharks, giant manta rays, and most salmonids may be more at risk due to their surface-oriented behavior patterns in offshore and continental shelf areas. These species often occupy the upper water column where they may encounter military expended materials during their initial impact with the water's surface. While surface-dwelling species face the highest immediate risk due to their proximity to

impact points, the rapid deceleration of materials in water significantly reduces risks to mid-water column species. As military expended materials settle to the seafloor, they create potential strike risks for bottom-dwelling species. Green sturgeon typically inhabit benthic environments where they may encounter settling materials.

When military expended materials such as projectiles, bombs, missiles, and rockets impact the water's surface, they experience rapid deceleration due to water resistance. This dramatic reduction in velocity creates a distinct pattern of strike risk zones: highest risk at the surface impact point, decreasing risk through the water column, and minimal risk by the time materials reach the seafloor. The rapid velocity decrease greatly reduces the risk of a strike to most fish species, which are highly mobile and agile. In addition, most fish species possess natural avoidance capabilities that help further mitigate strike risks. Their lateral line systems and other sensory mechanisms allow them to detect approaching objects through water displacement and pressure changes. As expended materials lose momentum while traveling through the water column, fish have increasing opportunity to detect and avoid these objects. This natural avoidance capability becomes more effective as projectile velocity decreases, though surface-dwelling species have less reaction time during initial impact events.

Given the vast expanse of the action area, the relatively localized nature of training activities, and the anticipated low density of ESA-listed fish in areas where HCTT activities involving military expended materials will occur, the likelihood of strikes affecting ESA-listed species is considerably lower than for more abundant non-listed species. Even in scenarios involving intensive training activities, such as small-caliber projectile exercises within a confined gunnery box, the mathematical probability of striking an individual ESA-listed fish remains extremely low.

The combination of natural avoidance capabilities, rapid projectile deceleration in water, and the relatively low density of ESA-listed fish species in areas where HCTT activities occur, suggests that while strike risks exist, the likelihood of a military expended material strike is extremely low. Given the effort expended by individuals to avoid military expended materials as they sink in the water column will be minimal (i.e., a few meters distance) and temporary, behavioral avoidance of military expended materials sinking through the water column is insignificant (i.e., so minor that the effects cannot be meaningfully evaluated).

For these reasons, the effects on ESA-listed fish from physical disturbance and strike with military expended materials are insignificant, in the case of behavioral disturbance, and discountable, in the case of strike. Thus, we conclude that the effects of military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following fish species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho

salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Invertebrates

Military expended materials that sink to the seafloor also have the potential for physical disturbance and strike to ESA-listed abalone and ESA-proposed sunflower sea star. However, disturbance or strike impacts on these species are not very likely because military expended materials do not generally sink rapidly enough to cause strike injury at the seafloor. In addition, given the low density of these species in areas where military expended materials would be used, the likelihood that military expended materials would fall directly on an ESA-listed abalone or ESA-proposed sunflower sea stars is extremely low. The Action Proponents typically do not conduct training and testing activities that use military expended materials in the shallow water habitat preferred by abalone or sunflower sea stars. The large majority of HCTT military expended materials would be located away from the coastline on the continental shelf, slope, and abyssal plain in water depths not occupied by abalone or sea stars (Navy 2024d). Bombs, missiles, and rockets are designed to explode within about 3 ft (1 m) of the sea surface, where abalone and sunflower sea stars do not occur. Navy ordnance use is not scheduled in areas of white abalone, black abalone, or sea star occurrence (Navy 2024d). Although sinking exercises produce large amounts of military expended materials, these exercises occur at least 50 nm from shore and in water depths greater than 6,000 ft (1,829 m), where abalone and sunflower sea stars do not occur (Navy 2024d).

In summary, we find it extremely unlikely that strike or physical disturbance from military expended materials will result in adverse effects to ESA-listed abalone or ESA-proposed sunflower sea star. This determination is based on the anticipated low densities of ESA-listed (or proposed) invertebrates in the action area, the small overlap between these species' preferred depth ranges and where the majority of military expended materials would be used, and the slow speed that such expended materials would be traveling when they reach the seafloor. Thus, we conclude that the effects of military expended materials resulting from the proposed action may affect, but are not likely to adversely affect white abalone, black abalone, or sunflower sea star.

Amphibious Landing Events

Amphibious warfare training ranges from individual, crew, and small unit events to large task force exercises. Individual and crew training include amphibious vehicles and naval gunfire support training. Such training includes shore assaults, boat raids, airfield or port seizures, and reconnaissance. Large scale amphibious exercises involve ship-to-shore maneuver and naval fire support.

Physical disturbance from these activities results from large unit forces involving amphibious vessels moving from the marine environment to the ashore environment. The platforms used include Landing Craft Air Cushion vehicles, Amphibious Combat Vehicle, and Lighter Amphibious Resupply Cargo vehicles, assault amphibian vehicle units, small boats and small

unit live-fire and non-live-fire operations. Training includes the driving of vehicles into the water, basic in-water vehicle maneuvers, and the driving of vehicles back to shore.

This section addresses the potential effects from physical disturbance stressors resulting from amphibious warfare activities to ESA-listed sea turtles occurring in nearshore environments and to Hawaiian monk seals occurring either on land (i.e., hauled out) or nearshore. The acoustic stressors (above), ingestion stressors (below), and stressors from sediment disturbance (below) in shallow coastal environments associated with amphibious warfare activities are addressed elsewhere in this section.

Amphibious operations will occur in the nearshore environment, including within designated critical habitat for Hawaiian monk seals and proposed critical habitat for the Central North Pacific DPS of green sea turtles. Activities like vessels approaching or landing on the shoreline and beach could disturb animals in the area, potentially disrupting their normal foraging and resting behaviors. However, there are numerous alternative locations nearby for these animals to retreat to.

The physical disturbances and strikes to the critical habitat from amphibious warfare activities include vessels, in-water strikes, seafloor devices, and military expended materials. Amphibious landings in sensitive marine habitats will be carefully located to minimize the affected area. Approaches from the ocean are not expected to impact these habitats because of the natural surf and tidal energy, along with the shifting sands.

Seafloor devices would primarily be placed in soft-bottom habitats, and most would be positioned to have only minor impacts on the bottom substrate. Over time, military expended materials on the seafloor could be colonized by benthic organisms, as these materials would act as anchor points in the shifting sands.

The combined physical disturbance impacts from amphibious warfare activities (identified above) on ESA-listed species habitats will also be temporary in nature and will not result in permanent alteration or degradation of critical habitats for monk seals or green sea turtles. The combined frequency, duration, and intensity of these impacts would be minimal, temporary, and short term. As noted above, there are additional locations for ESA-listed species to find refuge in acceptable habitat in close proximity.

Mitigation measures have been put into place and are identified in Sections 2.4.2 and additional information specific to Hawaiian Monk Seals in Section 4.1.3. The Action Proponents have standard operating procedures outlined in Marine Corps Base Hawaii Integrated Natural Resources Management Plan (USMC 2017) in Appendix C2 and Appendix D5 and Section 7.4: Coastal and Marine Resources Management to limit impacts in action area in the area seaward of Marine Corps Training Area Bellows (MCTAB) from the seafloor to 10 meters above the seafloor from the mean lower low water mark to the 200 m depth contour line was designated critical habitat for HMS.

In summary, we find it extremely unlikely that strike or physical disturbance from amphibious warfare training and landing events will result in adverse effects to ESA-listed Hawaiian monk seals or green sea turtles. This determination is based on the anticipated low densities of ESA-listed species in the amphibious landing locations in the action area, the small overlap between these species' preferred haul outs and foraging ranges and where the majority of amphibious activities would be located, and the mitigation measures put in place and standard operating procedures that Marine Corps Base Hawaii has implemented via the INRMP process. Thus, we conclude that the effects of amphibious warfare and landing activities described in the proposed action may affect, but are not likely to adversely affect Hawaiian monk seals or Central North Pacific DPS green sea turtles.

Entanglement Stressors

Some expended materials from the proposed military readiness activities may pose a risk of entanglement to ESA-listed species in the action area. These interactions 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 military expended materials have the potential to result in non-lethal adverse effects, mortality, or result in no impact. Expended materials that may pose an entanglement risk include wires and cables, decelerators and parachutes, and nets used for obstacle avoidance training. Additional discussion of entanglement stressors is included in Section 2.5.5.

Marine Mammals

There has never been a reported or recorded instance of a marine mammal entangled in military expended materials. NOAA (2014a) conducted a review of entanglement of marine species in marine debris with an emphasis on species in the United States. The review did not document any known instances where military expended material had entangled a cetacean. Instead, the vast majority of entanglements have been from actively fished or derelict fishing gear. For example, Knowlton et al. (2012), as cited in NOAA (2014a), conducted a 30-year comprehensive review of entanglement rates of North Atlantic right whales using photographs. In the report, 626 individuals were observed and the vast majority showed evidence of entanglement involving non-mobile pot gear and nets used for fishing. Military expended material has not been shown to entangle ESA-listed large whales despite the Navy expending such materials in the action area (and other range complexes throughout the Pacific) for decades.

If encountered, it is unlikely that an animal would get entangled in a fiber optic cable, sonobuoy wires, or guidewire 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 sinking rates) (Navy 2024d) the likelihood of entanglement from cables and wires is extremely low. Additionally, as noted above, though there are numerous documented cases where marine mammals have been entangled in anthropogenic materials (e.g., fishing gear), there have been no documented instances where a marine mammal has been entangled in military expended cables and wires

despite decades of training and testing activities being conducted in the action area and elsewhere utilizing wires and cable. For these reasons, the likelihood that ESA-listed cetaceans become entangled in military expended wires and cables in the action area is so low as to be discountable.

Decelerators/parachutes also may pose a risk of entanglement, though for the reasons described below, we do not believe such incidences are likely to occur. The Action Proponents use a variety of sizes of decelerators/parachutes in the action area (see Section 2.5.5 for details). The risk of marine mammal entanglement in small and medium decelerators/parachutes, which make up the majority of parachutes, is extremely low because these items are weighted and remain in the water column for just a few minutes before sinking to the seafloor. It is extremely unlikely that an ESA-listed marine mammal would encounter decelerators/parachutes on the seafloor.

The main potential for entanglement is with the large and extra-large decelerators/parachutes. While the large parachutes would eventually sink and flatten, there is the potential that these decelerators/parachutes, which do not have weights attached, could remain suspended in the water column or billow at the seafloor for a longer period of time before flattening (Navy 2024d). The length of the decelerator/parachute lines poses an entanglement risk as well. However, a limited number of these items are proposed for use each year (i.e., 50 large and 5 extra large) and during some activities (aerial targets launched from shore), efforts are made to recover large and extra large parachutes. The small number of large and extra large decelerators/parachutes proposed for use annually and the fact that some of these are recovered reduces the potential for ESA-listed marine mammals to encounter and become entangled in these items.

There are a number of key differences between parachutes/decelerators and fishing gear that result in the likelihood of entanglement in parachutes/decelerators being significantly lower than it is for fishing gear. First, as noted above, most decelerators/parachutes proposed for use will sink quickly to seafloor and will not remain suspended in the water column for extended periods of time. This is in contrast to fishing gear which can be left in the water for days or weeks at a time. Additionally decelerators/parachutes would be highly visible in the water column likely alerting a nearby animal to the presence of the obstacle (Figure 10), whereas fishing gear may consist of some buoys and traps that are visible, but also many hundreds of feet of rope in between these items that is not.

It is also possible that a bottom feeding animal (e.g., sperm whale) could become entangled when foraging in areas where parachutes have settled on 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, a relatively rare ESA-listed marine mammal encountering this billowing parachute, and the animal not detecting and avoiding the entanglement risk is so unlikely as to be considered discountable.



Figure 10. Aerial Target with Deployed Parachute (Navy 2024d)

The Navy has proposed to use submerged nets for subsurface obstacle avoidance training and testing with extra-large UUVs. While the nets are in the water, there is the potential for entanglement of ESA-listed marine mammals. Marine mammals are susceptible to entanglement in both derelict and operational fishing gear. Entanglement of marine mammals in fishing gear (e.g., nets, ropes, lines, and buoys) can cause chronic and acute injuries, which can lead to reduced fitness and mortality. Although entanglement in fishing nets represents a documented risk to many marine mammal populations worldwide, the Navy's proposed deployment of nets differs from commercial fishing operations in ways that significantly reduce this risk. Whereas, stationary (or "passive") commercial fishing nets are often left unattended for long periods when actively fishing, the Navy will monitor their submerged nets at all times. Because submerged nets will be continuously monitored and tethered to a support vessel(s), we anticipate the Navy will respond quickly if a marine mammal is approaching a net and/or at risk of entanglement. Navy nets will be submerged for a maximum of four hours at a time, and will be removed from the water at the end of each exercise. By comparison, fixed-net commercial fishing gear is typically left submerged for much longer periods, and is often reset in the same location after being checked. In general, we expect most marine mammals will avoid the immediate areas where the Navy will deploy submerged nets due to the acoustic and visual disturbance that the proposed obstacle avoidance exercises will create. The Navy proposed to monitor a 500-yd

mitigation zone around the submerged nets for the presence of marine mammals (see Table 7 for details regarding mitigation requirements). A Navy Lookout located on the support vessel will monitor the mitigation zone. If a marine mammal is within 500 yards of the submerged net, the support vessel will be notified to implement the activity-based mitigation requirements, as described in Table 8. The Navy also agreed to conduct obstacle avoidance testing activities involving nets only during daylight hours (Navy 2024c), which will further reduce the risk of marine mammal entanglement. Although submerged nets represent a potential risk of entanglement to marine mammals, we believe this is extremely unlikely to occur as a result of the proposed action given the 1) proposed mitigation and monitoring measures (discussed above), 2) relatively small amount of time nets would be in the water, and 3) the anticipated low densities of marine mammals in the action area in general, and in the vicinity of Navy obstacle avoidance exercises, in particular.

In summary, for the reasons described above, we consider it extremely unlikely for any marine mammals to become entangled in military expended materials including wires and cables, decelerators/parachutes, and nets used for obstacle avoidance. Therefore, potential effects on ESA-listed marine mammals from entanglement in military expended materials are discountable. We conclude that the effects of entanglement in military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

All of the ESA-listed sea turtles present within the action area could encounter materials that have the potential to entangle them such as wires and cables, decelerators and parachutes, and obstacle avoidance training nets that are used during HCTT activities. Sea turtles could encounter these items at the water's surface, in the water column, or along the seafloor. Many factors may influence the degree of entanglement risk for sea turtles such as 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 fishing gear (as opposed to military items) that float or are suspended at the ocean's surface for long periods of time.

As discussed above for marine mammals, the likelihood of sea turtle entanglement in cables and wires as a result of the proposed action is extremely low given the properties of these expended materials (i.e., low breaking strength and relatively high sinking rates). There have been no documented instances where a sea turtle has been entangled in military expended cables and wires despite decades of training and testing activities being conducted in the action area and elsewhere utilizing wires and cable. Given the low concentration of expended wires and cables, the rapid sink rates, the tensile strength and breakability of the material, and likely distribution of

sea turtles in the action area that may co-occur with areas where cables and wires are expended, we consider it extremely unlikely for any sea turtles to be exposed to entanglement in cables and wires as part of the proposed action. For these reasons, the potential impacts from these stressors on sea turtles are discountable.

ESA-listed sea turtles may overlap spatially with expended decelerators and parachutes from training activities throughout the action area. While in the water column, a sea turtle is not likely to become entangled in these materials because the parachute would have to land directly on the turtle, or the turtle would have to swim into the parachute and its lines, before it sank. Should this unlikely event occur, a sea turtle could likely free itself and swim away from the sinking object. Sea turtles at or near the water's surface are more likely to encounter and become entangled by floating decelerators, parachutes and cords than those animals located deeper in the water column or foraging along the seafloor. Small and medium decelerators and parachutes are not expected to remain at the water's surface for long, and would begin to sink within five to 15 seconds. While a decelerator and parachute is sinking, it could be carried along within a current, or become snagged on a hard structure near the substrate.

Once these smaller chutes reach the substrate, they will likely flatten and not "billow" from the bottom. If the parachute and associated lines sink to the seafloor in an area that is calm along the substrate, it would likely settle there and remain undisturbed. In an area with bottom currents or active tidal influence, the parachute may move along the seafloor, away from the location in which it was expended. But over time, it may become covered by sediment in most areas or colonized by attaching and encrusting organisms, which would further stabilize the material and reduce the potential for reintroduction as an entanglement risk. Any decelerators or parachutes that do settle have some small degree of risk to become resuspended, however it is more likely that these items would become buried in sediments as ocean currents move sediment around along the seafloor, or organisms colonize them.

The sink rates of small and medium decelerator and parachute assemblies would rule out the possibility of these drifting great distances into nearshore and coastal areas where benthic foraging species of sea turtles are more likely to occur. Given the sparse distribution of the small and medium decelerators and parachutes deployed and expended throughout the action area, as well as the patchy distribution and general behavior of sea turtles, the likelihood of a sea turtle entanglement in a small or medium sized decelerator or parachute is extremely low (i.e., discountable).

The likelihood for entanglement is higher for the large and extra-large decelerators and parachutes due to their size and length of the attachment cords, and because some of the large and extra-large decelerators and parachutes have the potential to be expended or drift into nearshore habitats, which have a higher concentration of sea turtles. Additionally, these sized parachutes and decelerators are not weighted with anything to help them sink rapidly, thus could potentially remain suspended in the water column for an extended period of time, increasing the chance of sea turtles encountering them in the water column. However, significantly fewer large

and extra-large decelerators and parachutes are expended annually during HCTT activities, and some of these would be recovered. The chance of a sea turtle encountering one of these expended materials is extremely low given sea turtle distributions and densities throughout the areas where large and extra-large decelerators and parachutes would be used. In addition, there are no known instances of sea turtle entanglement with a Navy or Coast Guard decelerator or parachute assembly in the action area despite decades of military readiness activities using these items. Therefore, the likelihood of a sea turtle entanglement in large or extra-large decelerators or parachutes is extremely low (i.e., discountable).

Similar to marine mammals, there is the potential for entanglement of ESA-listed sea turtles in nets used by the Navy for obstacle avoidance training. Although entanglement in fishing nets represents a documented risk to many sea turtle populations worldwide, as discussed above, the Navy's proposed deployment of nets differs from commercial fishing operations in ways that significantly reduce this risk. Whereas, stationary (or "passive") commercial fishing nets are often left unattended for long periods when actively fishing, the Navy will monitor their submerged nets at all times. Because submerged nets will be continuously monitored and tethered to a support vessel(s), we anticipate the Navy will respond quickly if a sea turtle is approaching a net and/or at risk of entanglement. Navy nets will be submerged for a maximum of four hours at a time, and will be removed from the water at the end of each exercise. By comparison, fixed-net commercial fishing gear is typically left submerged for much longer periods, and is often reset in the same location after being checked. Although submerged nets represent a potential risk of entanglement to sea turtles, we believe this is extremely unlikely to occur as a result of the proposed action given the 1) proposed mitigation and monitoring measures (discussed above and in Table 8), 2) relatively small amount of time nets would be in the water, and 3) the anticipated low densities of sea turtles in the action area in general, and in the vicinity of Navy obstacle avoidance exercises, in particular.

While these mitigation measures likely reduce the risk of a sea turtle encounter with the Navy's nets, given their smaller size (as compared to marine mammals) and time spent underwater, although unlikely, it remains possible that a sea turtle could swim into the net without being sighted by the Lookout. However, even if a sea turtle were to swim into the net, we would not expect it to become entangled given the small mesh size (i.e., one inch) of the Navy nets proposed for this activity. Studies of bycatch in commercial gillnet fisheries have shown that sea turtles are most susceptible to entanglement in larger mesh size nets (Price and Van Salisbury 2007; Gilman et al. 2010). For example, Putman et al. (2020)found a significantly lower amount of turtle bycatch in small-mesh (<10 centimeters) compared with large-mesh gillnets (>10 centimeters) in North Carolina. Gillnet fisheries that target sea turtles use a mesh size of between 20 and 60 centimeters (7.9 and 23.6 inches), presumably based on experience that these mesh sizes maximize turtle catch rates (Gilman et al. 2010). In a study of turtle bycatch in mid-Atlantic gillnet fisheries, Murray (2009) reported that 3.6 inches was the smallest mesh size that sea turtles were entangled in, and most entanglements occurred with much larger mesh sizes. In addition, fisheries regulations that specify a maximum mesh size to minimize turtle capture have

focused on mesh sizes that are much larger than the one inch nets proposed by the Navy (Putman et al. 2020). For example, in 2006 NMFS promulgated a rule prohibiting the use of gillnets with stretched mesh seven inches or larger in the Atlantic Exclusive Economic Zone during particular times of year and locations where sea turtles would be most vulnerable to entanglement. If a sea turtle were to swim into the Navy's submerged net, we expect the animal would recognize it as an obstruction and quickly change course to avoid the net. Any additional energy expenditure or behavioral response associated with avoiding the net would likely be extremely minor (i.e., insignificant), with no measurable impact on the turtle's ability to feed, migrate or reproduce.

In summary, for the reasons described above, we consider it extremely unlikely for a sea turtle to become entangled in military expended materials including wires and cables, decelerators/parachutes, and nets used for obstacle avoidance training. Any additional energy expenditure or behavioral response associated with avoiding military expended materials would be insignificant. Therefore, potential effects on ESA-listed sea turtles from entanglement in military expended materials are discountable, and the potential effects of behavioral responses to avoid entanglement are insignificant. Thus, we conclude that the effects of entanglement in military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

Some fish species are more susceptible than others to entanglement in derelict fishing gear and other marine debris. For example, the shape of the body of some elasmobranchs such as manta rays, increase their risk of entanglement compared to other, more streamlined fish. For many pelagic species, including salmonids, eulachon, oceanic whitetip sharks, and scalloped hammerhead sharks, the risk of entanglement is relatively low given their body shape and ability to avoid materials that could entangle them in the water column.

For certain testing activities, long-length (up to approximately 60 mi.) fiber optic cables would be deployed in the water column in the California portion of the action area (Navy 2024d). These long-length fiber optic cables are slightly negatively buoyant, have a breaking strength of approximately 100 lb. or less, and are designed to resist coiling or the forming of loops when unspooled (Navy 2024d). The fiber optic cable would be temporarily deployed in the water column between moored points at depths of approximately 600 to 850 ft (Navy 2024d), well below the depths that most ESA-listed fish would be found. Fiber optic cables do not easily form loops, are brittle, and break easily if bent, so they pose a negligible entanglement risk. Additionally, the encounter rate and probability of impact from guidance wires and fiber optic cables are low, as few are expended (Navy 2024d). Fiber optic cables that are not recovered are 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

(Navy 2024d). 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. However, because of 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. The likelihood of ESA-listed fish species being entangled in wires and cables expended during military readiness activities is further reduced because, in most instances, we would expect these species would see the wire or cable in the water column and be able to swim around it.

Sonobuoys consist of a surface antenna and float unit and a subsurface hydrophone assembly unit attached through a thin gauge, dual-conductor, and hard-draw copper strand wire (Navy 2024d). The tensile breaking strength of the wire and rubber tubing is no more than 40 lb. (Navy 2024d). 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 ft and is dependent on the water depth and type of sonobuoy (Navy 2024d). 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 (Navy 2024d). Sonobuoys may remain suspended in the water column for no more than 30 hours, after which they sink to the seafloor (Navy 2024d). Sonobuoy wires may be expended within any of the range complexes throughout the action area. However, 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 (Navy 2024d). The likelihood of an ESA-listed fish species encountering a sonobuoy wire is very low since most sonobuoys are expended in offshore areas where densities of these species are very low. Therefore, the effects of entanglement in wires or cable on ESA-listed fish in the action area are considered discountable (i.e., extremely unlikely to occur). The effects of behavioral disturbance or energy expenditure necessary to avoid entanglement in wires and cables on ESA-listed fish are insignificant.

There is the potential for entanglement of ESA-listed fish in nets used by the Navy for obstacle avoidance training. Many commercial fishing operations worldwide use gillnets and entangling nets that are specifically designed to entangle fish as a form of capture. Commercial gillnets use a variety of mesh sizes, depending on the target species as mesh size typically has a profound effect on capture probability of fish (Northridge et al. 2017). Some fish species are more susceptible to entanglement in fishing gear compared to other fish groups. For example, the shape of the body of some elasmobranchs such as manta rays, increase their risk of entanglement compared to other, more streamlined fish. For many pelagic species, including oceanic whitetip sharks, scalloped hammerhead sharks, and steelhead, the risk of entanglement is unlikely given their body shape and ability to avoid materials that could entangle them in the water column. The ESA-listed species in the action area are all of sufficiently large body size that they would not be susceptible to entanglement of their gills in the one-inch mesh size nets proposed for use by the Navy. While early life stages of steelhead or salmon could become entangled in a one-inch mesh

net, these smaller steelhead are not expected to overlap spatially with the areas proposed for Navy activities involving submerged nets. In the unlikely event that a shark or salmonid were to swim into the Navy's submerged net, we expect the animal would recognize it as an obstruction and quickly change course to avoid the net. Any additional energy expenditure or behavioral response associated with avoiding the net would likely be extremely minor (i.e., insignificant), with no measureable impact on the animal's ability to feed, migrate or reproduce.

Giant manta rays may be more at risk (compared to sharks or steelhead) of becoming entangled in a net given their body shape and swimming motion. However, given the anticipated low density of this species in the offshore areas they typically occupy, we do not expect giant manta rays would overlap with submerged nets deployed as part of the proposed action. Occurrence of the giant manta ray in the action area is not well documented, but they are believed to be extremely rare. Southern California is the northern edge of giant manta ray distribution (Miller and Klimovich 2017), further reducing the probability of exposures in the action area. Although manta rays have been reported as bycatch in U.S. Pacific Ocean fisheries, these reports are all from purse seine or longline fisheries operating in the Central and Western Pacific. While bycatch of the smaller bat ray, which is not ESA-listed, is a recognized problem in California's set gill net fisheries, we could not find any reports of giant manta ray bycatch in commercial fisheries off California. This is likely due to the much lower abundance of giant manta rays as compared to bat rays in the action area.

Therefore, the effects of entanglement in submerged nets on ESA-listed fish in the action area are considered discountable (i.e., extremely unlikely to occur). The effects of behavioral disturbance or energy expenditure necessary to avoid the submerged net on ESA-listed fish are insignificant.

Parachutes and decelerators could potentially be encountered by ESA-listed fish at the sea surface, in the water column, or on the seafloor. Throughout the action area, the vast majority of expended decelerator and parachutes are small (18 inches) cruciform shaped decelerators used with sonobuoys (Navy 2024d). 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 (Navy 2024d). Entanglement of an animal in a parachute assembly at the surface or within the water column would be unlikely, since the parachute would have to land directly on an animal, or an animal would have to swim into it before it sinks. While the decelerator/parachute is sinking, fish would have ample opportunity to swim away from the large moving object. Even if the decelerator/parachute landed directly on a fish, it would likely be able to swim away faster than the decelerator/parachute would sink because the resistance of the water would slow the decelerator/parachute's downward motion.

Compared to other species, the chance of an entanglement is likely greater for giant manta rays, which are known to be susceptible to entanglement in fishing gear. A study in Hawaii found 10% of manta rays (28 individuals out of a sample of 290) had cephalic fins (fins on either side of the mouth) amputated, disfigured, or were non-functioning (Deakos et al. 2011), apparently due to

entanglement in monofilament fishing line. Other evidence has documented mortality of manta rays from entanglement with anchor and mooring lines (Deakos et al. 2011). Manta ray susceptibility to entanglement is largely due to their unique body shape, particularly their cephalic fins. However, manta rays are highly mobile species that are expected to be able to avoid the small or medium-sized floating or suspended decelerators and parachutes, which comprise the majority of the decelerators and parachutes used in the action area. Furthermore, these small and medium decelerators and parachutes have weights attached, causing a more rapid sink rate, thereby decreasing the amount of time materials float at the surface, reducing the risk of a giant manta ray encountering them.

Due to the length and number of attached lines (large chutes have 28 cords, approximately 40 to 70 ft long; extra-large parachutes have 64 cords, up to 82 ft long) (Navy 2024d), large and extra-large decelerators and parachutes may pose a higher degree of entanglement risk for fish, particularly manta rays. Additionally, large and extra-large parachutes are not weighted with anything to help them sink rapidly, and could potentially remain suspended in the water column for an extended period of time (Navy 2024d). However, the chance of an encounter is remote given the small number of the large or extra-large parachutes proposed to be deployed and the anticipated low abundance of ESA-listed fish species in the action area. Given the vast area over which any one of these large or extra-large decelerators and parachutes would be deployed and the limited number of them deployed annually, the chances of a giant manta ray or other ESA-listed fish species 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 (Navy 2024d). This standard operating procedure could further reduce the potential entanglement of ESA-listed fish in decelerators/parachutes.

Additionally, available data indicates the entanglements and injuries described for giant manta rays are mostly due to exposure to fishing gear such as monofilament lines and large heavy mooring lines. The materials of parachutes and decelerators and lines are not the same, and are considered lighter and more likely to sink over some period of time and ultimately settle on the seafloor. Monofilament lines are hard to see for fish and can float indefinitely in the water column unless they become attached to something that anchors them or causes them to sink. They also can easily form multiple loops. Mooring lines are quite heavy and likely more difficult for animal to release itself from should it become ensnared in a mooring line. Furthermore, no cases of entanglement of any fish species have been reported for parachutes or decelerators (U.S. Department of the Navy 2001; Ocean Conservancy 2010). While we recognize there is a higher risk of entanglement for giant manta rays than for other fish species, giant manta rays are likely able to visually detect and avoid descending or sinking parachutes in the water column. This is expected to result in a minor behavioral response. In summary, due to the factors discussed above, we find it extremely unlikely (i.e., discountable) that ESA-listed fish will become entangled in parachutes or decelerators as a result of the proposed action, and the effects of any

behavioral response by ESA-listed fish to avoid parachutes or decelerators in the water column will be insignificant.

In summary, for the reasons described above, we consider it extremely unlikely for an ESA-listed fish to become entangled in military expended materials including wires and cables, decelerators/parachutes, and nets used for obstacle avoidance training. Any additional energy expenditure or behavioral response associated with avoiding military expended materials would be insignificant. Therefore, potential effects on ESA-listed fish from entanglement in military expended materials are discountable, and the potential effects of behavioral responses to avoid entanglement are insignificant. Thus, we conclude that the effects of entanglement in military expended materials resulting from the proposed action may affect, but are not likely to adversely affect the following species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Invertebrates

It is extremely unlikely that an abalone or sunflower sea star would become entangled in wires or cables expended during HCTT activities. These expended materials would be essentially linear and untethered after deployment, as they sink through the water column. Once the items reach the bottom, they could be moved into different shapes or could loop around objects due to water currents, but the items are not expected to form tight coils and the possibility of an invertebrate being ensnared is remote. The wires and cables would eventually become buried in sediment or encrusted by marine growth, which would eliminate or further reduce the entanglement potential. In addition, the vast majority of wires and cables used for HCTT would be expended offshore in deep water areas where abalone and sunflower sea stars are not known to occur. The likelihood of an ESA-listed abalone or sunflower sea star encountering wires or cables expended during HCTT activities in extremely low given the small number of wires and cables that would be expended in nearshore areas and the low densities of ESA-listed abalone or sunflower sea stars throughout the action area.

Nets used for UUV obstacle avoidance testing will be tethered to support vessels and constantly monitored during use. Although nets may make contact with the seafloor, they would be held vertically in the water column by the support vessels, making entanglement or smothering of benthic marine invertebrates extremely unlikely. In addition, obstacle avoidance testing with nets will primarily occur in offshore portions of the action area that are outside of abalone and sunflower sea star habitat. Since all nets used for HCTT will be retrieved after the activity is over, there is no possibility of nets sinking to the seafloor where they can entangle or smoother ESA-listed invertebrates.

Invertebrates such as abalone and sunflower sea stars could be susceptible to entanglement in decelerators and parachutes, with the principal mechanisms of impact being burial, smothering, or abrasion. However, the large majority of decelerators and parachutes will be deployed in offshore areas (i.e., typically greater than 3 nm from the shore and in water depths greater than 656 ft) where these species are not known to occur (Navy 2024d). Given the proposed deployment locations and the anticipated low densities of ESA-listed abalone and sunflower sea stars in area where decelerators and parachutes will be deployed, it is extremely unlikely that a parachute or decelerator could be transported the distance from where it deployed to settle directly on any of these species. The large majority of decelerators and parachutes are weighted and will likely sink to the seafloor before they can drift into nearshore habitats occupied by abalone or sunflower sea stars. Large and extra-large parachutes are not weighted with anything to help them sink rapidly, and could potentially remain suspended in the water column for an extended period of time (Navy 2024d). However, the chance of an encounter is extremely unlikely given the small number of the large or extra-large parachutes proposed to be deployed, the long distances these expended materials would typically need to travel to reach abalone or sunflower sea star habitats, the low density of these species throughout the action area, and that the Navy will attempt to recover large and extra-large decelerators and parachutes, to the maximum extent practicable consistent with personnel and equipment safety (Navy 2024d).

In summary, for the reasons described above, we consider it extremely unlikely for an ESA-listed (or proposed) invertebrate to become entangled in military expended materials including wires and cables, decelerators/parachutes, and nets used for obstacle avoidance training. Therefore, potential effects on ESA-listed invertebrates from entanglement in military expended materials are discountable. Thus, we conclude that the effects of entanglement in military expended materials resulting from the proposed action may affect, but are not likely to adversely affect white abalone, black abalone, or sunflower sea stars.

Ingestion Stressors

This section analyzes the potential impacts of ingestion stressors resulting from military readiness activities in the action area. Military readiness activities throughout the action area result in expended materials that could potentially be ingested. These expended materials are typically widely distributed and low in density, given the size of the action area and number of locations where such activities occur. This analysis includes the potential impacts from ingestion of the following types of military expended materials: 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 decelerators/parachutes. The majority of material expended during training and testing activities would likely become buried in sediments or penetrate the seafloor and not be accessible to most marine species that feed in the water column. Species that feed on benthic prey (e.g., green sturgeon) could still be susceptible to ingestion of military expended materials after they sink to the seafloor. Additional discussion of ingestion stressors is included in Section 2.5.6.

Marine Mammals

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 cetaceans 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. However, the relatively low density of both sperm whales and expended materials along the vast sea floor suggests ingestion of such items would be extremely unlikely. 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 cetaceans may also encounter military expended material that remains suspended in the water column for extended periods of time. Because 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. (2011) 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 cetacean that happened to encounter and accidentally swallow the item. 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 (Force 1997; Hullar et al. 1999; Arfsten et al. 2002). 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 military readiness activities, where they may persist for long periods, and therefore could be ingested by marine mammals 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.

In conclusion, because 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 to 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. Therefore, we conclude the ingestion of expended materials from activities associated with the proposed action may affect, but is not likely to adversely affect the following species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; MHI IFKW; Guadalupe fur seal; Hawaiian monk seal; North Pacific right whale; and Southern Resident DPS killer whale.

Sea Turtles

Sea turtles commonly mistake debris for prey. Ingestion of expended materials by sea turtles could occur in all large marine ecosystems and open ocean areas and can occur at the surface, in the water column, or at the seafloor, depending on the size and buoyancy of the expended object and the feeding behavior of the turtle (Navy 2024d). Susceptibility of sea turtles to ingestion risk is a factor of the life-stage of the individual sea turtle, foraging habits of the species, the location of the item within the water column, and the type of debris. For example, floating material could be eaten by turtles such as leatherbacks, juveniles, and hatchlings of all species that feed at or near the water surface, while materials that sink to the seafloor pose a risk to bottom-feeding turtles such as loggerheads.

Types of munitions that can result in fragments include demolition charges, projectiles, missiles, and bombs. The size of these fragments would vary depending on the net explosive weight, size, and munitions type. The munitions and other materials small enough to be ingested by ESA-listed sea turtles are small and medium caliber projectiles (up to 2.25 in), 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. Other munitions and munitions fragments such as large-caliber projectiles or intact training and testing bombs are too large for sea turtles to consume and are made of metal a sea turtle would not be able to break apart and ingest.

Fragments that could be encountered by sea turtles would most likely be those that have settled on the seafloor. Most expendable materials will be used over deep water and these items will sink quickly and settle on the seafloor, with the exception of chaff and some firing target materials. Ingestion of a decelerator/parachute by a sea turtle at the surface or within the water column would be unlikely, since the decelerator/parachute would not be available for very long before it sinks as these items are weighted (Navy 2024d).

While the smaller items discussed here may pose a hazard to sea turtles, as discussed for non-explosive practice munitions ingestion, the impacts of ingesting these forms of expended materials on sea turtles would be minor because of the following factors (Navy 2024d):

- The limited geographic area where materials other than munitions are expended during a given event.
- The limited period of time these MEM would remain in the water column.
- The unlikely chance that a sea turtle might encounter and swallow these items on the seafloor, particularly given that many of these items would be expended over deep, offshore waters.

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 material. If this occurs, chaff is not expected to impact sea turtles due to the low concentration that would be ingested and the small size of the fibers. Chaff is composed of fine fibers of silicon dioxide coated with aluminum alloy. Due to their light weight and small size they float and 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. Given the small size, low densities, and low toxicity of chaff, any accidental ingestion by ESA-listed sea turtles 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. Firing target materials are normally retrieved before sinking so it is not reasonable to expect ingestion of these items to occur.

Chaff cartridge plastic end caps and pistons would also be released into the marine environment, during Navy activities, where they may persist for long periods and therefore could be ingested by sea turtles while initially floating on the surface and sinking through the water column. However, these end caps would eventually sink to the seafloor where they would be less likely to be ingested by some life stages or species of sea turtles, such as hatchlings and pre-recruitment juveniles of all sea turtle species and all life stages age of leatherback sea turtles. As mentioned above, green, hawksbill, Kemp's ridley, and loggerhead sea turtles could have a higher potential to ingest these items if they settle in potential benthic feeding habitat.

Should a sea turtle encounter military expended materials, it is unlikely that it would ingest every fragment it encounters. Sea turtles may attempt to ingest a projectile and then reject it after realizing it is not a food item. Additionally, ingestion of items would not necessarily result in

injury or mortality to the individual if the item does not become embedded in tissue (e.g., Wells et al. 2008). It is likely that most ingested material would pass through the digestive tract of the animal. We are unaware of any data indicating military items have been found in sea turtles that have been necropsied, unlike plastics that appear similar to jellyfish or other prey to a sea turtle (Schuyler et al. 2016). Plastic bags are the most commonly ingested type of debris amongst sea turtles (NOAA 2014b). If material is ingested, ingestible-sized items would likely be spit out or passed through the digestive tract without significantly impacting the individual. In addition, given the limited geographic area where materials other than munitions are expended during a given event, and the short duration of time these military expended materials would remain in the water column, the probability of a sea turtle encountering these materials is low.

In summary, ingestion of military expended material of sufficient size to cause on adverse effect by ESA-listed sea turtles is extremely unlikely and thus make the effect of this stressor discountable. The effects of any accidental ingestion of military expended materials that are small enough to be ingested by ESA-listed sea turtles would be so minor as to be insignificant for the reasons discussed above. Thus, we conclude that ingestion of military expended materials resulting from the proposed action may affect, but is not likely to adversely affect the following species: green sea turtle – Central North Pacific DPS; green sea turtle – East Pacific DPS; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; and loggerhead sea turtle – North Pacific DPS.

Fish

As previously described, the Action Proponents expend 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). Only small- and medium-caliber munitions (or small fragments from larger munitions), chaff, and end caps and pistons from flares and chaff cartridges are considered to be of ingestible size for ESA-listed fish in the action area. Small- and medium-caliber projectiles include all sizes up to and including 2.25-in diameter. These solid metal materials would quickly move through the water column and settle to the seafloor. Ingestion of non-explosive practice munitions in the water column is possible when shiny fragments of the munitions sink quickly and could be ingested by fast, mobile predators that chase moving prey. Small-caliber projectiles would likely be more prevalent throughout the action area and thus more likely to be encountered and potentially ingested by ESA-listed fishes. For many small fish species and juvenile fishes, many of these items (with the exception of chaff) are too large to be ingested. If a larval or juvenile fish swallows chaff, studies have shown it to have limited effects on fishes due to the concentration levels at which it is released (Force 1997; Spargo 1999; Arfsten et al. 2002). No ingestion potential impacts on early life stages of fishes are likely to occur, with the exception of large juveniles that may be large enough to ingest military expended materials. Therefore, the

discussion in this section focuses on those ESA-listed fish species large enough to potentially ingest these materials.

Ingestion of military expended materials by fishes could occur in portions of the action area, and can occur at or just below the surface, in the water column, or at the seafloor, depending on the size and buoyancy of the expended object, and the feeding behavior of the fish. It is reasonable to assume that any item of a size that can be swallowed by a fish could be eaten at some time. Given the life histories and foraging strategies of the fish species considered in this opinion, ingestion of materials could occur at the water surface, in the water column, or on the seafloor. The potential for ESA-listed fish species to encounter and ingest expended materials is also evaluated with respect to their physical size and geographic range, which could also influence the probability that they would consume military expended materials.

Fish are known to ingest a variety of small items in the marine environment, including metal and plastics (Hoss and Settle 1990; Jackson et al. 2000). Metal items eaten by marine fish are generally small (such as fish hooks, bottle caps, and metal springs), suggesting that small and medium caliber projectiles, pistons, or end caps (from chaff canisters or flares) are more likely to be ingested. Many fishes are also known to accidentally ingest plastic materials and the extent to which an individual fish might discriminate between a plastic item perceived as prey and an indistinct or less appealing shape is not clear. Both physical and toxicological impacts could occur as a result of consuming metal or plastic materials (Davison and Asch 2011; Possatto et al. 2011; Dantas et al. 2012) (Danner et al. 2009). Plastics in particular have been shown to increase hazardous toxic burden in fish leading to organ (e.g., liver) toxicity (Rochman et al. 2013). Of these military expended materials that could potentially 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).

The potential for ESA-listed fish species to encounter and ingest expended materials is evaluated with respect to their feeding group, size, and geographic range, which influence the probability that they would eat military expended materials composed of a variety of materials in various sizes. Open-ocean predators (e.g., salmon, steelhead, and oceanic whitetip sharks) and open-ocean planktivores (e.g., giant manta ray and eulachon) may ingest materials in the water column. Open-ocean species demonstrate distinct patterns of potential exposure to military expended materials. Salmon, steelhead, eulachon, oceanic whitetip sharks, and giant manta rays, which inhabit the pelagic zone, may encounter floating munitions in the water column. The rapid sinking nature of military expended materials substantially reduces the window of opportunity for pelagic species to encounter and ingest these items. Recent studies indicate that when pelagic fish do encounter such materials, they typically demonstrate discriminatory feeding behavior, often rejecting non-food items after initial contact. While the most likely scenario would be for steelhead to ignore these objects, if a salmon or steelhead did ingest a fragment or other munition, it would most likely taste the item, then spit it out (Felix et al. 1995). Oceanic whitetip sharks are considered scarce in the action area, which would decrease their chance of

encountering sinking material in the water column. Once the item sinks to the seafloor, it would be unavailable to oceanic whitetip sharks. As with the other pelagic species, if an item were accidentally ingested by a shark, it would likely expel the item after it was determined to not be a prey item. Giant manta rays, being filter-feeders, are not expected to deliberately ingest munitions, although accidental ingestion is possible. Eulachon, which primarily feed on plankton, small crustaceans (e.g., copepods) and krill, are also not expected to confuse munitions or other expended materials for prey.

Coastal and benthic-dwelling species (e.g., green sturgeon and hammerhead sharks) face different exposure scenarios due to their habitat preferences and feeding strategies. Coastal and estuarine bottom-dwelling predators may ingest materials from the seafloor. Ingesting nonfood items is common among a variety of marine fishes, particularly those that feed on the seafloor (Hoss and Settle 1990; Jackson et al. 2000; Boerger et al. 2010). Species such as sturgeon and scalloped hammerhead sharks may interact primarily with expended materials that settle on the seafloor. Recent research demonstrates that bottom-dwelling species exhibit varying degrees of risk based on their specific habitat preferences, feeding behaviors, and other biological characteristics. For example, scalloped hammerhead sharks are equipped with highly developed electroreception organs that may improve their ability to distinguish between live prey and expended military munitions or other materials.

The diet of green sturgeon is comprised of some prey items that have hard body parts (e.g. shells and carapaces from mollusks and shrimp) so there is a potential for them to mistakenly ingest some of the expended munitions. Therefore, munitions and fragments of munitions on the bottom or within the substrate could possibly be mistaken for a food item or could be incidentally taken along with other food items. An encounter with a casing would not necessarily lead to ingestion or swallowing of the item, as a fish might "taste" the item and then expel it (Felix et al. 1995). However, because they normally are able to pass small, hard bodied prey, a small caliber casing is unlikely to cause a blockage or other digestive issues as these items are relatively small and smooth and would likely pass through a sturgeon's digestive tract without causing harm. In addition, Lari et al. (2013) demonstrates that while sturgeon might encounter settled materials during feeding, their ability to process hard-bodied prey items suggests minimal risk from ingesting smooth-surfaced materials such as small-caliber projectiles. In addition, green sturgeon exposure to military expended materials is likely very limited since this species' range only overlaps with a small portion of the action area in the Northern California Range Complex, where Action Proponent activities involving military expended materials will be limited.

In summary, 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. The combination of rapid sinking rates of materials, species' discriminatory feeding behaviors, and limited spatial overlap between ESA-listed fish and expended materials suggests that the likelihood of significant negative impacts is extremely low. Therefore, the effects of ingestion of expended materials resulting from the proposed action

may affect, but are not likely to adversely affect, the following ESA-listed fish species: giant manta ray; oceanic whitetip shark; scalloped hammerhead shark – Eastern Pacific DPS; green sturgeon – Southern DPS; eulachon – Southern DPS; steelhead (Southern California DPS, California Central Valley DPS, Central California Coast DPS, Northern California DPS, South-Central California Coast DPS); Chinook salmon (California Coastal ESU; Central Valley Spring-Run ESU; Sacramento River Winter-Run ESU); and coho salmon (Central California Coast ESU, Oregon Coast ESU, Southern Oregon and Northern California Coasts ESU).

Stressors Resulting in Effects to Habitat or Prey of ESA-listed Species

This section analyzes potential impacts to ESA-listed marine mammals, sea turtles, fish, and invertebrates exposed to stressors indirectly through impacts to their habitat or prey. The stressors evaluated in this section include: explosives; explosive byproducts and unexploded munitions; metals; and chemicals. These secondary (or indirect) stressors are analyzed collectively for all taxa because the potential effects to prey and habitat are the same across taxa.

Explosives

Underwater explosions could impact other species in the food web, including prey species that ESA-listed species feed upon. The impacts of explosions would differ depending on the type of prey species in the area of the blast, type of munition used, depths of explosions and environmental conditions. 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, with no lasting effect on prey availability or the pelagic food web expected. Therefore, impacts to prey from explosives associated with the proposed action are insignificant. We conclude that impacts to ESA-listed species prey from HCTT explosive use may affect but are not likely to adversely affect ESA-listed species.

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 three to six feet from the degrading munitions. Based on these results, while it is possible that ESA-listed species 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.

Given the large size of the ocean areas where explosives could be used, the concentration of munitions, explosives, expended material, or devices in any one location within the action area are expected to relatively small. 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 ESA-listed species prey abundance in the action area. For this reason, the effects of explosive byproducts and unexploded munitions on ESA-listed species through impacts on prey and water quality are considered insignificant. Therefore, we conclude that impacts to prey and water quality from explosive byproducts and unexploded munitions associated with the proposed action may affect, but are not likely to adversely affect ESA-listed species.

Metals

Metals are introduced into seawater and sediments as a result of training 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 (Navy 2013c; Briggs et al. 2016; Edwards et al. 2016; Kelley et al. 2016; Koide et al. 2016) indicate metal contamination resulting from military readiness activities 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 because 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). Shaw et al. (2023) found that Kailua Bay green sea turtles are exposed to elevated concentrations of lead, very likely caused by lead shot from the historic Honolulu skeet shooting range. However, the estimated daily intake of lead was significantly less than the acute concentrations that cause harm in red-eared slider turtles (Shaw et al. 2023). Most turtles sampled in this study were healthy and did not appear outwardly affected by their elevated blood lead concentrations, although the authors note that the effects of chronic lead exposure on sea turtles is unknown (Shaw et al. 2023). 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 the effects this may have on the availability of ESA-listed species prey are unknown. The research cited above indicates that metals introduced into the action area are unlikely to have measureable impacts on ESA-listed species prey or habitat. Thus, the effects of metals introduced into seawater and sediments on ESA-listed species through impacts to their prey or habitat are insignificant. Therefore, we conclude that

impacts to prey and water quality from metals introduced into seawater and sediments as a result of the proposed action may affect, but are not likely to adversely affect ESA-listed species.

Chemicals

Several military readiness 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 species or their prey, particularly given the anticipated low densities of ESA-listed species in portions of the action area where such chemical may be released. 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 (EPA 2014). However, such concentrations would be localized and are not likely to persist in the ocean. Research has demonstrated that perchlorate did not bio concentrate 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 species prey or habitat. Chemicals may also be introduced into the marine environment through oils, lubricants, hydraulic fluids and other vessel and equipment emissions. Since such releases would be in small quantities and dispersed throughout the large action area, we do not anticipate these pollutants would compromise water quality to the point that it would result in adverse effects on ESA-listed species, their prey or their habitat. For the reasons provided above, the effects of chemicals used during the proposed training and testing activities are insignificant, and not likely to adversely affect ESA-listed species, either directly or through impacts to their prey or habitat.

As described in Section 2.2.3, a VpCI solution will be used in electrical seafloor cable conduits as part of the Navy's recurring maintenance activities in the SOAR (Navy 2024d). The initial VpCI solution is made with 1.5% VpCI diluted into 98.5% potable water at the beginning of the 24-month period of use. This corrosion inhibitor dilute solution is routinely used for this type of application in offshore areas because of its low toxicity (Holden et al. 2010). At the time of release (after 24 months of use in the conduits) the dilute VpCI solution is considered biologically degraded by OECD 301B test methodology and is no longer considered chemically effective for corrosion prevention (Navy 2025k). The biologically degraded dilute solution is not anticipated to be toxic to marine life (Holden et al. 2010). From Holden et al. (2010), the 48-Hour Static Renewal Acute Toxicity for VCI A (VpCI-649), prior to biological degradation, for

two aquatic species at the No Observable Effect Concentration (NOEC) and Lowest Observable Effect Concentration (LOEC) is as follows:

- Inland silverside (*M. beryllina*): NOEC 1,000 parts per million (ppm); LOEC 2,500 ppm
- Mysid shrimp (*M. bahia*): NOEC 600 ppm; LOEC 1,000 ppm

In an individual conduit with approximately 2,000 gallons of solution there would be 1,970 gallons (98.5%) of potable freshwater and 30 gallons (1.5%) of corrosion inhibitor. That equates to a concentration of 15,228 ppm in the conduits (Navy 2025k). As the corrosion inhibitor solution is slowly released into the water column, the concentration of the inhibitor will rapidly decrease as the solution dissipates into the water column. Based on the initial concentration of corrosion inhibitor in the conduits, the NOEC for mysid shrimp of 600 ppm would be reached once the solution has expanded into a volume of approximately 190 m^3, or just under 6 m in any direction from the release valve (Navy 2025k). This is likely a conservative approximation since the NOEC from Holden et al. (2010) is based on the toxicity of the solution prior to biodegradation (Navy 2025k). Therefore, marine species would need to be in very close proximity (i.e., less than 6 m) from the release point at the conduit to be adversely affected by the VpCI solution.

The proposed release of solution would occur approximately 3,000ft offshore and in water depths of approximately 90 ft (Navy 2025k). While black abalone would not be expected to occur at this depth, this is within the depth range of white abalone and sunflower sea stars. However, given the generally low density of white abalone and sunflower sea stars in the action area (and particularly in the offshore locations of the three conduits), and low toxicity level of the solution as discussed above, it is extremely unlikely that one of these ESA-listed invertebrate species would occur close enough to the VpCI release point to be adversely affected. ESA-listed marine mammals, sea turtles, and fish are not expected to be in close proximity to areas where VpCI solution is released due to the ability of these highly mobile species to avoid areas where Navy divers would be conducting this activity, and the generally low densities of these species in areas where this activity would occur.

In summary, given the properties of this solution and likely quick dilution in the areas where it will be used, it is extremely unlikely that ESA-listed species or their prey will be exposed to this solution at a level that would result in a measureable adverse effect. Therefore, the effects of corrosion inhibitor solution used for maintenance activities in the SOAR are discountable, and not likely to adversely affect ESA-listed species, either directly or through impacts to their prey or habitat.

Sedimentation and Habitat Disturbance from Amphibious Warfare Activities

Amphibious warfare operations will take place in the nearshore environment of MCTAB, a designated critical habitat for Hawaiian monk seals and proposed for green sea turtles. These activities, including vessels, equipment and personnel approaching or accessing the shoreline and beach, may disturb sediments and increase water turbidity. The impact of vessels on the substrate

in the surf zone would be minor because of the dispersed nature of the amphibious landings and the dynamic nature of sediments in these areas of high-energy surf. This activity may disturb the water column enough to stir up bottom sediments, temporarily and locally increasing the turbidity. The shore environment is typically highly dynamic because of its constant exposure to wave action and cycles of erosion and deposition. As a result, disturbed areas of soft-bottom habitat would be reworked by waves and tides shortly after the disturbance. In deeper waters where the tide or wave action has little influence, sediments suspended into the water column would quickly settle to the seafloor or would be carried along the bottom by currents before settling again. In either case, these disturbances would not alter the overall nature of the sediments to a degree that would impair their function as habitat or change the character of the substrate. The amphibious landing area at MCTAB is located on the windward side of Oahu and the area is marked by a high energy shorebreak which indicates a naturally occurring environment of significant sediment disturbance and increased turbidity.

Amphibious warfare activities and associated vessel movements could affect bottom sediments during amphibious landings. Ocean approaches would not be expected to affect marine habitats because of the nature of surf and tidal energy in the area of MCTAB. The movement of sediment by wave and tidal energy would fill in disturbed soft-bottom habitat similar to sediment recovery from a severe storm. Impacts on substrate would be limited to suspended sediments that are carried away by ocean currents. Ocean currents, however, would carry sediments from other locations into the action area. Therefore, sedimentation and habitat disturbance from amphibious warfare activities in the MCTAB action area would not be expected to affect designated critical habitats for Hawaiian Monk seals or proposed CH for green sea turtles.

4.1.2 Species Not Likely to be Adversely Affected

In this section we identify those ESA-listed species that we determined are not likely to be affected by any of the stressors resulting from the proposed action. In Section 4.1.1 we identified those stressors that are not likely to adversely affect any of the ESA-listed species in the action area. In this section, we consider the effects of the following additional stressors (i.e., not discussed in Section 4.1.1) on the species below: sonar and other transducers, air guns, explosives, and vessel strike.

Southern Resident Killer Whale

NMFS (2021f) identified six areas along the Pacific Coast that contain essential habitat features for Southern Resident killer whales. Southern Resident killer whales could overlap with HCTT Northern California activities in Areas 5 and 6, as shown in Figure 11. Southern Resident killer whale K and L pods have been documented to use Area 5 based on sightings, acoustic detections, and satellite tag data (NMFS 2021f). From 1982-2016, of the 49 confirmed opportunistic sightings of Southern Resident killer whales in U.S. outer coastal waters, seven of these occurred in Area 5 in January-March and October (NMFS 2021f). Analysis of the numbers of observed detections on the Fort Bragg and Pt. Reyes acoustic recorders through 2011 were much lower than expected given the amount of monitoring effort there (Hanson et al. 2013), suggesting the whales use Area 5 primarily for transiting. The median depth of waters used by Southern

Residents in Area 5 was 72.5 m (237.9 ft) and median distance from shore was 4.0 km (2.5 mi) (NMFS 2021f). Area 6 is the southernmost feeding area for Southern Residents. Southern Resident killer whale K and L pods have been documented to use Area 6 based on sightings; no acoustic recorders were located in this area, and satellite-tagged whales did not travel farther south than Point Reyes in Area 5 (Hanson et al. 2017). From 1982-2016, of the 49 confirmed opportunistic sightings of Southern Resident killer whales in U.S. outer coastal waters, seven occurred in Area 6 in January-March (NMFS 2021f).

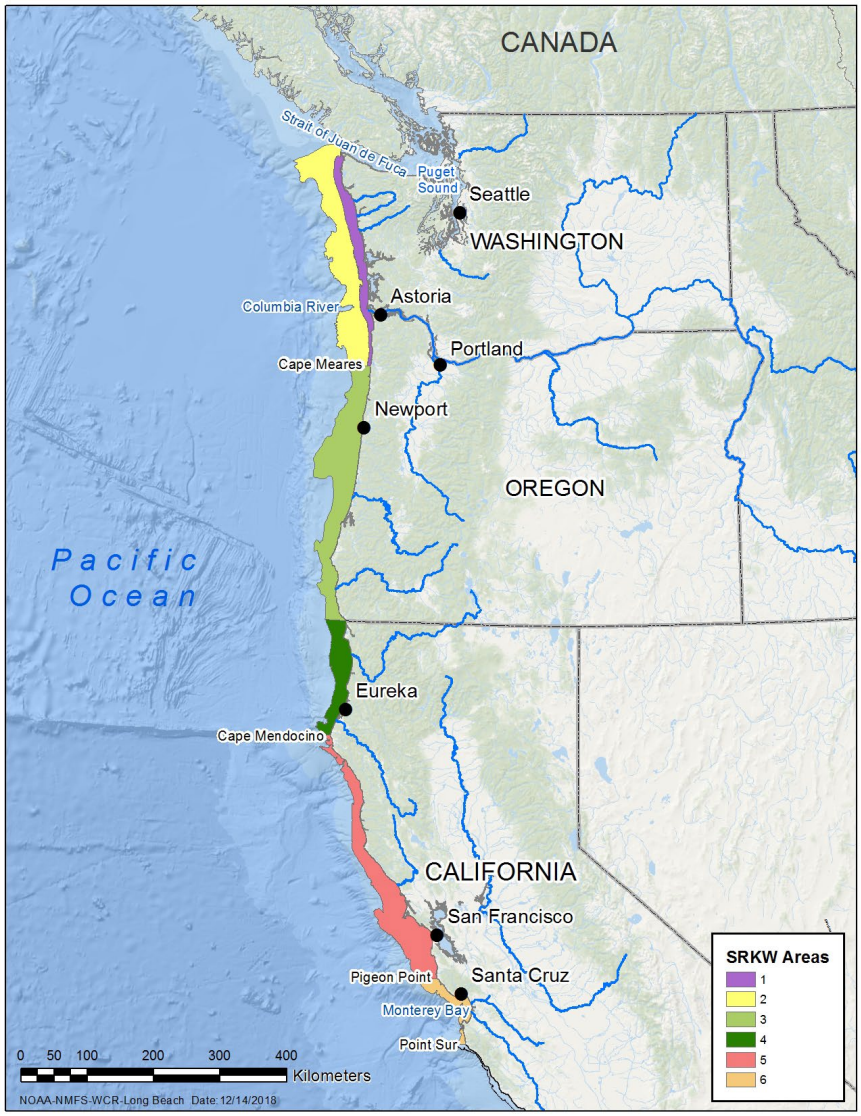


Figure 11. Areas with Essential Habitat Features of Southern Resident Killer Whale.

Although Southern Resident killer whales will overlap with the NOCAL RC portion of the action area, the large majority of proposed HCTT activities will occur outside of NOCAL RC (i.e., either in SOCAL RC or the HRC). Table 31 shows the HCTT activities with sonar and explosive stressors proposed for use within NOCAL RC. The probability of Southern Resident killer whales being exposed to explosives is very limited given the internal Navy ordnance restrictions

in NOCAL RC and the small degree of spatial overlap (2.2%) of the westernmost boundary critical habitat within W-260 (Navy 2024f). While sonar use is not restricted in W-260, the majority of proposed sonar use would likely be outside of the Southern Resident killer whale range (Navy 2024f). Based on the Navy's Acoustic Effects Model (NAEMO; see 6.1.1.1 for details), there are no predicted acoustic impacts to Southern resident killer whales from HCTT activities involving the use of sonar, explosives, or air guns. Therefore, adverse effects to Southern Resident killer whales from sonar, explosives, or air guns use are extremely unlikely (i.e., discountable). Thus, acoustic and explosive stressors resulting from the proposed action may affect, but are not likely to adversely affect, Southern resident killer whales.

Table 31. HCTT Activities with Sonar and Explosive Stressors Proposed for Use within NOCAL RC (Navy 2024f)

Activity *(training unless noted)*	Sources	Model Restrictions Within Study Area	Annual Activity Quantity Modeled
Acoustic & Oceanographic Research *(testing)*	Various sonar bins but no MFAS Acoustic impulse sources (2) Explosive bin E1, E3 (underwater)	>12 nm from shore Depths >20m	<1700 hours E1=180 E3 =90
ASW Tracking Exercise – Submarine	Various sonar bins but no MFAS No Explosives	Depths >190m	<11 hours
Composite Training Unit Exercise (COMPTUEX)- Amphibious Ready Group/Marine Expeditionary Unit	Various sonar bins No Explosives	Depths >140m	<14 hours
COMPTUEX- Carrier Strike Group	Various sonar bins No Explosives	Depths >140m	<60 hours
Medium Coordinated Anti-Submarine Warfare (ASW)	Various sonar bins No Explosives	Depths >140m	<748 hours
Small Joint Coordinated ASW	Various sonar bins No Explosives	Depths >140m	<120 hours
Submarine and UUV Subsea and Seabed Warfare Exercise	Various sonar bins but no MFAS Explosive bin E3 (underwater)	Depths >100m	<8 hours E3=5
Task Force/Sustainment Exercise	Various sonar bins No Explosives	Depths >140m	
Training and End-to-End Mission Capability Verification- Subsea and Seabed Warfare Kinetic Effectors	Explosive bin E3 (at/above surface)	(none)	E3=6
Training and End-to-End Mission Capability Verification- Unmanned Aerial Vehicle	Explosive bin E3 (at/above surface)	(none)	E3=8

MFAS= surface ship hull-mounted mid-frequency active sonar

Southern Resident killer whales could also potentially be impacted by vessel or unmanned surface vehicle strike as a result of the proposed action. 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 strikes of a Southern Resident killer whale by Action Proponent vessels or unmanned surface vehicles. Unlike large whales, which have been struck by Action Proponent vessels in recent years, the smaller size and greater maneuverability of killer whales 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 military vessels, which make up an extremely small proportion of vessel traffic in the action area. In addition, the area of overlap between the proposed action and Southern resident killer whales (i.e., NOCAL RC) represents the southern most portion of the populations' range. Densities of this extremely small population in this portion of their coastal range are anticipated to be very low throughout most of the year (NMFS 2021f). In addition, the Action Proponents will implement activity-based mitigation (Table 7) to avoid or minimize the potential for vessel strikes of marine mammals. In summary, given the low historical rate of vessel strikes of this population in general, the small number of Action Proponent vessels in the action area compared to commercial vessels, the very low densities of Southern residents expected within NOCAL RC, and the fact that an Action Proponent vessel or USV strike involving a Southern Resident killer whale has not been recorded in the past, it is extremely unlikely (i.e., discountable) that the proposed action will result in a vessel or USV strike of this species. Thus, the likely effects from vessel strike to Southern Resident killer whales resulting from the proposed action are considered discountable, and thus are not likely to cause adverse effects.

In summary, we find that the likely effects from acoustic, explosive, and vessel strike stressors to Southern Resident killer whales resulting from the proposed action are considered discountable, and thus are not likely to cause adverse effects. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect the Southern Resident killer whale.

North Pacific Right Whale

The North Pacific right whale inhabits the Pacific Ocean, particularly between 20 and 60 degrees latitude. The species is split into two populations: Eastern North Pacific and Western North Pacific. 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. The only recent estimate of North Pacific right whale abundance

comes from mark-recapture analyses of photo-identification and genetic data. Photographic (18 identified individuals) and genotype (21 identified individuals) data through 2008 were used to calculate the first mark-recapture estimates of abundance for right whales in the Bering Sea and Aleutian Islands, resulting in separate estimates of 31 and 28, respectively (Wade et al. 2011). These estimates may only relate to a subpopulation that uses the Bering Sea as there is no current estimate for right whales in the Gulf of Alaska. However, while the total size of the Eastern North Pacific population may be somewhat higher than the Wade et al. (2011) estimates, given the extreme paucity of sightings in the Gulf of Alaska, it seems unlikely that the overall abundance is significantly larger (Young et al. 2024).

While right whale sightings are reported south of the Bering Sea, such occurrences are considered rare events, particularly within the HCTT action area. There have been a total of 16 North Pacific right whale sightings along the coasts of California and Mexico between 23º N and ~37º N between the 1970s and present. Occasional sightings of right whales have been made off California, including two recent records from California in 2017 (off La Jolla and in the Channel Islands, both of which were single whales) as well as a sighting of a single skim-feeding right whale off Año Nuevo, CA in April 2022 and an animal in Monterey Bay in March 2023 (Young et al. 2024). Though the North Pacific right whale habitat off California may serve as viable calving grounds (Good and Johnston 2010), there has been no evidence that calving (or breeding) was taking place when these sightings occurred. North Pacific right whale sightings off of Hawaii are also rare events, with only three such sightings since the 1970s (NMFS 2024e).

Based on recent sightings, either within or in the vicinity of the HCTT action area, it is possible that North Pacific right whales could overlap with stressors resulting from the proposed action. However, the extremely low estimated population size of North Pacific right whale in the Eastern North Pacific Ocean, and the rarity of reported sighting throughout the HCTT action area, suggests that densities of this species within the action area are extremely low year-round. With such low densities, it is extremely unlikely that a North Pacific right whale would be exposed to HCTT acoustic (i.e., sonar or air guns) or explosives stressors, which occur sporadically and are dispersed throughout the large action area. Similarly, it is extremely unlikely that the proposed action would result in a North Pacific right whale vessel strike given the extremely small population size of this species, anticipated very low densities throughout the action area, and the fact that there are no historical records of a North Pacific right whale being struck by any vessel (military or commercial) in Hawaii or California.

In summary, we find that the likely effects from acoustic, explosive, or vessel strike stressors to North Pacific right whales resulting from the proposed action are considered discountable, and thus are not likely to cause adverse effects. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect, the North Pacific right whale.

Green Sturgeon – Southern DPS

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 (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.

Although green sturgeon may overlap with the NOCAL RC portion of the action area, the large majority of proposed HCTT activities will occur outside of the range of this species (i.e., either in SOCAL RC or the HRC). Based on the available information on green sturgeon Southern DPS distribution along the west coast, we find it extremely unlikely that this DPS will overlap with stressors from HCTT activities outside of NOCAL RC. Table 31 (above) shows the HCTT activities with sonar and explosive stressors proposed for use within NOCAL RC. The probability of a Southern DPS green sturgeon being exposed to sonar or explosives is also very limited given the internal Navy ordnance restrictions in NOCAL RC. The large majority of these activities, including all activities involving active sonar, will occur in ocean areas where green sturgeon either do not occur or are only rarely found (i.e., water depths greater than 100 m or areas beyond 12 nm from the shore). In addition, based upon the fish hearing and frequency overlap, it is anticipated that green sturgeon would not be able to hear Navy sonars or other transducers with operating frequencies greater than about 1 to 2 kHz. Given the anticipated low densities of Southern DPS green sturgeon in restricted areas where sonar would be used in NOCAL RC, the relatively small amount of low-frequency sonar activities in these areas, and the limited frequency range over which sturgeon may be affected by active sonar, it is extremely unlikely that this species will be adversely affected by HCTT sonar use.

The only planned NOCAL RC explosives activities with no restrictions on ocean location are Training and End-to-End Mission Capability Verification: Subsea and Seabed Warfare Kinetic Effectors and Unmanned Aerial Vehicle (Table 31). For these activities, all explosives will be detonated at or above the surface. Therefore, a large portion of the energy from these detonations will be reduced before entering the water column. Since green sturgeon typically occur on the seafloor, and are seldom seen at or near the surface in offshore areas, they would be less likely to be exposed to detonations at or above the surface. In addition, the explosive munitions used during these activities are considered relatively small (E3 > 0.5–2.5 lb. NEW), and only a small number of explosive detonations will be used (14 annually in a maximum year of training). All HCTT activities involving in-water explosives within NOCAL RC are restricted to ocean areas more than 12 nm from shore or depths greater than 100 m. For example, Acoustic & Oceanographic Research with relatively small NEW (bin E1 and E3 in-water explosives) will be limited to ocean areas greater than 12 nm from shore, and Submarine and UUV Subsea and Seabed Warfare exercises (bin E3 in-water explosives) will only occur in water depths exceeding 100 m. Green sturgeon are expected to be extremely rare in NOCAL RC areas with these spatial restrictions. Given the anticipated low densities of Southern DPS green sturgeon in restricted

areas where activities involving in-water explosives would be used in NOCAL RC, the limited number and type of such activities proposed in NOCAL RC in general, and the relatively small NEWs of the explosives used, it is extremely unlikely that this species will be adversely affected by HCTT explosive use.

Vessel traffic and USV use during HCTT activities would be concentrated in certain portions of the action area such as areas near ports 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 Action Proponent 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 HCTT vessel and USV traffic is concentrated is anticipated to be very low. As a benthic associated species, green sturgeon are most susceptible to vessel strike from large vessels operating in inland shallow water areas. However, the area of overlap between green sturgeon and HCTT vessels and USVs would primarily be in the deeper water ocean environment, where green sturgeon swimming along the ocean bottom are less likely to be affected by Action Proponent vessels or USVs. Given the anticipated low density of green sturgeon in the action area, the low level of vessel activity associated with HCTT activities relative to non-military traffic in the area where green sturgeon may occur, and the lack of documented cases of military vessels or USVs striking this species (or any other fish species) in the action area, it is extremely unlikely that a vessel or USV associated with HCTT activities will strike a green sturgeon. Therefore, potential effects on green sturgeon considered in this opinions from vessels and USV strike are discountable, and thus not likely to cause adverse effects.

In summary, we find that the likely effects from acoustic, explosive, or vessel strike stressors to green sturgeon Southern DPS resulting from the proposed action are considered discountable, and thus are not likely to cause adverse effects. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect, the green sturgeon Southern DPS.

Sunflower Sea Star

NMFS proposed to list the sunflower sea star as a threatened species under the ESA on March 16, 2023 (88 Fed. Reg. 16212). The sunflower sea star is a large, fast-moving, many-armed sea star native to the west coast of the continent of North America from the Aleutian Islands, Alaska, to Baja California, Mexico (Lowry et al. 2022). The species is considered a habitat generalist (Gravem et al. 2021) and occurs over a broad array of soft-, mixed-, and hard-bottom habitats. The sunflower sea star occupies a wide range of benthic substrates including mud, sand, shell, gravel, and rocky bottoms while roaming in search of prey (Lambert 2000; Konar et al. 2019). They dwell in the low intertidal and subtidal zones to a depth of 435 meters (1,427 feet) but are most common at depths less than 25 meters (82 feet) and rare in waters deeper than 120 meters (394 feet) (Lambert 2000; Hemery et al. 2016; Gravem et al. 2021; Lowry et al. 2022). Larval and pre-metamorphic sunflower sea stars are planktonic feeders, while the adult diet consists of

benthic and mobile epibenthic invertebrates, including sea urchins, snails, crab, sea cucumbers, and other sea stars (Mauzey et al. 1968; Shivji et al. 1983).

Prior to 2013, the sunflower sea star global abundance was estimated at several billion animals (Lowry et al. 2022). In Southern California, sites in the Channel Islands have been studied extensively: from 1982-2014 densities ranged from 0-0.25/m^2 (Bonaviri et al. 2017); from 1996-98 they ranged from 0- 0.02/m^2 (Eckert 2007); from 2003-2007 they ranged from 0-0.07/m^2 (Rassweiler et al. 2010); and from 2010-2012 they ranged from ~0.10-0.14/m^2 (Eisaguirre et al. 2020).

From 2013-2017 sea star wasting disease reached pandemic levels, killing an estimated 90% or more of the population (Hamilton et al. 2021; Heady et al. 2022; Lowry et al. 2022). Prentice et al. (2025) identified the bacterium *Vibrio pectenicida* as the causative agent of sea star wasting disease. While populations of this species have declined sharply in all 12 defined regions (Hamilton et al. 2021), the 6 regions south of the Washington outer coast have experienced the largest declines (over 99.5%). At present, sunflower sea stars are exceptionally rare south of Washington, and there are no reported observations of recruits or adults in California or Mexico since 2017 (Lowry et al. 2022). No sunflower sea stars were sighted during 18 surveys at San Clemente Island from 2014-2022 (Navy 2023).

Marine invertebrates are likely only sensitive to water particle motion caused by nearby low-frequency sources (less than 1 kHz), and likely do not sense distant or mid- and high-frequency sounds. Sunflower sea star hearing is expected to be similar to other marine invertebrates, which are generally thought to perceive sound via either external sensory hairs or internal statocysts. Many aquatic invertebrates have ciliated "hair" cells that may be sensitive to water movements, such as those caused by currents or water particle motion very close to a sound source (Budelmann 1992b; Budelmann 1992a; Mackie and Singla 2003). This may allow sensing of nearby prey or predators, or help with local navigation. Detection of particle motion is thought to occur in mechanical receptors found on various body parts (Roberts et al. 2016).

The effects of acoustic stressors associated with HCTT activities on ESA-listed invertebrate species are not known. Compared to some other taxa of marine animals (e.g., fishes, marine mammals), little information is available on the potential impacts on marine invertebrates from exposure to sonar and other sound-producing activities (Hawkins et al. 2015). Historically, many studies focused on squid or crustaceans and the consequences of exposures to broadband impulsive air guns typically used for oil and gas exploration. More recent investigations have included additional taxa (e.g., molluscs) and sources, although extensive information is not available for all potential stressors and impact categories. Furthermore, the shallow water, inter-tidal zones at San Clemente Island that serve as sunflower sea star habitat are not areas the Navy would train or test with acoustic sources. Although some limited testing with mid to high-frequency sources could occur in open waters adjacent to San Clemente Island, these sound sources would likely be outside the anticipated hearing range of most marine invertebrates. Therefore, based on the best available information, we find it extremely unlikely that sunflower

sea stars would be exposed to acoustic stressors (i.e., sonar and other transducers, and air guns) resulting from HCTT activities.

Sunflower sea star could be exposed to HCTT surface and underwater explosions and associated underwater impulsive sounds. The majority of explosions would occur in the air or at the surface, with relatively few at the bottom (Navy 2024d), which would decrease the potential for impacts to benthic invertebrate species. The Action Proponents typically do not conduct training and testing activities that use in-water explosives in the shallow water habitat preferred (i.e., depths less than 25 meters) by the sunflower sea star. Large explosive activities are conducted mainly at designated deep water offshore subareas, typically greater than 1,400 feet water depth, where the sunflower sea star does not occur (Navy 2024d). We anticipate the large majority of annual underwater detonations within the California portion of the action area would occur at designated areas within the Silver Strand Training Complex (Navy 2024d) where the sunflower sea star is not known to occur(Navy 2023). Therefore, there will be no exposures of sunflower sea star along the California mainland, San Clemente Island, and San Nicolas nearshore subtidal zones due to large explosions. Based on the extremely low probability of occurrence, coupled with the other assumptions described above, we consider it extremely unlikely that the sunflower sea star will be exposed to explosive stressors as part of the action. Therefore, potential effects to sunflower sea star from explosive stressors are considered discountable.

Species that do not occur near the surface within the action area, including sunflower sea stars, would not be exposed to vessel strikes. Sunflower sea stars could potentially be affected by amphibious landings (amphibious assault, insertion, and extraction), which would occur in areas where amphibious forces could make a beach landing on sandy beaches (Navy 2024d). The Stern Landing Vessel jacking legs and footpads would make contact with soft bottom nearshore areas and would not be used near hardbottom habitats (Navy 2024d). Where the jacking legs are lowered, sunflower sea stars if present directly under the footpads would likely be crushed. However, due to the infrequent use of the Stern Landing Vessel in amphibious landings (Navy 2024d), and the anticipated low density of sunflower sea stars throughout California portion of the action area, the incidence of striking a sunflower sea star would be discountable.

In summary, as discussed above and in Section 4.1.1, any effects to the sunflower sea star from the proposed action are expected to be either discountable or insignificant. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect, the sunflower sea star.

Black Abalone

The black abalone was listed as endangered on February 14, 2009 (74 Fed. Reg. 1937). The primary threat leading to the listing of this species is withering syndrome, which has reduced black abalone populations by 99% in some areas. Black abalone historically occurred from Crescent City, California to southern Baja California, Mexico (Butler et al. 2009), but today the

species' constricted range occurs from Point Arena, California to Bahia Tortugas, Mexico, and it is rare north of San Francisco, California (Butler et al. 2009), and south of Punta Eugenia, Mexico (76 Fed. Reg. 66805).

The black abalone is a large (up to eight inches), long-lived (up to 30 years), herbivorous marine gastropod found in rocky intertidal and subtidal habitats. Abalone are slow-moving bottom dwellers that attach to rocks and other hard surfaces using their muscular foot. During low tides, they can typically be found wedged into crevices, cracks, and holes of intertidal and shallow subtidal rocks, where they are fairly concealed (Leighton 2005). They generally occur in areas of moderate to high surf and range vertically from the high intertidal zone to a depth of about 20 ft (6 m) and are typically found in middle intertidal zones. Variation in wave exposure and where drift kelp (an important food item for black abalone) accumulates may result in animals being distributed primarily in high or low intertidal zones depending on the local conditions at particular locations.

Occurrence in the HCTT Action Area

The U.S. Geological Survey monitors a suite of intertidal black abalone sites at San Nicolas Island, California, in cooperation with the U.S. Navy. The San Nicolas Island black abalone population sustained significant declines between the late 1980s and early 1990s, though to a lesser degree than other geographical locations. During the last two decades, the total monitored black abalone population around the island has grown approximately ten-fold after the disease related decline, from about 200 to more than 2,000 abalone (Kenner and Yee 2022). The total San Nicolas Island black abalone count in 2022 was 2,156, which was 7.9 percent lower than the total count in 2020 but 6.6 percent higher than in 2021 (Kenner and Yee 2025). There were increases and decreases among the sites and transects within each site in 2022, but six of the nine sites had higher counts than in the previous year. The 2022 count is one of the highest since 1993, second only to the 2020 count. In 2022, the annual recruitment rate, defined as the percentage of measured black abalone with a shell length of 3 cm or less, was the second highest recorded, only slightly less than in 2017 (Kenner and Yee 2025). Since it was first consistently measured in 2005, the mean distance between adjacent black abalone off San Nicolas Island has decreased substantially from approximately 50 cm to less than 15 cm, indicating that abalone are close enough together at several of the sites to reproduce successfully (VanBlaricom et al. 2015; Kenner and Yee 2022; Kenner and Yee 2025).

An intensive survey aimed at recording black abalone distribution at San Clemente Island was conducted in January 2008 (Tierra Data 2008). The survey was performed at 61 locations between Northwest Harbor and Pyramid Head along the west shore, within primary abalone habitat. Ten black abalone were recorded, with most occurring at locations previously documented to support abundant populations (e.g., West Cove, Eel Point, Mail Point; See Figure 12). Black abalone recorded all ranged from four to five inches (100 to 130 mm) long and there

were no signs of recruitment (fresh shells). Based on the area surveyed, approximate black abalone density at San Clemente Island is one abalone per 2.3 acres (0.9 ha).

From 2011 to 2013, researchers from the University of California Santa Cruz surveyed between 13.6% and 20.7% of the rocky coastline on San Clemente Island, with sites located on all sides of the island (Tierra Data 2013). From this study, Raimondi (2012) estimated the black abalone population size in the nearshore waters of San Clemente Island was 187, with a 90% confidence interval between 91 and 344 individuals. The average size in 2011-2012 was about 4.7 inches (119.5 mm), which is similar to the average size of black abalone measured in the 2008 surveys. There were no individuals smaller than 3.1 inches (80 mm) found, and individuals were significantly larger in moderate habitat than in good habitat (Tierra Data 2013). Rocky intertidal areas surveyed around San Clemente Island contained more poor abalone habitat than good and moderate habitat combined (Tierra Data 2013).

In 2016, six sites along the west coast of San Clemente Island were surveyed for previously documented abalone (Fletcher et al. 2021). A total of 11 abalone were found compared to the 31 abalone that were previously found at these locations in the 2011-2013 surveys. Additional surveys conducted in 2019 and 2020 at eight sites found 25 black abalone, compared to 42 individuals that were documented at these sites during the 2011-2013 surveys (Fletcher et al. 2021). Overall, these data show a 65% decline in the number of black abalone found in 2011-2013 surveys and 2016 efforts and a 41% decline between 2011-2013 and 2019-2020 surveys (Fletcher et al. 2021). Although no signs of withering syndrome were observed in the San Clemente Island population, this decline correlates with the anomalous warm water event that occurred in the eastern Pacific from 2014 to 2015 (Fletcher et al. 2021).

ESA-listed abalone could potentially be exposed to acoustic stressors as a result of the proposed action. Abalone are likely only sensitive to water particle motion caused by nearby low-frequency sources, and likely do not sense distant or mid- and high-frequency sounds. Abalone hearing is expected to be similar to other marine invertebrates, which are generally thought to perceive sound via either external sensory hairs or internal statocysts. Many aquatic invertebrates have ciliated "hair" cells that may be sensitive to water movements, such as those caused by currents or water particle motion very close to a sound source (Budelmann 1992b; Budelmann 1992a; Mackie and Singla 2003). This may allow sensing of nearby prey or predators, or help with local navigation. Detection of particle motion is thought to occur in mechanical receptors found on various body parts (Roberts et al. 2016). Aquatic invertebrates that are able to sense local water movements with ciliated cells include molluscs such as abalone (Budelmann 1992b; Budelmann 1992a; Popper et al. 2001). A range of marine invertebrates are reported to be sensitive to low-frequency (10 to 150 Hz) hydroacoustic disturbances induced by sound waves or other sources – e.g., jellyfish, crustaceans, arrow worms, octopus, and squid (Western Australian Department of Industry and Resources 2002; Albert 2011; Sole et al. 2016). This sensitivity overlaps the frequency range of low-frequency active (LFA) sonar (100 to 500 Hz), indicating

that some mobile invertebrates could perceive LFA sonar training and testing activities (Western Australian Department of Industry and Resources 2002; Albert 2011; Sole et al. 2016).

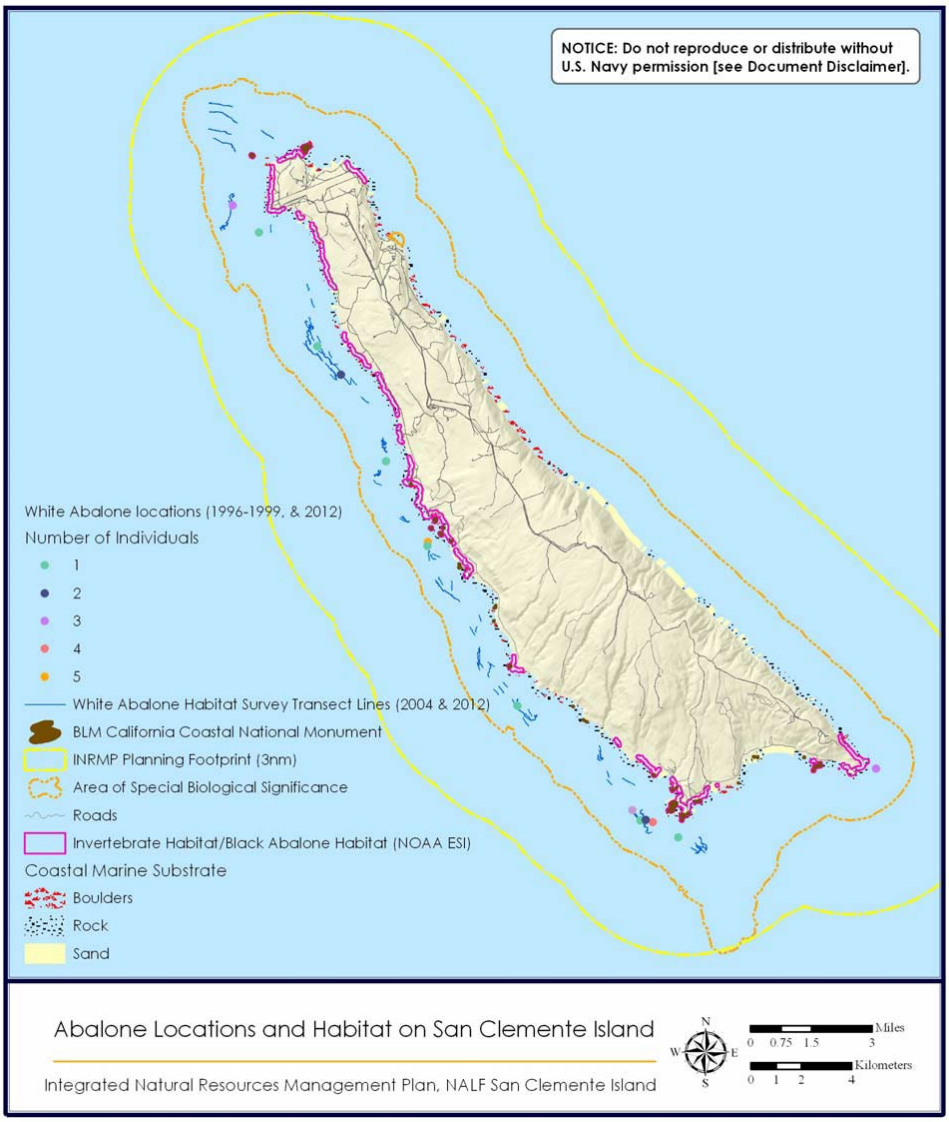


Figure 12. White and Black Abalone Habitat in the Nearshore Waters of San Clemente Island (Tierra Data 2013).

The effects of acoustic stressors associated with HCTT activities on ESA-listed abalone species are not known. Compared to some other taxa of marine animals (e.g., fishes, marine mammals), little information is available on the potential impacts on abalone from exposure to sonar and other sound-producing activities (Hawkins et al. 2015). Historically, many studies focused on squid or crustaceans and the consequences of exposures to broadband impulsive air guns typically used for oil and gas exploration. More recent investigations have included additional taxa (e.g., molluscs) and sources, although extensive information is not available for all potential stressors and impact categories. Furthermore, the shallow water, inter-tidal zones at San

Clemente Island that serve as black abalone habitat are not areas the Action Proponents would train or test with acoustic sources, although some limited testing with mid to high-frequency sources could occur in open waters adjacent to San Clemente Island. While ESA-listed invertebrates that detect low-frequency sounds may exhibit behavioral changes during training or testing activities, current scientific understanding does not support that acoustic sources at predicted levels in the operational areas would cause significant biological impacts. The relatively simple sensory mechanisms of these species, combined with their habitat distribution patterns and existing HCTT operational protocols, indicates that acoustic impacts from military readiness activities likely have insignificant effects on ESA-listed abalone. Therefore, the likely effects from exposure to sonar and other transducers resulting from the proposed action on black abalone are considered minor and insignificant, and thus are not likely to adversely affect this species.

Under the proposed action, black abalone could be potentially exposed to surface and underwater explosions and associated underwater impulsive sounds. As discussed above, black abalone generally occur in areas of moderate to high surf and range vertically from the high intertidal zone to a depth of about 20 ft (6 m) and are typically found in middle intertidal zones (Butler et al. 2009). Black abalone are generally not found in the offshore areas where the Action Proponents train or test with explosives (Navy 2024d). While explosives could be used throughout the SOCAL RC, in practice most explosive activities are conducted within designated deep water (i.e., > 1,400 ft) offshore subareas where black abalone do not occur (Navy 2024d). A large majority of underwater detonations within the SOCAL RC will likely occur at designated areas within the Silver Strand Training Complex, where black abalone are not known to occur. It is extremely unlikely for black abalone to be exposed stressors from explosives along the California mainland, San Clemente Island, and San Nicolas intertidal zones. In addition, the majority of HCTT explosions in the California portion of the action area will occur in the air or at the surface, with relatively few at the bottom (Navy 2024d), which would decrease the potential for impacts to benthic dwelling species such as abalone. A relatively small number of underwater detonations associated with Mine Warfare training are expected to occur in the Northwest Harbor of San Clemente Island (Navy 2024d). The shallow water areas of Northwest Harbor are primarily sandy bottom habitat that is not suitable for black abalone. The Navy has committed to avoiding conducting underwater detonations in shallow, rocky bottom habitats around San Clemente Island where suitable black abalone habitat could occur.

As discussed in Section 2.4.2, mitigation to avoid impacts from explosives on seafloor resources may further reduce the likelihood of exposure to benthic invertebrates, including ESA-listed abalone. Explosive mine countermeasure and neutralization activities would not be conducted within a specified distance (i.e., 350 yards) of shallow-water coral reefs, precious coral beds, artificial reefs, shipwrecks, or intertidal and nearshore areas of San Nicolas Island. HCTT underwater explosions would also not overlap with designated black abalone critical habitat, as described in Section 4.1.3.

Based on the extremely low probability of occurrence, coupled with the other assumptions described above, we consider it extremely unlikely for black abalone to be exposed to stressors resulting from HCTT explosives. Therefore, potential effects on black abalone from explosive stressors are considered discountable.

Species that do not occur near the surface within the action area, including ESA-listed black abalone and white abalone, would not be exposed to vessel strikes. In addition, these species would not be affected by amphibious landings (amphibious assault, insertion, and extraction) since abalone inhabit rocky shores and hard bottom, which are not used for amphibious landings. These activities would occur only where amphibious forces could make a beach landing on sandy beaches. The Stern Landing Vessel jacking legs and footpads would make contact only with soft bottom nearshore areas and would not be used near hardbottom habitats where white and black abalone are commonly found (Navy 2024d).

In summary, as discussed above and in Section 4.1.1, any effects to black abalone from the proposed action are expected to be either discountable or insignificant. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect, the black abalone.

White Abalone

White abalone are herbivorous gastropods that live in rocky ocean waters. They are generally 5-8 inches (13-20 cm) long, but can grow to as big as 10 inches (25 cm), and weigh about 1.7 lbs. (0.8 kilograms [kg]) on average. White abalone are the deepest occurring abalone species on the U.S. West Coast. They are found in greatest abundance at depths of 80-100 ft (25-30 m), but have been reported at depths up to 200 ft (Cox 1960; Tutschulte 1976). White abalone occupy open low relief rock or boulder habitat surrounded by sand (Tutschulte 1976; Davis et al. 1996). Sand channels may be important for the movement and concentration of drift macroalgae and red algae, upon which white abalone are known to feed.

Historically, white abalone were found in the Pacific Ocean from Point Conception, California, to Punta Abreojos, Mexico (Hobday and Tegner 2000). In the northern part of the California range, white abalone were reported as being more common along the mainland coast. However, in the middle portion of the California range, they were noted to occur more frequently at the offshore islands, especially San Clemente and Santa Catalina Islands (Hobday and Tegner 2000). White abalone populations throughout southern California are severely depleted. Densities of white abalone were estimated to be as high as 2,300 per hectare (Tutschulte 1976) and the total population was estimated to be between 700,000 and 4.2 million individuals across their entire range (Hobday et al. 2000). A more conservative density estimate, based solely on fishery-dependent information in California, is 479 per hectare, which translates to a population size of 360,476 individuals for California alone (Rogers-Bennett et al. 2002). Over the past several decades, the white abalone populations have declined precipitously in abundance primarily as a result of exploitation.

At Tanner Bank, the white abalone population has declined by nearly 73% between 2002 and 2010 (Stierhoff et al. 2012). In 2014, Stierhoff et al. (2015) optically surveyed white abalone by conducting 28 transects at Tanner Bank and 16 transects at Cortes Bank during five days at sea. At Tanner Bank, 19 white abalone were observed, all in 40-50 m depths; no white abalone were observed at Cortes Bank. Based on this survey, Stierhoff et al. (2015) estimated the white abalone population at Tanner Bank was 3,745 animals, which was not significantly greater than the 2010 estimate of 3,375 animals. Within the 40-50m depth range, the estimated density was 8.8 white abalone per hectare. The authors concluded that these results indicate that the white abalone population at Tanner Bank has not recovered from severe depletion (Stierhoff et al. 2015). The high coefficient of variation (87 to 303%) from white abalone estimates (i.e., population and density) reflects the low numbers and patchiness of white abalone at Tanner Bank and Cortes Bank. For example, of the 44 transects surveyed in 2014, zero white abalone were observed on 35 transects; one was observed on six transects; and more than one (two, five, and six individuals) were observed on three transects.

At one time, San Clemente Island served as an important commercial and recreational source of white, green, pink, and black abalone and the island has been highlighted as an important area for current monitoring and future abalone restoration efforts. In October 1999, surveys were conducted in potential white abalone habitat areas on San Clemente Island (Figure 12). This survey was limited to the northern, western, and southern sides of the island. Most of the individuals observed were found offshore of the center of the island on the west side of San Clemente Island (Tierra Data 2013). Individuals and groups of two or more individuals were most abundant offshore from Seal Cove and Seal Point. A total of 24 white abalone were found, ranging from one to six individuals per site, at ten of the 26 sites surveyed. Abalone were found in 100 to 200 ft (30–60 m) of water, with most at approximately 157 ft (48 m) (Tierra Data 2013).

In 2004, underwater surveys using a remotely operated vehicle identified several remnant populations of white abalone, including one along the west and south shores of San Clemente Island. The surveys were conducted over a ten-day period off the west shore of San Clemente Island from Castle Rock south to China Point and consisted of multibeam and sidescan sonar from the seaward edge of the kelp beds at 82 ft (25 m) out to approximately 245 ft (75 m) (Butler et al. 2006). Extensive remotely operated vehicle surveys were conducted where suitable habitat was found to measure abalone densities. Butler et al. (2006) found all abalone at 100 to 130 ft (30–40 m) and 130 to 165 ft (40–50 m) depth ranges with none sighted at 165 to 200 ft (50–60 m). Suitable habitat on San Clemente Island was measured at 2,220 acres (889 ha) (Butler et al. 2006). In 2012, another survey of white abalone habitats was conducted at San Clemente Island to examine potential changes in that population. A total of 48 remotely-operated vehicle transects were conducted along the west and south edges of San Clemente Island using methods from the 2004 surveys. Both surveys (2004 and 2012) found that white abalone are sparse at San Clemente Island; only five white abalone (mean shell length of 17.2 cm [standard deviation = 2.2

cm] and 17.7 cm [standard deviation = 1.8 cm] in 2004 and 2012, respectively) were observed in each of the two surveys (Stierhoff et al. 2014). Average densities were 0-1.24 abalone per hectare (ha-1) in 2004 and 0.27-1.44 abalone ha-1 in 2012, which resulted in a slight increase in the population from 353 (standard error = 62) to 565 (standard error = 136) white abalone during that time (Stierhoff et al. 2014). However, the low density and patchy distribution of white abalone at San Clemente Island resulted in high coefficients of variation for population estimates in all years and depths (coefficient of variation = 0.70-0.96). In 2016, a remotely operated vehicle was used to conduct another visual transect surveys of white abalone from 30 to 60 m depths at San Clemente Island (Neuman et al. 2017). Two white abalone were observed in 36 visual transects. The low numbers of observations resulted in estimates of population size that were too imprecise for statistical comparisons to results from the 2004 or 2012 survey.

These surveys located only adult white abalone, suggesting that recruitment may not be occurring at the surveyed locations within the action area (Stierhoff et al. 2014). White abalone, like other abalone species, are cryptic and often difficult to detect during visual surveys (Stierhoff et al. 2014). They preferentially inhabit rocky substrates and are often covered in the same encrusting algae and kelp that cover their habitat, which provides effective camouflage and makes detection and positive identification challenging. The challenges associated with detection become even greater as shell size decreases, making the ability to monitor any recent recruitment or gauge recovery more difficult (Stierhoff et al. 2014).

Based on modeled historical (using fishery-dependent data from 1955–1996) and contemporary (using fishery-independent data from 1996–2017) distributions of white abalone, DiNardo et al. (2021) identified a limited amount of relative suitable habitat for white abalone. San Clemente Island exhibited the highest relative habitat suitability within the Southern California Bight, and habitat suitability was moderately high at San Nicolas Island, Santa Catalina Island, northern San Clemente Island, and at Tanner and Cortes Banks (DiNardo et al. 2021). Based on fine-scale, fishery-independent models, the most suitable habitat occurred in areas with hard substrate between 30 and 70 meter (98 and 230 feet) depth at offshore sites (San Clemente Island, Santa Catalina Island, Santa Barbara Island, Tanner Bank, and Cortes Bank); and between the depths of 10 and 30 meters (33 and 98 feet) along the mainland coast in Southern California (DiNardo et al. 2021).

There is little information available regarding the white abalone population around San Nicholas Island. Considering the historical abalone landings and requirements for suitable habitat and growth, white abalone are thought to be rarer around San Nicolas Island than San Clemente Island or other parts of the species' range.

White abalone could potentially be exposed to HCTT sonar and other transducers. We anticipate that the effects of acoustic stressors resulting from the proposed action on white abalone would be similar to the effects on black abalone described above (Section 4.1.2). Therefore, the likely effects from exposure to sonar and other transducers resulting from the proposed action on white

abalone are considered minor and insignificant, and thus are not likely to adversely affect this species.

As part of the proposed action, white abalone could be exposed to surface and underwater explosions and associated underwater impulsive sounds from high-explosive munitions (including bombs, missiles, torpedoes, and naval gun shells), mines, and demolition charges. White abalone are the deepest living abalone species on the west coast and occur at depths up to almost 200 ft. As shown in Figure 12, white abalone are generally not found in the offshore areas where the Action Proponents typically train or test with explosives (Navy 2024d). While explosives could be used throughout the SOCAL RC, in practice most explosive activities are conducted mainly at designated deep water (i.e., > 1,400 ft) offshore subareas where white abalone do not occur (Navy 2024d). A large majority of underwater detonations within the SOCAL RC will likely occur at designated areas within the Silver Strand Training Complex, where white abalone are not known to occur. It is extremely unlikely for white abalone to be exposed stressors from large explosives along the California mainland, San Clemente Island, and San Nicolas intertidal zones. In addition, the majority of HCTT explosions in the California portion of the action area will occur in the air or at the surface, with relatively few at the bottom (Navy 2024d), which would decrease the potential for impacts to benthic dwelling species such as abalone. This species could potentially be exposed to noise from surface explosions, but the likelihood of surface explosions affecting white abalone is extremely low. Surface explosions typically occur at offshore locations more than 12 nm from shore and in locations where white abalone are not known to occur. A relatively small number of underwater detonations associated with Mine Warfare training are expected to occur in the Northwest Harbor of San Clemente Island (Navy 2024d). The shallow water areas of Northwest Harbor are primarily sandy bottom habitat that is not suitable for black abalone. The Navy has committed to avoiding conducting underwater detonations in shallow, rocky bottom habitats around San Clemente Island where suitable white abalone habitat could occur. Mitigation measures (Section 2.4.2) to avoid impacts from explosives on seafloor resources may further reduce the likelihood of exposure to benthic invertebrates, including white abalone.

Based on the extremely low probability of occurrence, coupled with the other assumptions described above, we consider it extremely unlikely for white abalone to be exposed to explosions as part of the proposed action. Therefore, potential effects on white abalone from explosive stressors are considered discountable.

As discussed above for black abalone, we do not anticipate effects to white abalone from either vessel strike or amphibious landing activities (See Section 4.1.2 for details).

In summary, as discussed above and in Section 4.1.1, any effects to white abalone from the proposed action are expected to be either discountable or insignificant. Therefore, we conclude that the proposed action may affect, but is not likely to adversely affect, the white abalone.

4.1.3 Critical Habitat Not Likely to be Adversely Affected

Humpback Whale – Mexico and Central America DPSs

NMFS designated critical habitat for the Mexico and Central America DPSs of humpback whale on April 21, 2021 (86 Fed. Reg. 21082). The wintering areas for the Mexico DPS are the waters and islands off Mexico, and, for the Central America DPS, wintering areas are waters from southern Mexico and south along the coast of Central America. Critical habitat for both DPSs extends beyond the HCTT action area but includes the California Current Ecosystem, which overlaps with nearshore potions of the Action Area but does not include all of the Southern California Bight (Figure 13). The critical habitat is defined by water depth, with the nearshore limit set at 15m, 30 m, or 50 m and the offshore limit at 2,000 m, 3,000 m, or 3,700 m for different segments along the California coastline. Prey species (primarily euphausiids and small schooling fishes) of sufficient quality, abundance, and accessibility were identified by NMFS as the essential feature of the designated critical habitat.

Critical habitat units 17 and 18 overlap with the PMSR portion of the HCTT action area (Figure 14). High concentrations of krill are found in these areas supporting high density feeding aggregations of whales. These areas were found to have very high conservation value for the Central America DPS and high conservation value for the Mexico DPS of humpback whales. Unit 17, referred to as the "Central California Coast Area," covers an area of 6,697 nm^2 extending from 36 degrees 00 seconds to 34 degrees 30 seconds North latitude. Within those north and south boundaries, Unit 17 begins at the 30-m depth contour and extends out to the 3,700-m depth contour. Unit 18, referred to as the "Channel Islands Area," covers an area of 9,799 nm^2 extending from 34 degrees 30 seconds North latitude, south to a boundary line seaward to the southeast from Oxnard, CA along the 3,700-m depth contour. The shoreward boundary is formed by the 50-m depth contour.

The one PBF of designated critical habitat for the humpback whale Central America and Mexico DPSs is: euphausiids and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth (86 Fed. Reg. 21082). These schooling prey fishes include species such as capelin, herring, and mackerel. Here, we evaluate the effects of the proposed action on this PBF.

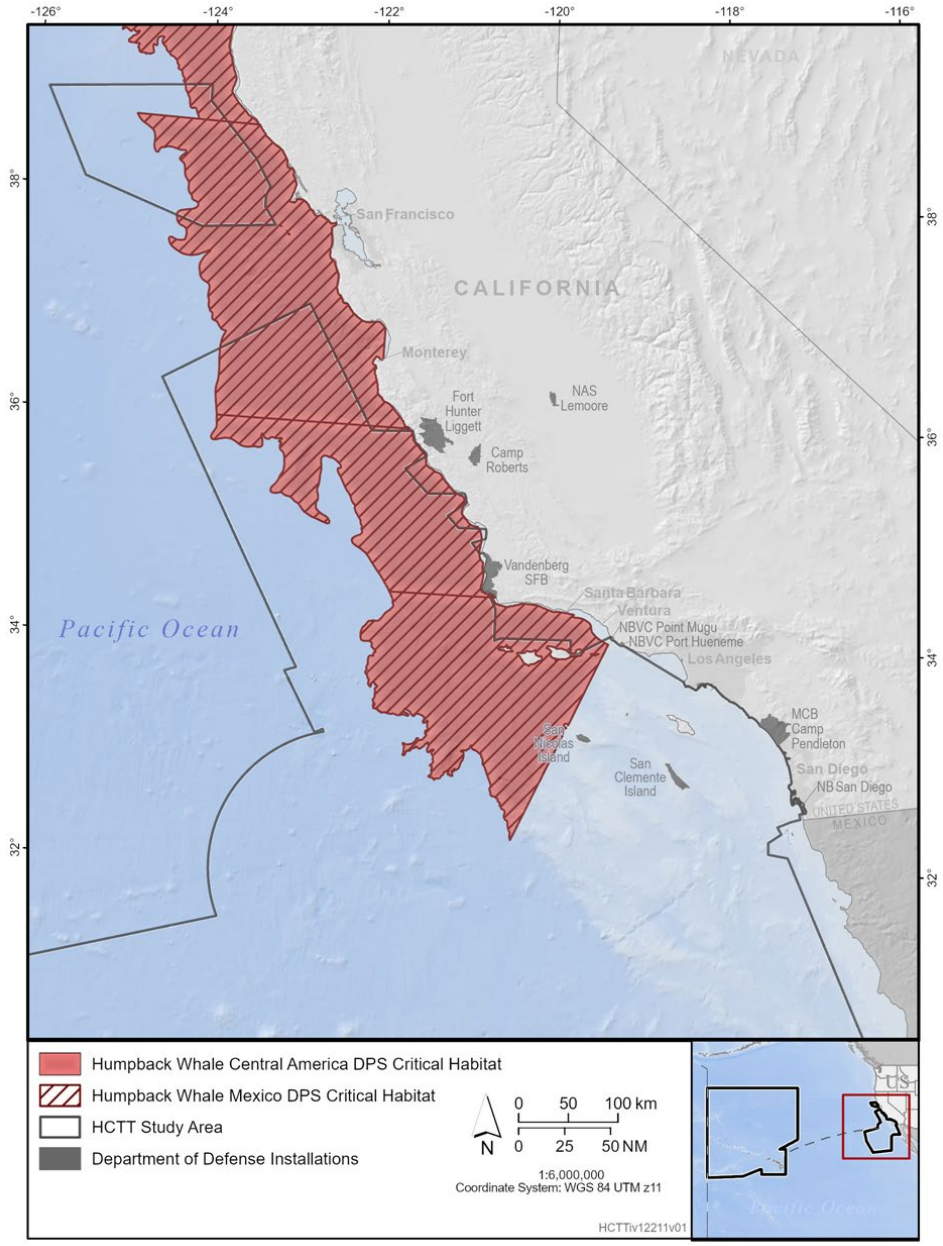


Figure 13. Humpback Whale Critical Habitat in the HCTT Action Area (Navy 2024e)

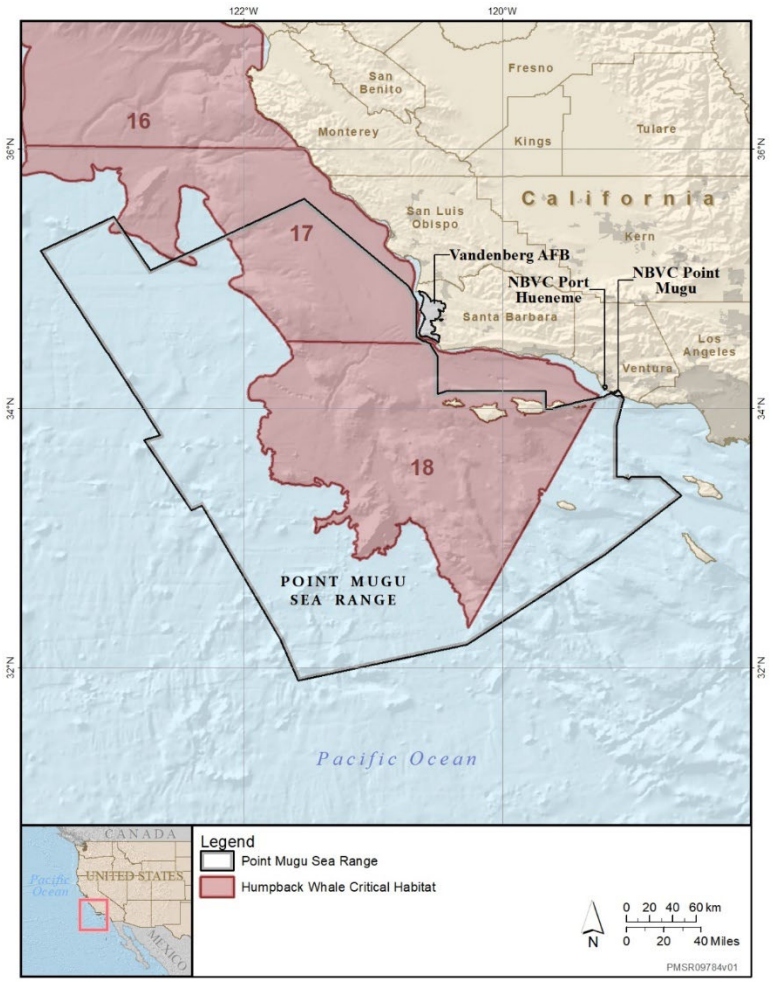


Figure 14. Overlap of Humpback Whale Critical Habitat Units 17 and 18 with the PMSR

Stressors Previously Determined NLAA for Fish and Invertebrates

In Section 4.1.1 we determined that several of the stressors associated with the proposed action were not likely to adversely affect ESA-listed fish species because any effects from these stressors were either discountable or insignificant. For the same reasons provided in Section 4.1.1 (see Fish subsections), we find that the following stressors are not likely to adversely affect the prey (small schooling fish and krill) PBF of humpback whale critical habitat: vessel noise, aircraft noise and disturbance, weapons noise, in-water electromagnetic devices, in-water devices, seafloor devices, and military expended materials.

Acoustic Effects from Sonar, Explosives, and Air Guns

The HCTT proposed action includes a relatively small number of activities involving low-frequency sonar use in areas that would overlap with fish and invertebrate prey species of humpback whales. Humpback whale prey species may only detect low-frequency sonar systems with higher source levels within a few kilometers (km); and most other, less powerful low-frequency sonar systems, at much shorter ranges. Prey species 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. Prey species located closer to the sonar sound source would likely experience more substantive responses, whereas those located further away from the source are less likely to react to the sound levels. However, because HCTT sonar sources are continuously moving, and prey are also capable of moving away from the disturbance, the overall exposure duration is expected to be brief. 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 humpback whale prey species are expected, and the effects described above are not anticipated to lead to a significant disruption of normal behavior patterns. As such, the effects from sonar and other transducers on the humpback whale critical habitat prey PBF are considered insignificant.

ESA-listed humpback whale prey species may be affected by impulsive acoustic sources (i.e., explosives and airguns) in the offshore portion of the action area overlapping designated critical habitat. Adverse effects may include injury, TTS, physiological stress, behavioral reactions, and mortality. Those fish that are killed within the designated 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. McCauley et al. (2017) found that the use of a single airgun lead to a decrease in zooplankton abundance by over 50% and a two- to three-fold increase in dead adult and larval zooplankton when compared to control scenarios. In addition, effects were found out to 1.2 km (0.6 nm), the maximum distance to which the sonar equipment used in the study was able to detect changes in abundance. Richardson et al. (2017) scaled up the results of McCauley et al. (2017) to examine the effects of a hypothetical seismic survey on zooplankton off the coast of Australia. Based on their results, seismic surveys had a significant impact on the abundance of zooplankton within and near the seismic survey area, but such effects were short-lived and minimized by ocean circulation.

The use of HCTT explosives and air guns within (or adjacent to) areas of coastal designated 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 nm from shore.

The anticipated impacts from explosives and air guns 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 HCTT offshore explosives and air guns 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 HCTT explosives and air guns is extremely low relative to the number of prey items found throughout the action area and would not be detectable above baseline conditions.

If prey items are killed within humpback whale designated 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 would be highly dependent on the limited number of activities involving explosives and air guns that overlap critical habitat and the actual presence of prey species at the time such activities occur. The portion of HCTT activities using explosives and air guns within the 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 designated critical habitat compared to the relatively small size of the offshore area where detonations or air guns 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 2.4.2) and the relatively large number of prey items available throughout the critical habitat, we conclude that any impacts of explosives and air guns resulting from HCTT activities on prey availability for the humpback whale Central America and Mexico DPSs would be insignificant at the population level.

Capture in Submerged Nets used for Obstacle Avoidance

The Navy has proposed to use submerged nets for subsurface obstacle avoidance training and testing with unmanned underwater vehicles. Given the small mesh size of the Navy's proposed nets, it is possible that some humpback whale prey species could be entangled, which could lead to injury and mortality of entangled individuals. However, given the relatively small amount of time that nets would be in the water, and the relatively small area that the nets would cover in relation to the large area designated as critical habitat (within the action area), we expect any impacts to the prey abundance available to humpback whales to be insignificant (i.e., not measureable). Therefore, the likely effects of entanglement in submerged nets on the humpback whale prey PBF are considered insignificant.

Summary

In summary, the effects of all stressors analyzed on the essential humpback whale prey PBF were found to be either insignificant or discountable and not likely to reduce the conservation value of designated critical habitat. Although the use of impulsive acoustic sources (i.e., explosives and air guns) and submerged nets with small mesh size would likely result in injury and mortality to humpback whale prey species within designated critical habitat areas, we have no information to indicate that these stressors 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. Therefore, for the reasons provided above, we determine that the proposed action may affect, but is not likely to adversely affect, critical habitat for the Central America and Mexico DPSs of humpback whales.

Southern Resident Killer Whale

In 2006 NMFS designated critical habitat for the Southern Resident killer whale DPS (71 Fed. Reg. 69054). On August 2, 2021, NMFS revised the critical habitat designation for Southern Resident killer whales by expanding it to include six new areas along the U.S. West Coast (84 Fed. Reg. 49214), while keeping the current designated critical habitat area in Washington (Figure 15). The new areas along the U.S. West Coast include roughly 15,910 mi^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. Critical habitat for Southern Resident killer whales overlaps with a small portion of the HCTT action area in the NOCAL RC (Figure 16). The spatial overlap is 127 nm^2 (435 km^2), or 2.2% of W-260, and 0.051% of the California portion of the HCTT action area. The area of overlap is designated as Unit 5 North Central CA Coast Area and Unit 6 Monterey Bay Area of Southern Resident killer whale critical habitat.

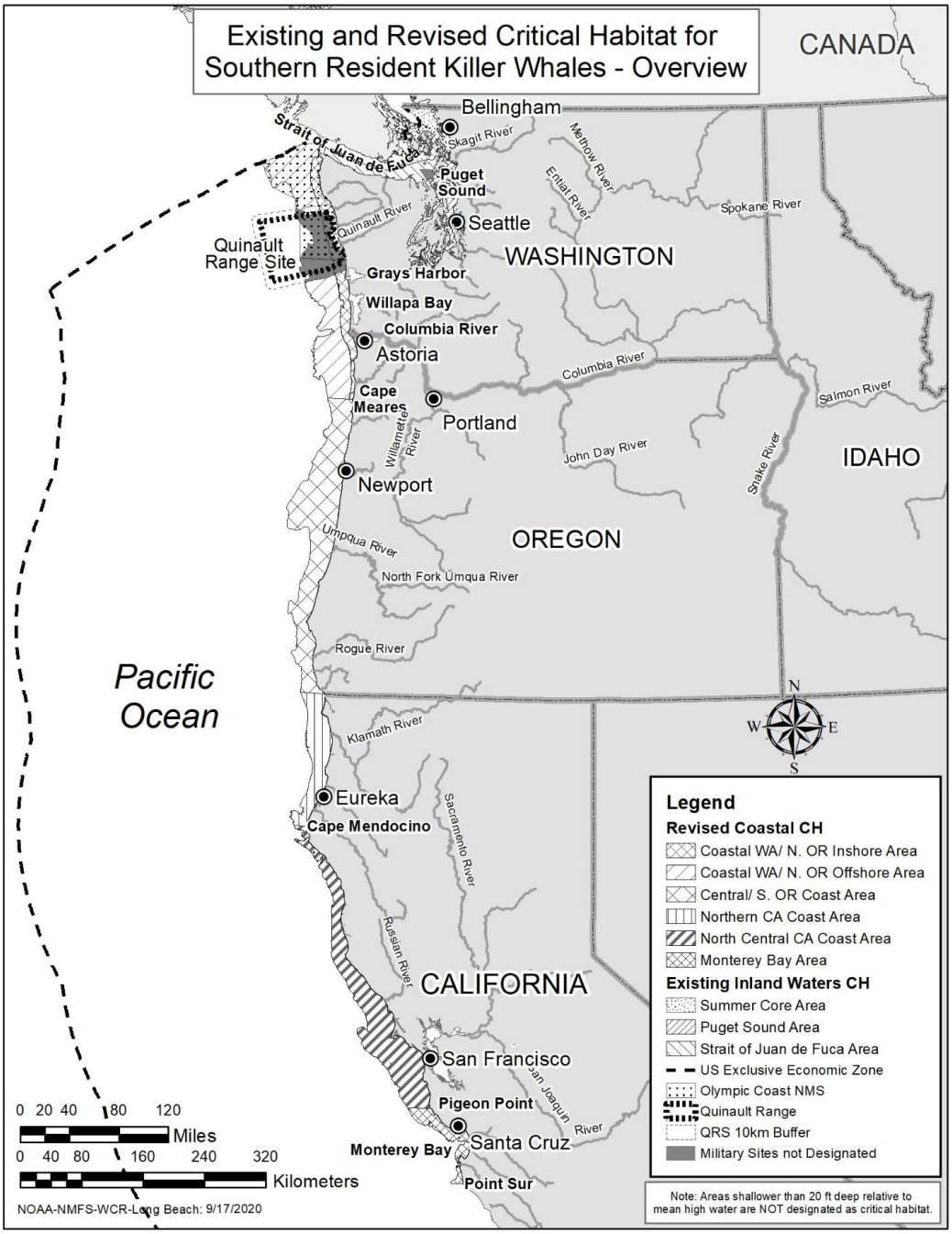


Figure 15. Southern Resident Killer Whale Designated Critical Habitat

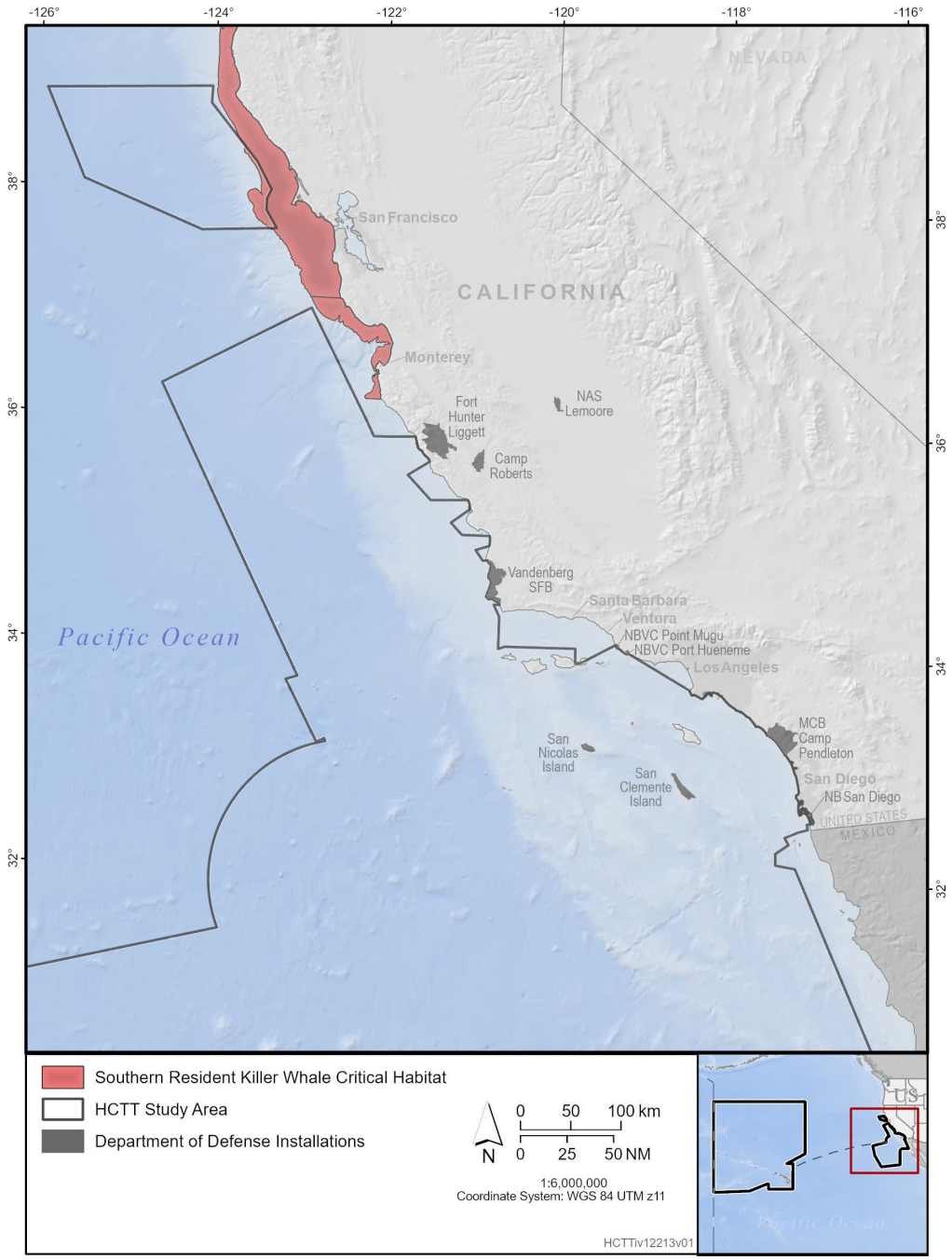


Figure 16. Map Showing Overlap between Southern Resident Killer Whale Critical Habitat and the HCTT Action Area (Navy 2024e)

The PBFs of coastal critical habitat that are essential to the conservation of Southern Resident killer whale DPS 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) Passage conditions to allow for migration, resting, and foraging.

Although Southern Resident killer whale critical habitat overlaps with the NOCAL RC, the large majority of proposed HCTT activities will occur outside of NOCAL RC (i.e., either in SOCAL RC or the HRC). In addition, the area of overlap represents an extremely small portion of both Southern Resident killer whale critical habitat and the HCTT action area. In general, these factors limit both the magnitude and type of stressors that Southern Resident killer whale critical habitat could potentially be exposed to.

Stressors Previously Determined NLAA

In Section 4.1.1 we analyzed potential impacts to ESA-listed species exposed to stressors indirectly through impacts to their habitat or prey. The stressors evaluated in that section included explosives, explosive byproducts and unexploded munitions, metals, and chemicals. For each of these stressors, we concluded that the effects to water quality resulting from the proposed action were insignificant (i.e., so minor as to not be measureable). Based on this analysis, we reach the same conclusion (i.e., insignificant) for the effects of the proposed action on the water quality PBF of Southern Resident killer whale critical habitat.

In Section 4.1.1 we determined that several of the stressors associated with the proposed action were not likely to adversely affect ESA-listed fish species because any effects from these stressors were either discountable or insignificant. For the same reasons provided in Section 4.1.1 (see Fish subsections), we find that the following stressors are not likely to adversely affect Southern Resident killer whale fish prey or the critical habitat prey PBF (#2 above): vessel noise, aircraft noise and disturbance, weapons noise, in-water electromagnetic devices, in-water devices, seafloor devices, military expended materials, entanglement in wires and cables, decelerators and parachutes, and nets used for obstacle avoidance training, and ingestion of military expended materials.

Acoustic Effects from Sonar, Explosives, and Air Guns

The HCTT proposed action includes a relatively small number of activities involving sonar that could potentially overlap with Southern Resident killer whale critical habitat (see Table 31 above). Testing activities involving air guns could be used up to 57 days per years within the NOCAL RC. Given the very small proportion that critical habitat makes up of the NOCAL RC, we would expect that a very small portion of these activities will overlap with critical habitat. In-water explosives will only be used in a small portion of the NOCAL RC (i.e., Inert Bomb Box within W-283), and there is no overlap between stressors from in-water explosives and Southern Resident killer whale critical habitat.

Southern Resident killer whale prey species may only detect low-frequency sonar systems with higher source levels within a few kilometers; and most other, less powerful low-frequency sonar systems, at much shorter ranges. Fish prey species 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 prey species located closer to the

sonar sound source would likely experience more substantive responses, whereas those located further away from the source are less likely to react to the sound levels. However, because HCTT sonar sources are continuously moving, and prey are also capable of moving away from the disturbance, the overall exposure duration is expected to be brief. 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 Southern Resident killer whale prey species are expected, and the effects described above are not anticipated to lead to a significant disruption of normal behavior patterns. In addition, the area of overlap where fish prey could potentially be exposed represents an extremely small portion of the overall coastal Southern Resident killer whale critical habitat. As such, the effects from sonar and other transducers on the Southern Resident killer whale critical habitat prey PBF are considered insignificant.

Southern Resident killer whale fish prey species may be affected by air gun use in the offshore portion of the action area overlapping designated critical habitat. Adverse effects may include injury, TTS, physiological stress, behavioral reactions, and mortality. Those fish that are killed within the designated critical habitat would no longer be available to killer 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.

Exposure would be highly dependent on the limited number of activities involving air guns that overlap critical habitat and the actual presence of prey species at the time such activities occur. The anticipated impacts from air guns on killer whale fish prey, in terms of either total numbers of animals affected or percent of the population affected, are anticipated to be extremely minor. The use of air guns within the NOCAL RC is very limited, and, as noted above, Southern Resident killer whale critical habitat overlaps with an extremely small proportion of the range complex. The number of fish killed is anticipated to represent an extremely small fraction of the total number of prey items needed per year by an individual killer whale to maintain its energy requirements. 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 populations and the effect on overall quantity, quality and availability of prey items would be insignificant.

Given the large area of the designated critical habitat compared to the relatively small size of the offshore area where air gun shots would occur, the frequency of how often these events would occur, the short duration of these events, and the relatively large number of prey items available throughout the critical habitat, we conclude that any impacts of air guns resulting from HCTT activities on prey availability for the Southern Resident killer whale would be insignificant at the population level.

Given the small area of overlap between activities involving acoustic stressors (i.e., sonar and air guns) and Southern Resident killer whale critical habitat, it is unlikely that sonar or air guns will

have a measureable impact on the species' passage conditions to allow for migration, resting, or foraging. This determination is further supported by NAEMO results (see 6.1.1.1 for details), which predicted no acoustic impacts to Southern resident killer whales from HCTT activities. As noted above, since both the whales and the sonar sources are continuously moving any impacts on passage conditions will be short term and minor, with the individual whales quickly returning to their normal behavioral activities.

Summary

In summary, we find that 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 Resident killer whale 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 Resident killer whale designated critical habitat.

Hawaiian Monk Seal

Hawaiian monk seal critical habitat was originally designated on April 30, 1986 (51 Fed. Reg. 16047) and was extended on May 26, 1988 (53 Fed. Reg. 18988). It originally included all beach areas, sand spits, and islets (including all beach crest vegetation to its deepest extent inland), lagoon waters, inner reef waters, and ocean waters out to a depth of 37 m (121.4 ft) around the northwestern Hawaiian Islands breeding atolls and islands. On September 21, 2015, NMFS published a final rule to revise designated critical habitat for Hawaiian monk seals (80 Fed. Reg. 50925), extending the current designation in the northwestern Hawaiian Islands out to the 200 m (656.2 ft) depth contour (including Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island). The 2015 rule also designated six new areas in the main Hawaiian Islands (i.e., terrestrial and marine habitat from 5 m [16.4 ft] inland from the shoreline extending seaward to the 200 m (656.2 ft) depth contour around Kaula, Niihau, Kauai, Oahu, Maui, Nui, and Hawaii). Only the marine portion of designated Hawaiian monk seal critical habitat, which includes the seafloor plus waters 10 m (33 ft) above seafloor off the Main and Northwest Hawaiian Islands from depths of 80 to 200 m (262.5 to 656.2 ft), occurs within the action area and thus may be affected by the proposed action (Figure 17).

In addition, certain locations of Hawaiian monk seal critical habitat have been excluded from designation under section 4(a)(3) of the ESA because they are managed under military Integrated Natural Resources Plans. These areas in the Hawaiian Islands include: 1) Marine Corps Base Hawaii, Oahu—a 500-yd (91 m) buffer zone in the waters surrounding the base and the Puuloa Training Facility on the Ewa coastal plain, Oahu; 2) Joint Base Pearl Harbor-Hickam, Oahu inclusive of Nimitz Beach, White Plains Beach, Naval Defensive Sea Area, Barbers Point Underwater Range, and Ewa Training Minefield; 3) Pacific Missile Range Facility, Kauai, Offshore Areas plus Kaula Island and the coastal and marine areas to the 33 ft (10-m) isobath surrounding the Island of Niihau; 4) Kingfisher Underwater Training area, off the northeast coast

of Niihau; 5) Puuloa Underwater Training Range off Pearl Harbor, Oahu; and 6) Shallow Water Minefield Sonar Training Range, off the western coast of Kahoolawe in the Maui Nui area.

However, NMFS determined that MCTAB offshore did not warrant exclusion due to potential impacts on national security because the boundaries of this area remain ill-defined and other Federal activities occurring within this area may affect essential features. The area seaward of MCTAB from the seafloor to 10 meters above the seafloor, and from the mean lower low water mark to the 200 m depth contour line was designated critical habitat. MCTAB's terrestrial environment (shoreline) is precluded from critical habitat designation. Due to this unique split designation of critical habitat the action proponents have included specific mitigation related to Amphibious Training Events (see Section 2.4.2) to minimize impacts to Hawaiian monk seals and their habitat resulting from these activities.

The PBFs of Hawaiian monk seal critical habitat that support the species' life history needs include:

1) Terrestrial areas and adjacent shallow, sheltered aquatic areas with characteristics preferred by monk seals for pupping and nursing;

2) Marine areas from 0 to 200 meters (0 to 656.2 feet) in depth that support adequate prey quality and quantity for juvenile and adult Hawaiian monk seal foraging; and

3) Significant areas used by monk seals for hauling out, resting, or molting.

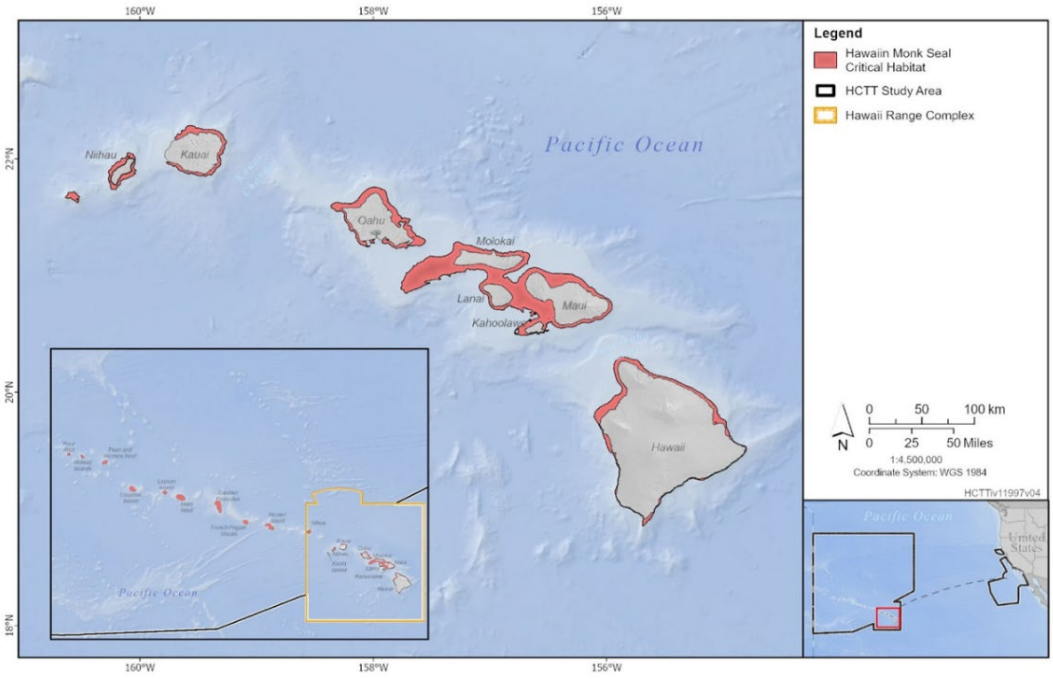


Figure 17. Critical Habitat for Hawaiian Monk Seal in the HCTT Study Area

Stressors Previously Determined NLAA for Fish and Invertebrates

In Section 4.1.1 we determined that several of the stressors associated with the proposed action were not likely to adversely affect ESA-listed fish or benthic invertebrate species because any effects from these stressors were either discountable or insignificant. For the same reasons provided in Section 4.1.1 (see Fish and Invertebrate subsections), we find that the following stressors are not likely to adversely affect the quantity and quality of Hawaiian monk seal benthic and demersal prey (i.e., fish, cephalopods, and crustaceans) within critical habitat: vessel noise, aircraft noise and disturbance, weapons noise, in-water electromagnetic devices, in-water devices, seafloor devices, and military expended materials.

Acoustic Effects from Sonar and Explosives

Similar to our analysis of the effects of HCTT sonar on the prey PBF of humpback whale critical habitat and Southern resident killer whale critical habitat (Section 4.1.3), we expect that any effects of sonar on fish species included in the prey PBF of Hawaiian monk seal critical habitat will be insignificant. Because HCTT sonar sources are continuously moving, and monk seal fish prey are also capable of moving away from the disturbance, the overall exposure duration is expected to be brief. Any physiological stress responses or behavioral reactions would also be expected to be temporary, lasting only a few seconds or minutes during sonar pings. Additionally, as indicated by Popper et al. (2014a), the relative risk of a fish exhibiting a behavioral reaction in response to low-frequency sonar is low, regardless of the distance from the sound source. For these reasons, no long-term consequences for any exposed monk seal fish prey species are expected, and the effects described above are not anticipated to lead to a significant disruption of normal behavior patterns. In addition, the area of overlap where fish prey could potentially be exposed represents an extremely small portion of marine areas from 0 to 200 m in depth that support adequate prey quality and quantity for juvenile and adult Hawaiian monk seal foraging.

Invertebrate prey species could also be exposed to LFA sonar within Hawaiian monk seal critical habitat. Mooney et al. (2016) reported on the results of a behavioral study that showed one species of squid possessed optimal hearing in the range from 200 to 400 Hz, with responses to 80 Hz. Behavioral responses to sound stimuli included escape and predator avoidance behaviors. These responses included inking (which occurred at the lowest sound frequencies and highest sound levels), body color changes, and jetting) (Mooney et al. 2016). Common cuttlefish respond behaviorally to sounds below 1,000 Hz (maximum sensitivities near 150 Hz), with escape responses (inking, jetting) observed between 80 and 300 Hz, sound levels above 140 dB re: 1 µPa (rms), and particle acceleration of 0.01 msec-2; body pattern changes and fin movements were observed at exposures from 80 to 1000 Hz, sound pressure levels of 85 to 188 dB re: 1 µPa (rms), and particle accelerations of 0 to 17.1 msec-2 (Samson et al. 2014). These studies suggest that exposure to low frequency sound, such as HCTT LFA sonar, has limited potential to adversely affect fishes and invertebrates. Available evidence does not suggest low-frequency non-impulsive sound sources would typically be expected to cause mortality or physiological damage in marine invertebrates. Though squid and some other invertebrates appear to exhibit

alarm responses and avoidance of sound sources, individuals would be expected to resume normal behaviors immediately after initial exposure. Therefore, we expect that any effects of HCTT sonar on invertebrate species included in the prey PBF of Hawaiian monk seal critical habitat will be insignificant. Since availability of prey species is the only PBF of Hawaiian monk seal critical habitat that could be impacted by LFA sonar, we conclude that HCTT LFA sonar in the action area is not likely to adversely affect Hawaiian monk seal designated critical habitat.

Proposed activities involving explosives overlap with marine areas designated as critical habitat for juvenile and adult monk seal foraging. These areas are vital for supporting adequate prey quality and quantity, which is a PBF of critical habitat. Hawaiian monk seals prey include various fish and invertebrate species, which could be injured or killed by HCTT explosives. However, the area that would be affected by HCTT explosives is extremely small compared to the vast monk seal foraging areas within critical habitat. Most detonations in nearshore waters occur in established ranges that have either been excluded from critical habitat designation, or are in waters deeper than 200 m, outside the designated area. In addition, the explosives used in nearshore areas are typically of smaller net explosive weight, resulting in lower impact zones for monk seal prey. The Action Proponents will also avoid using explosives within 350 yards of mapped shallow water coral reefs, which are high-density areas for monk seal prey. Although we anticipate reductions in abundance of Hawaiian monk seal prey in designated critical habitat from the use of explosives, these reductions will be highly localized and temporary and would have an insignificant effect (i.e., so minor that the effect cannot be meaningfully evaluated) on prey abundance and quality in marine waters designated as critical habitat. Therefore, the effects of HCTT explosive use on prey resources in Hawaiian monk seal critical habitat are insignificant.

Physical Disturbance in Shallow Water Marine Areas

The HCTT proposed action and subsequent activities are not expected to result in any permanent alterations to or effects on the characteristics of terrestrial areas and nearshore waters. Disturbance to terrestrial and significant areas on beaches and in nearshore waters will be transient. Given the localized and short-term nature of military HCTT activities on terrestrial and nearshore aquatic areas of critical habitat, they are expected to have an insignificant effect on the PBFs of Hawaiian monk seal designated critical habitat. For more details on the effects of physical disturbance in shallow water marine areas see Section 4.1.1: Physical Disturbance and Strike Stressors - Amphibious Landings Events.

Effects of Vessel Movement and Noise

Next, we consider the effects of stressors associated with HCTT vessel activity on the PBFs of monk seal critical habitat described above. In particular, we evaluate the effects of vessels on: PBF #1 - Terrestrial areas and adjacent shallow, sheltered aquatic areas with characteristics preferred by monk seals for pupping and nursing; and PBF #3 - Significant areas used by monk seals for hauling out, resting, or molting.

Vessel movements involve transits to and from ports to various locations within the HRC. HCTT activities involving vessel movements occur intermittently and are variable in duration. Operation of HCTT vessels will result in a temporary increase of vessel traffic within designated critical habitat. The physical transit of Action Proponent vessels through shallow, sheltered aquatic areas within critical habitat will be for a brief period of time. This activity may result in short-term obstruction of surface waters with minor, temporary changes in dissolved oxygen levels, water temperature, and currents. Any effects from vessel movement and disturbance to these shallow, sheltered aquatic areas used for monk seal pupping and nursing will be insignificant.

Transiting vessels also produce a variety of underwater sounds characterized as low-frequency, continuous, or tonal, with sound pressure levels at a source varying according to speed, burden, capacity, and length (Richardson et al. 1995a; Kipple and Gabriele 2007; Mckenna et al. 2012). However, vessel noise will not physically obstruct water passage or affect water properties, depth, sea state, benthic features, or oceanographic features. While HCTT vessel noise may temporarily change the soundscape of terrestrial and adjacent shallow water areas used by monk seals for pupping, nursing, hauling out, resting, or molting, any effects from this stressors will be minor and short-term. An extremely small fraction of Action Proponent vessel activity within HRC will be in shallow water areas within Hawaiian monk seal critical habitat where these PBFs could be exposed to vessel noise. Vessel noise is a transient stressor that would only affect the terrestrial and nearshore PBFs for the short amount of time the vessel is operating in very shallow waters within critical habitat. Therefore, any effects of vessel noise on the PBFs of Hawaiian monk seal critical habitat will be insignificant.

Effects of Seafloor Devices

Anchors used by HCTT vessels may remain on the seafloor for an unspecified period of time and could potentially affect shallow, sheltered aquatic areas with characteristics preferred by Hawaiian monk seals for pupping and nursing, and marine areas from 0 to 200 m in depth that supports adequate prey quality and quantity for juvenile and adult Hawaiian monk seal foraging (PBFs #1 and #2 above). Anchors associated with seafloor devices used during training and testing activities may, in some cases, be left behind and could cover a crack in the seafloor where Hawaiian monk seal prey may have been found. However, a very small area of habitat may be exposed to physical disturbance from anchors (i.e., anchors are generally 14 inches in diameter and a limited number are left behind in Hawaiian monk seal critical habitat). The area of Hawaiian monk seal critical habitat exposed to physical disturbance from anchors will be extremely small relative to the large area designated as critical habitat. Most of the kinetic energy from an anchor dissipates within the first few meters or feet of the object entering the water, causing it to slow considerably by the time it reaches the seafloor. Thus, the anchors are not anticipated to physically disturb or strike prey items of Hawaiian monk seals. For these reasons, the effects of HCTT activities involving anchors on the PBFs of Hawaiian monk seal critical habitat are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Effects of In-water Devices

The presence of cables and remotely operated vehicles may affect the adequate space for movement within the monk seal habitat. These activities may impact the shallow, sheltered aquatic areas preferred for pupping and nursing, as well as marine foraging. While Hawaiian monk seals may temporarily avoid a small area during these activities, the avoidance will be short in duration and localized. Animals in the vicinity will be able to slightly alter course and access preferred habitats nearby. The presence of these devices does not prevent animals from accessing preferred habitat areas. It is extremely unlikely that the presence of cables and remotely operated vehicles will prohibit access to shallow, sheltered aquatic areas for pupping and nursing and marine areas for foraging. Any effects to prey abundance from in-water devices will be short-term and minor, and unaffected prey animals in close proximity will likely move into the area that was disturbed to utilize the unoccupied habitat. If reductions in Hawaiian monk seal prey items (e.g., inshore, benthic, and offshore teleosts, cephalopods, and crustaceans) do occur, they will be highly localized and temporary. Also, relative to the vast area that has been designated as critical habitat, the area that may be affected by HCTT activities is extremely small, and we anticipate a correspondingly small percentage of prey items could be affected. Therefore, the effects of in-water devices on the PBFs of Hawaiian monk seal critical habitat are insignificant.

Summary

In summary, we find that 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 Hawaiian monk seal 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, Hawaiian monk seal designated critical habitat.

Leatherback Sea Turtle

On January 20, 2012, NMFS issued a final rule to designate additional critical habitat for the leatherback sea turtle along the U.S. West Coast (50 CFR 226). This designation includes approximately 43,798 km^2 stretching along the California coast from Point Arena to Point Arguello east of the 3,000 m depth contour; and 64,760 km^2 stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meters depth contour (Figure 18). The designated areas comprise approximately 108,558 km^2 of marine habitat and include waters from the ocean surface down to a maximum depth of 80 meters.

Leatherback sea turtle designated critical habitat overlaps with the action area in the northeast portion of the HCTT (Figure 19). The PBF for leatherback sea turtle critical habitat is 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. Here, we evaluate the effects of the proposed action on this PBF.

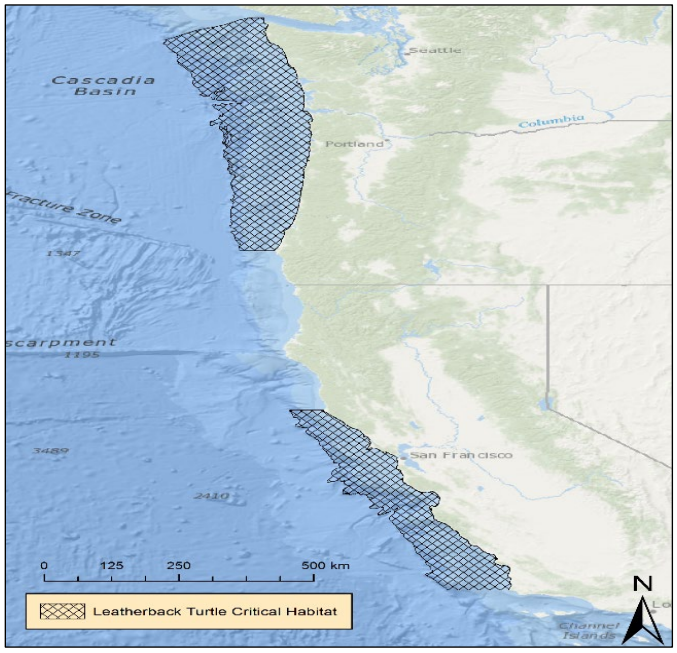


Figure 18. Leatherback Sea Turtle Designated Critical Habitat along the U. S. Pacific Coast

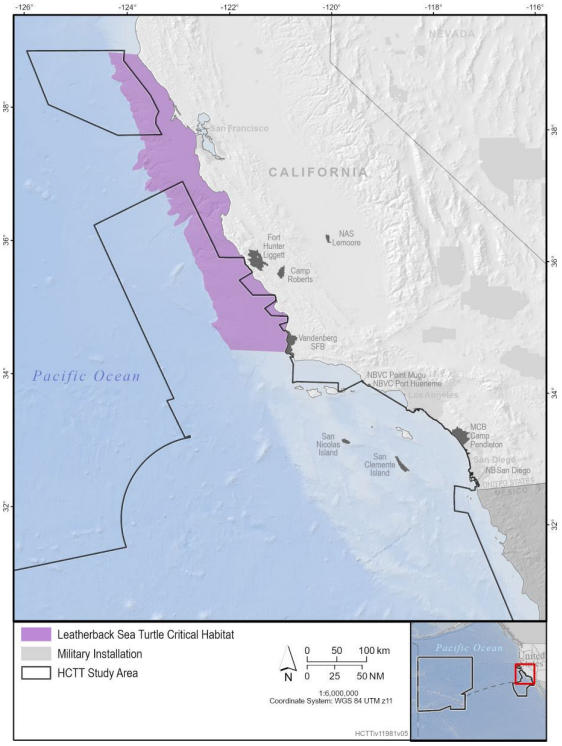


Figure 19. Area of Overlap Between Leatherback Sea Turtle Designated Critical Habitat (purple area) and the HCTT Action Area (outlined in black) (Navy 2024e)

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. Exposure to acoustic stressors (i.e., vessel noise and weapons noise) would likely either have no effect or very minor effects 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. Although some jellyfish species appear to experience acoustic trauma as the result of exposure to low frequency sonar at frequencies between 50 to 400 Hz and at received levels up to 175 dB re: 1 µPa (rms) (Sole et al. 2016), acoustic stressors are not mentioned by the leatherback Critical Habitat Review Team as an activity that may impact the prey essential feature. Thus, we find that any impacts on leatherback critical habitat through effects to prey from HCTT acoustic stressors (i.e., sonar, air guns, and explosives) to be insignificant.

Leatherback prey species could be affected by physical disturbance and strike stressors including vessel strike, military expended materials, and in-water devices (see Section 2.5.4 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, based on our analysis of the pathways to effects from HCTT stressors, we have determined that none of the stressors would significantly affect the prey PBF of leatherback designated critical habitat. Therefore, we conclude that the designated critical habitat for leatherback sea turtles is not likely to be adversely affected by the proposed action.

Green Sturgeon – Southern DPS

On October 9, 2009, NMFS designated final critical habitat for the Southern DPS of green sturgeon. Specific areas include the following: Coastal U.S. marine waters within 60 fathoms (fm) 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 20). Green sturgeon Southern DPS critical habitat includes approximately 515 km (320 mi) of freshwater river habitat, 2,323 km^2 (897 mi^2) of estuarine

habitat, 29,581 km^2 (11,421 mi^2) of marine habitat, 784 km (487 mi) of habitat in the Sacramento-San Joaquin Delta, and 350 km^2 (135 mi^2) of habitat within the Yolo and Sutter bypasses (Sacramento River, CA).

The following areas have been excluded from designation because the economic benefits of exclusion outweigh the benefits of inclusion, and exclusion will not result in the extinction of the species: Coastal U.S. marine waters within 60 fm depth from the California/Mexico border north to Monterey Bay, CA, and from the Alaska/Canada border northwest to the Bering Strait; the lower Columbia River from river kilometer 74 to the Bonneville Dam; and certain coastal bays and estuaries in California (Elkhorn Slough, Tomales Bay, Noyo Harbor, and the estuaries to the head of the tide in the Eel and Klamath/Trinity rivers), Oregon (Tillamook Bay and the estuaries to the head of the tide in the Rogue, Siuslaw, and Alsea rivers), and Washington (Puget Sound). Particular areas are also excluded based on impacts on national security and impacts on Indian lands. The areas excluded from the designation comprise approximately 0.2 km (0.1 mi) of freshwater habitat, 2,945 km^2 (1,137 mi^2) of estuarine habitat and 1,034,935 km^2 (399,590 mi^2) of marine habitat.



Figure 20. Critical Habitat for the Green Sturgeon Southern DPS

As shown in Figure 21, the HCTT action area overlaps a very small portion of the coastal nearshore marine critical habitat within the 60 fm depth contour in the NOCAL RC. The essential PBFs of green sturgeon Southern DPS critical habitat in nearshore coastal marine areas are:

a) Migratory corridor: A migratory pathway necessary for the safe and timely passage of all life stages within marine and between estuarine and marine habitats.

b) Water quality. Coastal marine waters with adequate dissolved oxygen levels and with acceptably low levels of contaminants (such as pesticides, polycyclic aromatic hydrocarbons [PAHs], and elevated levels of heavy metals) that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon.

c) Food resources. Abundant prey items for subadults and adults, which likely include benthic invertebrate and fish.

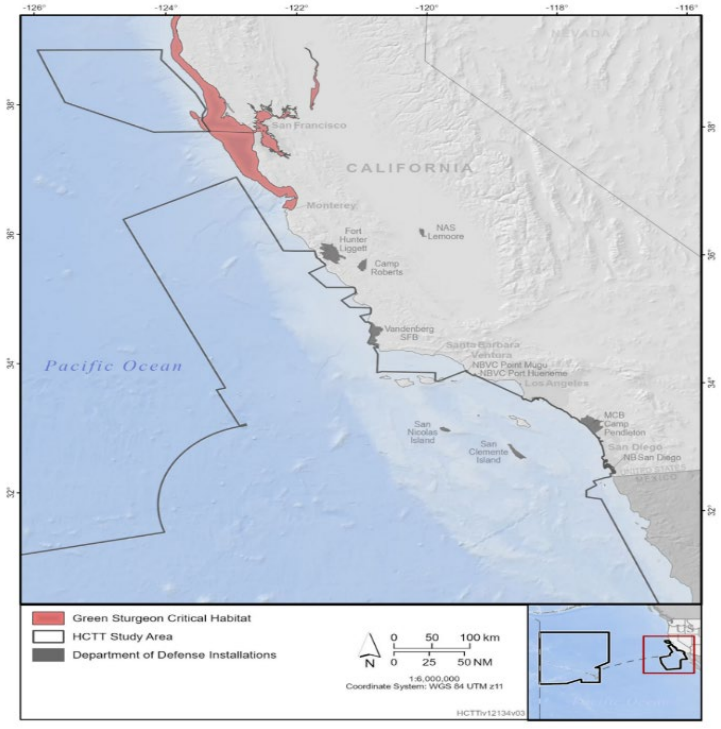


Figure 21. Area of Overlap Between Designated Critical Habitat for the Green Sturgeon Southern DPS and the HCTT Action Area (Navy 2024e)

Stressors Previously Determined NLAA for Fish and Invertebrates

Many of the stressors resulting from HCTT activities would have no effect on green sturgeon food resources, migratory corridor, or water quality within designated critical habitat. As discussed in Section 4.1.1 (Stressors Resulting in Effects to Fish Habitat or Prey of ESA-listed Species), 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 HCTT activities on ESA-listed fish via impacts on water quality were found to be insignificant or discountable, these stressors are not expected to reduce the value of the water quality PBF of green sturgeon coastal marine critical habitat.

In Section 4.1.1 we determined that several of the stressors associated with the proposed action were not likely to adversely affect ESA-listed fish and invertebrate species because any effects from these stressors were either discountable or insignificant. For the same reasons provided in Section 4.1.1 (see Fish and Invertebrate subsections), we find that the following stressors are not likely to adversely affect the benthic prey of green sturgeon critical habitat: vessel noise, weapons noise, in-water electromagnetic devices, in-water devices, seafloor devices, and military expended materials.

Acoustic Stressors

Table 31 (above) shows the HCTT activities with sonar and explosive stressors proposed for use within NOCAL RC. An extremely small portion of these activities could potentially occur within green sturgeon critical habitat given the very small area of overlap. The probability of Southern DPS green sturgeon critical habitat being exposed to sonar, air guns or in-water explosives is also very limited given the internal Navy restrictions on these activities in this portion of the action area. The large majority of these activities are restricted to ocean areas farther offshore (i.e., beyond the 60 fm or 110 m depth contour) than where green sturgeon critical habitat occurs. The only planned NOCAL RC explosives activities with no restrictions on ocean location are Training and End-to-End Mission Capability Verification: Subsea and Seabed Warfare Kinetic Effectors and Unmanned Aerial Vehicle (Table 31). For these activities, all explosives will be detonated at or above the surface. Therefore, a large portion of the energy from these detonations will be reduced before entering the water column. Since green sturgeon typically feed on benthic prey on the seafloor, the prey PBF of critical habitat would be less likely to be exposed to detonations at or above the surface. In addition, the explosive munitions used during these activities are considered relatively small (E3 > 0.5–2.5 lb. NEW), and only a small number of explosive detonations will be used (14 annually in a maximum year of training). All HCTT activities involving in-water explosives within NOCAL RC are restricted to ocean areas more than 12 nm from shore or depths greater than 100 m. For example, Acoustic & Oceanographic Research with relatively small NEW (bin E1 and E3 in-water explosives) will be limited to ocean areas greater than 12 nm from shore, and Submarine and UUV Subsea and Seabed Warfare exercises (bin E3 in-water explosives) will only occur in water depths exceeding 100 m. Given the anticipate low levels of exposure of green sturgeon critical habitat to HCTT acoustic stressors (i.e., sonar and explosives), any effects of such exposure to the migratory corridor or food resources PBFs of critical habitat will be insignificant. If green sturgeon prey items are

killed or injured within critical habitat, it is likely that only a very small 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. Thus, acoustic stressors (i.e., sonar, air guns, and explosives) are not expected to reduce the value of the water quality or food resources PBF of green sturgeon coastal marine critical habitat.

Summary

In summary, we find that 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.

Black Abalone

Critical habitat was designated for black abalone on October 27, 2011 (76 Fed. Reg. 66805). Most of the designated critical habitat lies along the California coast north of the action area (Figure 22). Designated critical habitat includes rocky intertidal and subtidal habitats from the mean high water line to a depth of approximately 20 ft (6 m), as well as the waters encompassed by these areas. Designated critical habitat extends from Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula.

Within the action area, critical habitat occurs on Santa Catalina and Santa Barbara Islands. The specific areas proposed for designation off San Nicolas and San Clemente Islands were determined to be ineligible for designation because the Navy's Integrated Natural Resources Management Plans (INRMPs) provide benefits to black abalone in those areas (see below for discussion of INRMPs). The critical habitat designation identified PBFs (formerly referred to as primary constituent elements), which are habitat elements essential for the conservation of the species. The PBFs for black abalone are rocky substrate, food resources, juvenile settlement habitat, suitable water quality, and suitable nearshore circulation patterns.

The current distribution of black abalone ranges approximately from Point Arena in northern California to Bahia Tortugas and Isla Guadalupe in Mexico (Butler et al. 2009). Although the geographic range of black abalone extends to northern California, the most abundant populations historically have occurred in the Channel Islands (Butler et al. 2009).

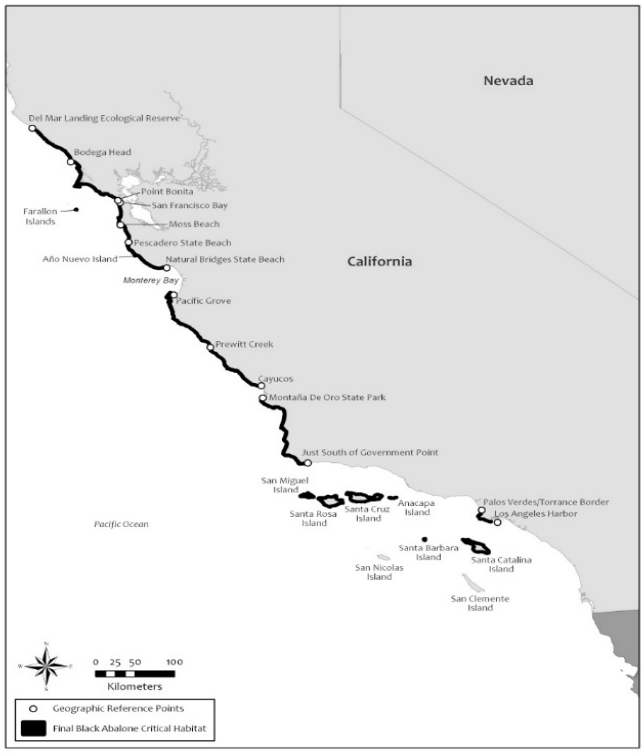


Figure 22. Designated Critical Habitat for Black Abalone

Military Areas Ineligible for Designation

Recent amendments to the ESA preclude the Secretary from designating military lands as critical habitat if those lands are subject to an INRMP under the Sikes Act and the Secretary certifies in writing that the plan benefits the listed species (section 4(a)(3)(B) of the ESA). The INRMPs provide guidelines, goals, and objectives for species management in an effort to recover, improve, or restore natural resources while ensuring continued support of the military mission. Prior to publication of the proposed rule, NMFS contacted the Department of Defense (DOD) and requested information on all INRMPs for DOD facilities that overlap with the specific areas being considered for designation and that might provide benefits to black abalone. In response to this request, the DOD identified four installations owned or controlled by the U.S. Navy for which INRMPs have been prepared and that may occur within the specific areas being considered by NMFS for designation as black abalone critical habitat. These installations include: San Clemente Island; San Nicolas Island; Naval Weapons Station Seal Beach; and Naval Base Ventura County (Point Mugu and Port Hueneme).

NMFS determined that the Naval Weapons Station Seal Beach and Naval Base Ventura County installations do not overlap with the specific areas being considered for designation, but that the San Clemente Island and San Nicolas Island installations do occur within the specific areas being considered for designation. Because black abalone were not listed under the ESA when the current INRMPs were developed, the INRMPs do not include specific protections for black

abalone, although they do include protections for other rocky intertidal species that may also benefit black abalone. The Navy recently revised and amended the INRMPs for these two installations to include specific management actions and conservation benefits for black abalone, with input from NMFS. As described below, NMFS determined that the revised San Nicolas Island INRMP and the amended San Clemente Island INRMP provide benefits to black abalone. Therefore, NMFS determine under section 4(a)(3)(B) of the ESA that San Nicolas Island and San Clemente Island are no longer eligible for designation as critical habitat.

Analysis of Effects to Black Abalone Critical Habitat

The stressors associated with the proposed action that could potentially affect black abalone critical habitat include explosives, entanglement stressors, physical disturbance and strike stressors, and disturbance from seafloor devices.

Acoustic Effects from Sonar and Explosives

Roberts et al. (2016) and Edmonds et al. (2016) report that marine invertebrates are generally not directly affected by sounds from explosives, but likely have mechanical receptors that may be connected to the central nervous system that can detect some movements or vibrations that are transmitted through substrate. Black abalone would not likely be exposed to surface explosions and associated underwater impulsive sounds from high-explosive munitions (including bombs and missiles). Black abalone critical habitat in the action area is generally not in areas where the Action Proponents test or train with explosives. This species is not found in offshore areas where ordnance typically would be used, which is most frequently greater than 12 nm from shore and in water depths greater than 200 ft. Because the number of detonations for proposed testing and training activities under the proposed action is very small, and testing and training activities would take place in areas where this species does not occur, or is exempted from critical habitat designations (see above), the probability of this critical habitat being exposed to explosive detonation effects is extremely low (i.e., extremely unlikely).

Effects of explosives and unexploded ordnance on black abalone via sediment are possible near the ordnance. However, the relatively low solubility of most explosives and their degradation products indicate that concentrations of these byproducts in the marine environment are relatively low and readily diluted. Because most ordnance is deployed as projectiles, multiple unexploded or low-order detonations would accumulate on spatial scales of 1 to 6 ft (0.3 to 1.8 m); therefore, potential impacts are likely to remain local and widely separated. Given these conditions, the possibility of effects to black abalone is extremely unlikely and we consider the effects from this stressor on black abalone to be discountable.

Therefore, we consider the effects from HCTT explosives on black abalone critical habitat to be discountable.

Effects of Entanglement

Parachutes and decelerators used as part of the proposed action could potentially overlap areas of black abalone critical habitat. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with these materials have the potential to result in modification of critical habitat. Large and extra-large decelerators and parachutes may pose a higher degree of risk for black abalone because these parachutes have long lines (large chutes have 28 cords, approximately 40 to 70 ft long; extra-large parachutes have 64 cords, up to 82 ft long) associated with them. However, the chance of an encounter is remote given the small number of large and extra-large decelerators parachutes proposed to be deployed that would not be recovered. Given the vast area over which any one of these decelerators and parachutes would be deployed and the limited number of non-recovered items annually, the chances of a black abalone critical habitat being modified is extremely unlikely (i.e., discountable).

Physical Disturbance in Shallow Water Marine Areas

Physical disturbance or strikes by vessels, military expended materials, and in-water devices on black abalone critical habitat is possible at the seafloor. However, disturbance or strike impacts from vessels are extremely unlikely because the monitoring locations of the black abalone habitat around San Nicolas Island are located in shallow water and are known to the Navy. In these and other areas, impacts from vessels would only occur in the unlikely event that a Navy vessel ran aground. Disturbance or strike impacts on critical habitat by military expended materials and in-water devices are extremely unlikely because these materials generally sink rapidly enough to deeper waters outside of black abalone critical habitat and the chances that military expended material would fall into specific cracks and crevices where black abalone are found are extremely low. These materials would also not be expected to affect black abalone because of the limited amount of items that would be expended in water depths less than 20 ft. It is conceivable for an item expended offshore to drift shoreward and reach water depths where black abalone critical habitat may be present. The majority of military expended material in nearshore and offshore waters surrounding Tanner Bank is chaff and flares, which have a small potential for impacts. The probability of physical disturbance or strikes from military expended materials affecting black abalone critical habitat is very low (i.e., extremely unlikely) because the predominant use of these materials is outside of black abalone habitat. Therefore, we consider the effects from HCTT physical disturbance or strikes by vessels, military expended materials, and in-water devices on the black abalone critical habitat to be discountable.

Effects of Seafloor Devices on Critical Habitat

Metals are introduced into seawater and sediments as a result of testing and training activities involving vessel hulks, targets, ordnance, munitions, and other military expended materials. Effects of metals on black abalone via sediment and water involve concentrations several orders of magnitude lower than concentrations achieved via bioaccumulation. Black abalone may be exposed by contact with the metal, contact with trace amounts in the sediments or water, and ingestion of sediments. Ingested metals are toxic at substantially lower effective concentrations

than metals dissolved or suspended in the water. Given the small size of black abalone compared to most military expended materials, direct ingestion of metals is unlikely. Because metals often concentrate in sediments, potential adverse effects are much more likely via sediment than via water. However, black abalone do not feed directly on sediment and sediments would have to be relatively fine in order to be consumed. Metal fragments will likely be too large to be ingested and are therefore unlikely to cause injury or mortality to black abalone. As a threat to black abalone, heavy metals are considered to be "low" (VanBlaricom et al. 2009). For these reasons, we consider effects from metals to black abalone to be extremely unlikely and thus insignificant.

Several Navy testing and training activities introduce potentially harmful chemicals into the marine environment; principally, flares and propellants from rockets, missiles, and torpedoes. The greatest risk to black abalone from flares, missiles, and rocket propellants is perchlorate, which is highly soluble in water, persists in the environment, and is known to impact metabolic processes in many plants and animals. Lethal potassium perchlorate concentration 50 values of 72, 5, and 56 millimoles per liter have been obtained for marine algae, zooplankton, and earthworms respectively (Acevedo-Barrios et al. 2018). Black abalone may be exposed by direct contact with a chemical found in the sediments or water or through accidental ingestion of sediments containing trace amounts of a chemical. For perchlorate, these pathways are limited given that rapid dilution within the water column would be expected and missile and rocket propellant is mostly, if not completely, expended before the munition enters the water. Military expended materials will also be used predominantly outside of black abalone habitat. Additionally, perchlorate does not readily absorb into sediments. The principal toxic components of torpedo fuel, propylene glycol dinitrate and nitrodiphenylamine, do readily absorb into sediments but are readily degraded by physical and biological processes. For these reasons, we consider effects from chemicals to black abalone critical habitat to be extremely unlikely.

In summary, we find that 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 black abalone 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 black abalone designated critical habitat.

4.2 Status of the Species and Critical Habitat Likely to be Adversely Affected

The remainder of this opinion examines the status of each species and critical habitat that is likely to be adversely affected by the proposed action. The status is an assessment of the abundance, recent trends in abundance, survival rates, life stages present, limiting factors, and sub-lethal or indirect changes in population trends such as inter-breeding period, shifts in distribution or habitat use, and shifts in predator distribution that contribute to the extinction risk that the listed species face. The status of each species below is described in terms of life history, threats, population dynamics, and recovery planning. The status of each critical habitat is described in terms of the PBFs essential to the conservation of the species; the status, function,

and extent of those PBFs based on best available scientific and commercial data; and the conservation needs of the species in terms of habitat to support a recovered population.

The information used for each species is based on parameters considered in documents such as status reviews, recovery plans, and listing decisions, and based on the best available scientific and commercial information. This section establishes a reference to the condition of the species' likelihood of both survival and recovery in terms of "reproduction, numbers, or distribution" as described in 50 CFR §402.02. This section also examines the condition of critical habitat throughout the species' range, evaluates the conservation value of the various components of the habitat (e.g., watersheds, ocean basins and coastal and marine environments) that make up the designated area, and discusses the function of the essential PBFs that help to form that conservation value. 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 the NMFS OPR web site (https://www.fisheries.noaa.gov/species-directory/threatened-endangered).

4.2.1 Blue Whale

The blue whale is a widely distributed baleen whale found in all major oceans (Figure 23). The blue whale consists of five currently recognized subspecies (NMFS 2020d): *B. m. musculus* is the northern blue whale (North Atlantic and North Pacific Oceans); *B. m. intermedia* is the Antarctic blue whale; *B. m. brevicauda* is colloquially known as the "pygmy" blue whale; and *B. m. indica* is the northern Indian Ocean blue whale. The 2020 blue whale recovery plan identified the following nine management units: North Atlantic; Eastern North Pacific; Western/Central North Pacific; Northern Indian Ocean; Madagascar; Western Australia/Indonesia; New Zealand; Chilean; and Antarctic.

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 kg (7,936.6 lbs.) daily. Feeding aggregations are often found at the continental shelf edge, where upwelling produces concentrations of krill at depths of 90 to 120 m (295.3 to 393.7 ft).

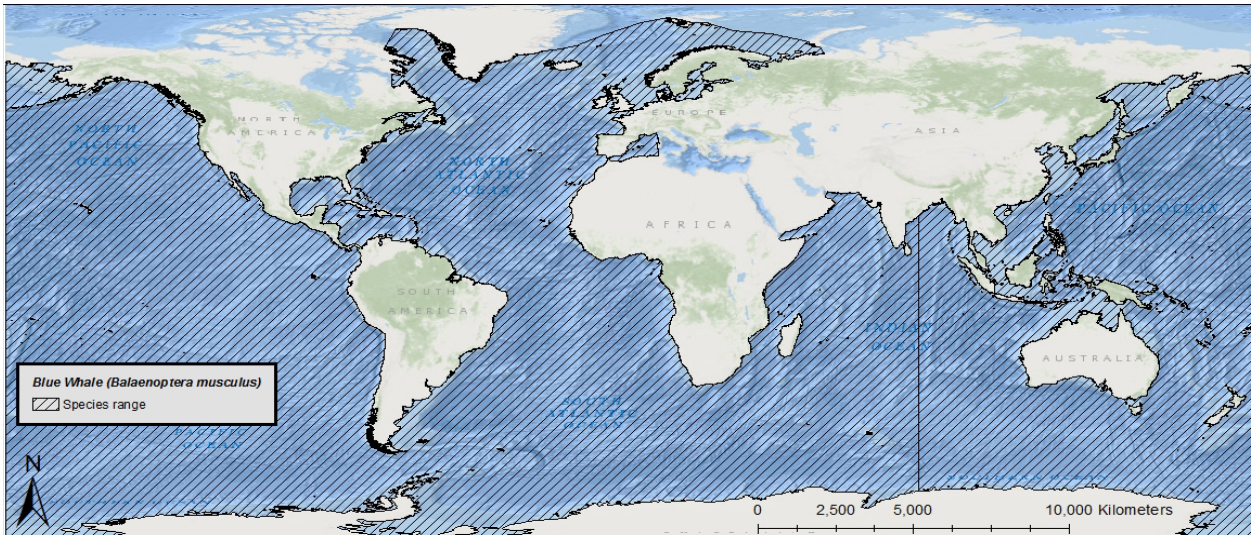


Figure 23. Map Identifying the Range of the Endangered Blue Whale

Blue whales spend more than 94% of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5 to 20 times at 12 to 20 second intervals before a deep dive of 3 to 30 minutes (Leatherwood et al. 1976; Maser et al. 1981; Yochem and Leatherwood 1985; Croll et al. 1999). Average foraging dives are 140 m deep and last for 7.8 minutes (Croll et al. 2001a). Non-foraging dives are shallower and shorter, averaging 68 m and 4.9 minutes (Croll et al. 2001a). However, dives of up to 300 m are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 m).

Blue whales occur singly or in groups of two or three (Ruud 1956; Slijper 1962; Nemoto 1964; Mackintosh 1965; Pike and Macaskie 1969; Aguayo 1974). However, larger foraging aggregations, even with other species such as fin whales, are regularly reported (Schoenherr 1991; Fiedler et al. 1998). Little is known of the mating behavior of blue whales.

Vocalizations and Hearing

Blue whale vocalizations tend to be 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 (McDonald et al. 1995; Ketten 1998; McDonald et al. 2001; Mellinger and Clark 2003). As with other baleen whales, blue whale vocalization function is unknown, although numerous hypotheses exist including maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources (Payne and Webb. 1971; Thompson et al. 1992; Edds-Walton 1997; Oleson et al. 2007b).

Vocalizations are predominantly songs and calls. 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 (Payne and Mcvay 1971; Mellinger and Clark 2003). 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). Blue whale 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 (80 to 20 Hz), with seasonally variable occurrence. Blue whale calls have high acoustic energy, with reports of source levels ranging from 180 to 195 dB re: 1 µPa at 1 m (Cummings and Thompson 1971b; Aburto et al. 1997; Ketten 1998; McDonald et al. 2001; Clark and Gagnon 2004; Berchok et al. 2006; Samaran et al. 2010). Calling rates of blue whales tend to vary based on feeding behavior. Stafford et al. (2005) recorded the highest calling rates when blue whale prey was closest to the surface during its vertical migration. Wiggins et al. (2005) reported 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 (less than 30 m whales), while deeper diving whales (greater than 50 m) were likely feeding and calling less.

Although general characteristics of blue whale calls are shared in distinct regions (Thompson et al. 1996; McDonald et al. 2001; Mellinger and Clark 2003; Rankin et al. 2005), some variability appears to exist among different geographic areas (Rivers 1997). Blue whale 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 (Mellinger and Clark 2003; Berchok et al. 2006; Samaran et al. 2010).

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 (Richardson et al. 1995b; Ketten 1997). Based on vocalizations and anatomy, blue whales are assumed to predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Stafford and Moore 2005; Oleson et al. 2007c). In terms of functional hearing capability, blue whales are included in the low frequency[12] cetacean group, which have a hearing range of 7 Hz to 36 kHz (NMFS 2024g).

Population Dynamics

The global, pre-exploitation estimate for blue whales is approximately 181,200 (IWC 2007). The current blue whale global mature population size is uncertain, but estimated to be in the range of 5,000-15,000 mature individuals (NMFS 2020d). Calambokidis and Barlow (2020) reported the best estimate of current blue whale abundance for CA/OR/WA waters, based on the most-recent 4 years (2015-2018) of capture-recapture data, is 1,898 whales. The minimum population estimate of blue whales, calculated as the lower 20th percentile of the 2018 mark-recapture estimate, is 1,767 whales (Carretta 2022). Based on mark-recapture estimates, there may be evidence of a population size increase since the 1990s, but a formal trend analysis is lacking and the current population trend is unknown (Carretta 2022). Results from a population dynamics model suggest that density dependence, and not vessel strike impacts, explains the observed lack of a population size increase of the Eastern North Pacific Ocean feeding stock (Monnahan et al.

[12] As discussed below (Section 6.1.1.3.2), for the Phase IV NAEMO auditory impacts analysis blue whales are included in the very-low frequency cetacean hearing group.

2014). A 2010 shipboard line-transect survey of the entire Hawaiian Islands Exclusive Economic Zone (EEZ) resulted in a summer/fall abundance estimate of 133 (Coefficient of Variation [CV] = 1.09) blue whales (Bradford et al. 2017). This is currently the best available abundance estimate for the stock within the Hawaii EEZ, although this may be a minimum estimate since the majority of blue whales would be expected to be at higher latitude feeding grounds at this time of year (Carretta 2021b). Current estimates indicate a growth rate of around 3-4% per year for the eastern North Pacific stock (Calambokidis et al. 2009b; Carretta 2021b).

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 U.S. waters: the Eastern North Pacific Ocean, Central North Pacific Ocean, and Western North Atlantic Ocean. Current estimates indicate a population growth rate of around 3-4% per year for the eastern North Pacific stock (Calambokidis et al. 2009b; Carretta 2021b). 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.

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.

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 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.

The U.S. west coast is known to be a feeding area for blue whales during summer and fall (Calambokidis et al. 2009a; Bailey et al. 2010; Calambokidis et al. 2015; Mate et al. 2015; Abrahms et al. 2019), although primary occurrence for this species is south of 44 degrees North (Hamilton et al. 2009; Forney et al. 2012; 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.

Ryan et al. (2022) found that blue whales in the Monterey Bay National Marine Sanctuary track the coastal upwelling circulation in which their most essential resource, aggregated krill, occurs. Szesciorka et al. (2020) investigated Southern California blue whale migration timing, environmental conditions, and prey concentrations. They found that blue whales are arriving up to one month earlier but not departing earlier leading to longer residency times off of Southern California. Based on acoustic call detections over 10-years (2008-2017), blue whales arrived in May and departed in November, remaining at the feeding grounds an average of 8.4 months

(Szesciorka et al. 2020). Blue whales vary the timing of their departure from foraging habitat by up to four months from year to year, potentially as a strategy to maximize energy gain before the breeding season (Oestreich et al. 2020; Oestreich et al. 2022).

Blevins et al. (2022) reported sex- and age-specific migratory strategies of blue whales in the northeast Pacific Ocean, based on carbon and nitrogen isotope values measured along the length of baleen plates collected from 13 whales. Adult females exhibited relatively stable seasonal movements between temperate latitude foraging grounds and subtropical breeding grounds, although two skipped migration one year and subsequently moved to the same subtropical breeding ground near the Costa Rica Dome. Adult males exhibited two movement strategies with most remaining at temperate latitudes for 3-4 years before death, while two migrated to subtropical breeding grounds. Movement patterns in juvenile blue whales were more erratic (Blevins et al. 2022).

Status

Due largely to commercial whaling during the 19th and 20th centuries, blue whale populations declined, with over 380,000 blue whales taken between 1868 and 1978, predominantly from Antarctic waters (NMFS 2020d). In the North Atlantic Ocean, at least 11,000 blue whales were harvested from the late 19th to mid-20th centuries. The global mature population size in 1926 was around 140,000. The current global population size is estimated to be in the range of 5,000-15,000 mature individuals (NMFS 2020d). This corresponds to a reduction of 89%-97% compared to the 1926 global population estimate (Cooke et al. 2019). Commercial whaling no longer occurs, but blue whales are threatened by vessel strikes, entanglement in marine debris and fishing gear, pollution, harassment due to whale watching, anthropogenic noise, and reduced prey abundance and habitat degradation due to changing environmental conditions and ecosystem changes. Although still depleted compared to historical abundance, blue whale populations around the world show signs of growth (NMFS 2020d). This suggests that the species appears to be somewhat resilient to current threats.

Critical Habitat

No critical habitat has been designated for the blue whale.

Recovery Planning

In response to the current threats facing the species, NMFS developed goals and objectives to recover blue whale populations. The two main recovery objectives from the 1998 blue whale recovery plan (NMFS 1998) were to: 1) increase blue whale resiliency and ensure geographic and ecological representation by achieving sufficient and viable populations in all ocean basins and in each recognized subspecies, and 2) increase blue whale resiliency by managing or eliminating significant anthropogenic threats. Threats to blue whales are discussed in further detail in the environmental baseline section of this opinion (Section 5).

In 2020, NMFS published the first revision to the 1998 plan (NMFS 2020d). Key elements of the recovery strategy laid out in the revised plan include: 1) maintaining the international ban on

commercial hunting, 2) improving our understanding of how potential threats may be limiting blue whale recovery and implementing actions where populations may be vulnerable, and 3) outlining a research strategy to obtain data necessary to inform estimation of population abundance and trends. Once the populations and their threats are more fully understood, the blue whale recovery plan will be modified to include actions to minimize any threats that are determined to be limiting recovery.

The 2020 revision to the blue whale recovery plan includes specific down listing/delisting criteria for each of the nine management units. Recovery of all nine management units is important for achieving geographic and ecological representation of blue whales in the world's oceans, and to ensure conservation of the breadth of genetic variability (NMFS 1998). One criterion for all units is that anthropogenic threats have been identified and demonstrably minimized; i.e., there is information indicating they are not contributing to the species being in danger of extinction (within the foreseeable future, for delisting) throughout all or a significant portion of its range. The following specific downlisting/delisting criteria are specified for the Eastern North Pacific management unit, which occurs within the action area: minimum abundance of 2,000 for downlisting and 2,500 for delisting; stable or increasing abundance for downlisting and increasing abundance trend for delisting (over the most recent 30-year period assessed) (NMFS 2020d).

4.2.2 Fin Whale

The fin whale was originally listed as endangered on December 2, 1970. The fin whale is a large, widely distributed baleen whale found in all major oceans and comprised of three subspecies: *B. p. physalus* in the Northern Hemisphere, and *B. p. quoyi* and *B. p. patachonica* (a pygmy form) in the Southern Hemisphere (Figure 24).

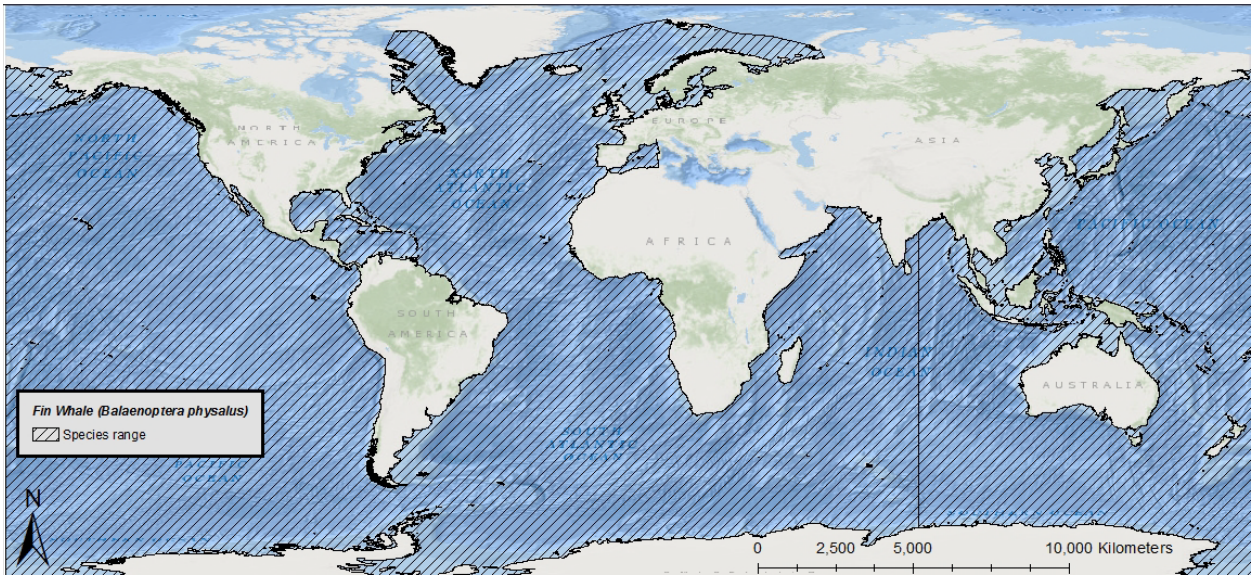


Figure 24. Map Identifying the Range of the Endangered Fin Whale

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. Sexual maturity is reached between six and ten years of age with an average calving interval of two to three years. The generation time may be as long as 25.9 years (NMFS 2019e). 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). Fin whale 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 lance. 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 (Soule and Wilcock 2013; Muto et al. 2019).

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-second duration, followed by a deep dive of 1.5 to 15 minutes (Gambell 1985; Stone et al. 1992; Lafortuna et al. 2003). Other authors have reported that the fin whale's most common dives last 2 to 6 minutes (Watkins 1981c; Hain et al. 1992). Croll et al. (2001a) reported average fin whale dives of 98 m and 6.3 minutes for foraging fin whales, while non-foraging dives are 59 m and 4.2 minutes. Lafortuna et al. (1999) found that foraging fin whales have a higher blow rate than when traveling. Foraging dives in excess of 150 m are known (Panigada et al. 1999). Keen et al. (2019) documented fin whale diel occurrence within the California Bight with fin whales having a strong diel dive pattern behavior including remaining near the surface more at night.

Individuals or groups of less than five individuals represent about 90% of the observations. Barlow (2003) reported mean group sizes of 1.1 to 4.0 during surveys off California, Oregon, and Washington.

Vocalizations and Hearing

Fin whales produce a variety of low frequency sounds in the 10 to 200 Hz range (Watkins 1981b; Watkins et al. 1987; Edds 1988; Thompson et al. 1992). Typical vocalizations are long, patterned pulses of short duration (0.5 to 2 seconds) in the 18 to 35 Hz range, but only males are known to produce these (Patterson and Hamilton 1964; Clark et al. 2002). The most typically recorded call is a 20 Hz pulse lasting about one second, and reaching source levels of 189 ±4 dB re: 1 µPa at 1 m (Watkins 1981b; Watkins et al. 1987; Edds 1988; Richardson et al. 1995b; Charif et al. 2002; Clark et al. 2002; Sirovic et al. 2007). 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 (Clark and Charif 1998). Richardson et al. (1995b) reported this call occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns in winter. The seasonality and stereotype nature of these vocal sequences suggest that they are male reproductive displays (Watkins 1981b; Watkins et al. 1987); a notion further supported by data linking these vocalizations to male fin whales only (Croll et al. 2002). Some researchers have also recorded moans of 14 to 118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34 to 150 Hz, and songs of 17 to 25 Hz (Watkins 1981b; Edds 1988; Cummings and Thompson 1994). In general, source levels for fin whale vocalizations are 140 to 200 dB re: 1 µPa at 1 m (as compiled by Erbe 2002; see also Clark and Gagnon 2004).

Although their function is still in doubt, low frequency fin whale vocalizations travel over long distances and may aid in long distance communication (Payne and Webb. 1971; Edds-Walton 1997). 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 humpback whales (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999). Also, it has been suggested that some fin whale 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).

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 (Richardson et al. 1995b; Ketten 1997). 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 Hz and 12 kHz and a maximum sensitivity to sounds in the 1 to 2 kHz range. In terms of functional hearing capability, fin whales belong to the low-frequency[13] group, which have a hearing range of 7 Hz to 36 kHz (NMFS 2024g).

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 fin whale.

The pre-exploitation estimate for the fin whale population in the North Pacific Ocean was 42,000 to 45,000 (Ohsumi and Wada 1974). There are three fin whale stocks in U.S. Pacific Ocean waters: Northeast Pacific – estimated minimum of 1,368 individuals; Hawaii - approximately 203 individuals, estimated minimum of 101; and California/Oregon/Washington - approximately 11,065 individuals, estimated minimum of 7,970 (Becker et al. 2020; Bradford et al. 2021; Carretta 2022).

The Northeast Pacific stock has an estimated annual growth rate of 4.8% and the California/Oregon/Washington stock has a stable population abundance (Nadeem et al. 2016).

[13] As discussed below (Section 6.1.1), for the Phase IV NAEMO auditory impacts analysis fin whales are included in the very-low frequency cetacean hearing group.

Indications of recovery in California coastal waters date back to 1979 (Barlow 1994), but there is now strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 based on analysis of line transect surveys (Nadeem et al. 2016; Becker et al. 2020; Carretta 2022). Estimated annual rates of increase in the California Current averaged 7.5% from 1991 to 2014 (Nadeem et al. 2016). However, it is unknown how much of this growth is due to immigration rather than birth and death processes (Carretta 2022).

Three subspecies of fin whale are currently recognized: *B. p. physalus* in the Northern Hemisphere, and *B. p. quoyi* and *B. p. patachonica* (a pygmy form) in the Southern Hemisphere. Archer et al. (2013) examined the genetic structure and diversity of fin whales globally. Full sequencing of the mitochondrial 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 fin whales appear to be more closely related to the Southern Hemisphere population, as compared to fin whales in the North Pacific Ocean. Genetic, acoustic, and isotope analyses indicate there may be some population differentiation within the North Atlantic, and there is acknowledged differentiation between the North Atlantic and Mediterranean. Generally speaking, haplotype diversity was found to be high both within and across ocean basins. 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.

Fin whales are distributed widely in every ocean except the Arctic Ocean, occurring primarily in the North Atlantic Ocean, North Pacific Ocean, and Southern Hemisphere where they appear to be reproductively isolated. Fin whales are highly migratory and exhibit complex movement patterns. Movements can be either inshore/offshore or north/south (NMFS 2019e). Fin whales occur year-round in a wide range of latitudes and longitudes, but the density of individuals in any one area changes seasonally. Movements of individuals in a given year may vary according to their energetic and reproductive condition, climatic variables, and other factors. The availability of prey, sand lance in particular, is thought to have had a strong influence on the distribution and movements of fin whales.

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 1985). The overall distribution may be based on prey availability. Falcone et al. (2022) used sighting histories of 932 individuals photographed in the California Current System from 1987 to 2018 to investigate fin whale movements and residency patterns. Their results suggest that the Southern California Bight is used seasonally by fin whales from the larger California Current System stock but is also

home to a smaller, year-round resident subpopulation, which primarily utilizes the nearshore waters.

Status

The fin whale is endangered as a result of past commercial whaling. Fin whale populations declined, due largely from direct harvest, by more than 70% from 1929-2007 (NMFS 2019e). A small number of fin whales may be killed under "aboriginal subsistence whaling" in Greenland, under Japan's scientific whaling program, and Iceland's formal objection to the International Whaling Commission's ban on commercial whaling. Besides direct harvest, all other threats in the fin whale recovery plan were 'low', 'unknown', or 'unknown, but potentially high' in terms of relative impact to recovery (NMFS 2010b). The two threats that were categorized as 'unknown, but potentially high' were ship strikes and loss of prey base due to changing environmental conditions and ecosystem change or shifts in habitat. Estimated vessel strike mortality for the California/Oregon/Washington stock is 43 whales annually, or approximately 0.4% of the estimated population size (Carretta 2022).

Overall species abundance is in the tens of thousands distributed across major ocean basins and hemispheres. This level of abundance and extent of distribution indicates that fin whales have a low probability of experiencing the deleterious effects of small population size such as depensatory processes and random biological and/or environmental variation (NMFS 2019e). The increasing trend for the large central North Atlantic population and increasing populations in the North Pacific indicate that reproduction is exceeding mortality (NMFS 2019e). The species' overall large population size may provide some resilience to current threats, but trends are largely unknown. NMFS is currently considering downlisting the fin whale from endangered to threatened status based on the latest 5-year status review (NMFS 2019e) and information since that review was published, and the fin whale recovery plan criteria.

Critical Habitat

No critical habitat has been designated for the fin whale.

Recovery Planning

In response to the current threats facing the species, NMFS developed goals to recover fin whale populations. These threats will be discussed in further detail in the Environmental Baseline section of this consultation. See the 2010 Final Recovery Plan for the fin whale for complete downlisting/delisting criteria for both of the following recovery goals:

1. Achieve sufficient and viable population in all ocean basins.
2. Ensure significant threats are addressed.

Threats to fin whales are discussed in further detail in the environmental baseline section of this opinion (Section 5).

Downlisting criteria for fin whales include:

- Factors that may limit population growth (i.e., those that are identified as either high, medium, or unknown impact to recovery) have been identified and are being or have been addressed to the extent that they allow for continued growth of populations in each ocean basin in which it occurs (North Atlantic, North Pacific and Southern Hemisphere).
- The population in each ocean basin satisfies the risk analysis standard for threatened status (has no more than a 1% chance of extinction in 100 years) and has at least 500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males) (NMFS 2010b).

4.2.3 Gray Whale – Western North Pacific Population

The gray whale was originally listed as endangered on December 2, 1970 under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained listed after the passage of the ESA in 1973. The Eastern North Pacific stock was officially delisted on June 16, 1994 when it reached pre-exploitation numbers. The Western North Pacific population of gray whales remains listed as endangered.

The gray whale is a baleen whale and the only species in the family Eschrichtiidae. There are two isolated geographic distributions of gray whales in the North Pacific Ocean: the Eastern North Pacific stock, found along the west coast of North America, and the Western North Pacific or "Korean" stock, found along the coast of eastern Asia (Figure 25).

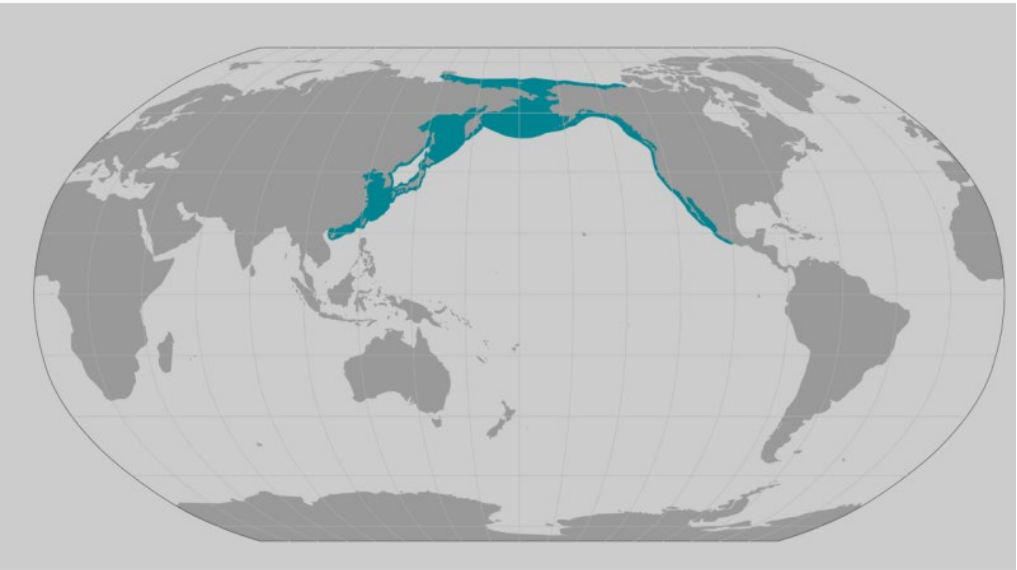


Figure 25. Map Identifying the Range of the Gray Whale

Life History

The average life span of gray whales is unknown but it is thought to be as long as 80 years. They have a gestation period of twelve to thirteen months, and calves nurse for seven to eight months. Sexual maturity is reached between six and 12 years of age with an average calving interval of

two to four years (Weller et al. 2009). Gray whales mostly inhabit shallow coastal waters in the North Pacific Ocean. Some Western North Pacific gray whales winter on the west coast of North America while others migrate south to winter in waters off Japan and China, and summer in the Okhotsk Sea off northeast Sakhalin Island, Russia, and off southeastern Kamchatka in the Bering Sea (Burdin et al. 2013). Gray whales travel alone or in small, unstable groups and are known as bottom feeders that eat "benthic" amphipods.

Vocalization and Hearing

No data are available regarding Western North Pacific population of gray whale hearing and little regarding communication. The U.S. Navy has recorded short-duration (approximately one second) frequency sweeps at 55 Hz in the East China Sea, the likely source of which was determined to be Western North Pacific gray whales (Gagnon 2016). These sweeps are often emitted in pairs or triplets with an intersweep interval of approximately three or four seconds. These vocalizations contain multiple harmonics; the first harmonic is the weakest while the second and third harmonics are usually the strongest. Otherwise, we assume that Eastern North Pacific population of gray whale communication is representative of the Western North Pacific population of gray whale and present information stemming from this population. Individuals produce broadband sounds within the 100 Hz to 12 kHz range (Thompson et al. 1979; Dahlheim et al. 1984; Jones and Swartz 2002). The most common sounds encountered are on feeding and breeding grounds, where "knocks" of roughly 142 decibels re: 1 µPa at 1 m (source level) have been recorded (Cummings et al. 1968; Richardson et al. 1995b; Jones and Swartz 2002). However, other sounds have also been recorded in Russian foraging areas, including rattles, clicks, chirps, squeaks, snorts, thumps, knocks, bellows, and sharp blasts at frequencies of 400 Hz to 5 kHz (Petrochenko et al. 1991). Estimated source levels for these sounds ranged from 167 to 188 decibels re: 1 µPa at 1 m (Petrochenko et al. 1991). Low frequency (less than 1.5 kHz) "bangs" and "moans" are most often recorded during migration and during ice-entrapment (Carroll et al. 1989; Crane and Lashkari. 1996). Sounds vary by social context and may be associated with startle responses (Rohrkasse-Charles et al. 2011). Calves exhibit the greatest variation in frequency range used, while adults are narrowest; groups with calves were never silent while in calving grounds (Rohrkasse-Charles et al. 2011). Based upon a single captive calf, moans were more frequent when the calf was less than a year old, but after a year, croaks were the predominant call type (Wisdom et al. 1999).

Auditory structure suggests hearing is attuned to low frequencies (Ketten 1992b; Ketten 1992a). Responses of free-ranging and captive individuals to playbacks in the 160 Hertz to 2 kHz range demonstrate the ability of individuals to hear within this range (Cummings and Thompson 1971a; Dahlheim and Ljungblad 1990; Buck and Tyack 2000; Wisdom et al. 2001; Moore and Clark 2002). Responses to low-frequency sounds stemming from oil and gas activities also support low-frequency hearing (Malme et al. 1986; Moore and Clark 2002).

Population Dynamics

The current best estimate of the Western North Pacific population of gray whales is 140 (N_{min}=135) individuals (Carretta et al. 2018). The current best growth rate estimate for the Western North Pacific population of gray whale stock is 3.3% annually.

Contemporary gray whale genomes, both eastern and western, contain less nucleotide diversity than most other marine mammals and evidence of inbreeding is greater in the Western Pacific than in the Eastern Pacific populations (Brüniche-Olsen et al. 2018). The reduced diversity and increased inbreeding identified among the western gray whales are likely related to the small size of the western gray whale population, raising concerns about the loss of its adaptive potential (Brüniche-Olsen et al. 2018). Based on allele frequencies of 12 microsatellite loci, Lang et al. (2022) estimated the contemporary effective size of the western gray whale population to be 80 individuals.

Western gray whales migrate over 10,000 km (6,214 mi) to feed within a spatially small foraging ground off Sakhalin Island. Gailey et al. (2020) found that poor body condition and prolonged inter-birth intervals of Western gray whales in some years coincided with restricted ability to access the Sakhalin foraging habitat due to sea ice concentrations in the previous year. Gray whale density patterns off Sakhalin in 2015 were associated with the energy content of amphipods, cumaceans, and isopods, suggesting that gray whales primarily forage on crustacean species, and did not appear to target patches with high total benthic energy content or energy of other prey species groups such as polychaetes, bivalves, and small fish (Gailey et al. 2022). Amphipod biomass has decreased in nearshore areas on the Sakhalin feeding ground (Blanchard et al. 2019; WGWAP 2019).

Gray whales in the Western North Pacific population are thought to feed in the summer and fall in the Okhotsk Sea, primarily off Sakhalin Island, Russia and the Kamchatka peninsula in the Bering Sea, and winter in the South China Sea. However, tagging, photo-identification, and genetic studies have shown that some whales identified as members of the Western North Pacific stock have been observed in the Eastern North Pacific Ocean, which may indicate that not all gray whales share the same migratory patterns.

NMFS (2023j) determined the probability of encountering gray whales from the Eastern North Pacific population and the Western North Pacific DPS in various areas of the North Pacific Ocean migratory corridor using the abundance estimate of 14,526 individuals (Eguchi et al. 2023) and the 60 individuals (48 individuals in the lagoons in Mexico, 12 individuals in the migratory corridor off the West Coast of North America, and 4 individuals in both areas) photographed and identified that have migrated from the western North Pacific Ocean to Mexico. Based on this approach, it is expected that between 0.4-1.6% (with 1.2% considered the best estimated probability) of the gray whales found in the eastern North Pacific Ocean wintering areas (Mexico) and migratory corridor (United States West Coast [Alaska, Washington, Oregon, and California], Canada, and Mexico), from March through June and November through January, are from the Western North Pacific DPS (NMFS 2023j). The Western North Pacific

gray whale population annual growth rate has been positive in recent years. However, estimates of reproductive females are low, between 51 and 72 (Cooke et al. 2019), and estimated calf survival rate (0.65 ±0.07) is also low (Cooke et al. 2019).

Status

The Western North Pacific population of gray whale is endangered as a result of past commercial whaling and may still be hunted under "aboriginal subsistence whaling" provisions of the International Whaling Commission. Current threats include vessel strikes, fisheries interactions (including entanglement), habitat degradation, oil spills, harassment from whale watching, illegal whaling or resumed legal whaling, and noise. Due to elevated levels of strandings, in December of 2018 NMFS declared an Unusual Mortality Event (UME) along the West Coast of North America for the eastern North Pacific gray whale that lasted until November of 2023. However, it is unknown if oceanographic conditions related to this UME affected Western and Eastern gray whale populations similarly (Carretta 2021b). While the western gray whale population has slowly increased over the past several decades, recovering from a point of near extinction, ongoing threats from human activities such as entanglement in fishing gear, eco-tourism activities, and vessel strikes continue to impact this population's recovery (NMFS 2023i). This coupled with the population's small size and limited reproductive output makes it highly vulnerable to extinction (NMFS 2023i).

Critical Habitat

No critical habitat has been designated for the Western North Pacific population of gray whale. NMFS cannot designate critical habitat in foreign waters.

Recovery Planning

NMFS has not prepared a recovery plan for the Western North Pacific population of gray whale.

4.2.4 Humpback Whale – Central America and Mexico DPSs

The humpback whale is a widely distributed baleen whale found in all major oceans (Figure 26). Humpback whales are distinguishable from other whales by long pectoral fins and are typically dark grey with some areas of white. They humpback whale was originally listed as endangered on December 2, 1970 (35 Fed. Reg. 18319) under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained listed after the passage of the ESA in 1973. 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).

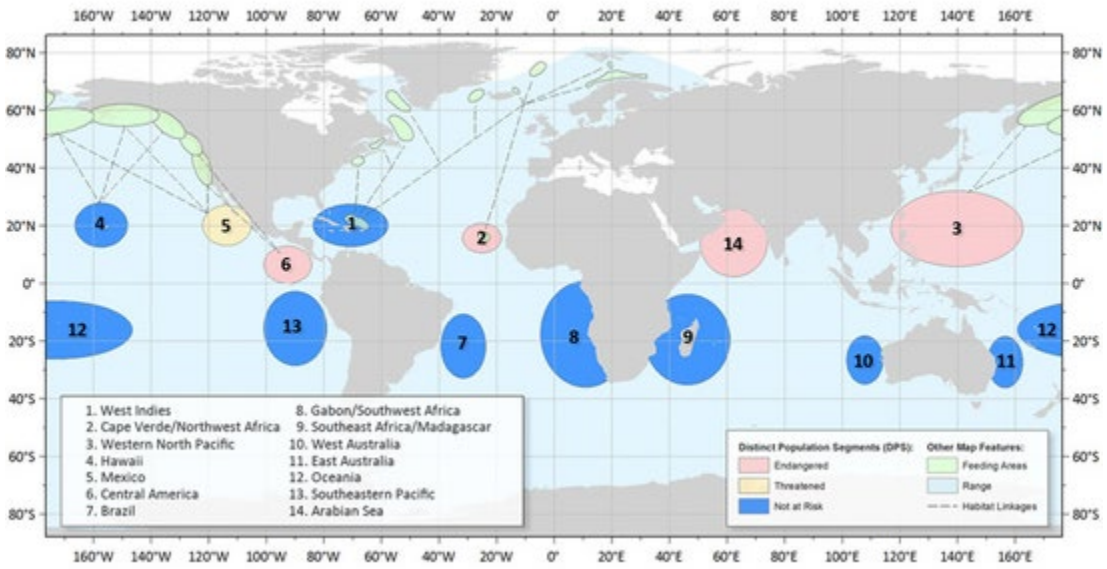


Figure 26. Map Identifying 14 Humpback Whale DPSs, Including One Threatened and Four Endangered, Based on Primarily Breeding Locations, Range, and Feeding Areas (Bettridge et al. 2015)

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 give 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. 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. 2015).

Maximum diving depths are approximately 170 m (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1 to 5.1 minutes in the north Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 minutes for feeding whales, 3.0 minutes for non-feeding whales, and 4.3 minutes for resting whales (Dolphin 1987). Because most humpback prey is likely found within 300 m 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 (Jurasz and Jurasz 1979; Hain et al. 1982; Weinrich et al. 1992; Hain et al. 1995). There is good evidence of some territoriality on feeding and calving areas (Tyack 1981; Clapham 1994; Clapham 1996).

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 m (Dolphin 1987).

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). 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 (Schevill et al. 1964; Helweg et al. 1992; Gabriele and Frankel. 2002; Clark and Clapham 2004; 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). Au et al. (2006) 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. Components of the song range from below 20 Hz up to 10 kHz, with source levels measured between 144 and 189 dB re: 1 µPa-m and high-frequency harmonics extending beyond 24 kHz (Winn et al. 1970; Richardson et al. 1995b; Frazer and Mercado Iii 2000; McCauley et al. 2000; Au et al. 2006). Perazio and Mercado III (2018) found that while frequencies from humpback whale songs in the Gulf of Tribugá in the Pacific ranged from 10 Hz to over 10 kHz, a much narrower frequency band of around 250 to 425 Hz was typical. Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144 to 174 dB (Winn et al. 1970; Richardson et al. 1995b; Frazer and Mercado Iii 2000; McCauley et al. 2000; Au et al. 2006). Singing is most common on breeding grounds during the winter and spring months, but is occasionally heard in other regions and seasons (Mcsweeney et al. 1989; Gabriele and Frankel. 2002; Clark and Clapham 2004).

Social calls range from 20 Hz to 10 kHz, with dominant frequencies below 3 kHz (D'Vincent et al. 1985; Silber 1986; Simao and Moreira 2005; Dunlop et al. 2008). Males also produce sounds associated with aggression, which are generally characterized by frequencies between 50 Hz to 10 kHz with most energy below 3 kHz (Tyack 1983; Silber 1986). Such sounds can be heard up to 9 km (4.9 nm) away (Tyack 1983). 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 Hz, and mean center frequencies of around 500 to 600 Hz (Zoidis et al. 2008). While the significance of these calls are unknown, they may serve as contact calls to the calf's mother (Zoidis et al. 2008; Indeck et al. 2020). 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).

While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25 to 89 Hz) and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz), which can be very loud (175 to 192 dB re: 1 µPa at 1 m) (Payne and Mcvay 1971; Thompson et al. 1992; Edds-Walton 1997; Oleson et al. 2007a). However, humpback whales tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995b). 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 kHz.

Recalde-Salas et al. (2020) recorded non-song vocalizations from humpback whales off Western Australia. The frequencies of these sounds ranged from 9 Hz to 6 kHz, the majority being under 200 Hz. 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 kHz (NMFS 2018a). Humpback whale audiograms using a mathematical model based on the internal structure of the ear estimate sensitivity is from 700 Hz to 10 kHz, with maximum relative sensitivity between 2 kHz and 6 kHz (Ketten and Mountain 2014). Research by Au et al. (2001) and Au et al. (2006) off Hawaii indicated the presence of high-frequency harmonics in vocalizations up to and beyond 24 kHz. 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 kHz 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 kHz at 219 dB re: 1 µPa-m or frequency sweep of 3.1 to 3.6 kHz. In addition, the system had some low frequency components (below 1 kHz) 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.

Population Dynamics

Based on habitat models and 1991-2018 line-transect data, Becker et al. (2020) estimated that humpback whale abundance in California, Oregon, and Washington waters is 4,784 whales (CV=0.31). However, this estimate reflects only humpback whale densities within the study area during summer and autumn when the line-transect surveys were conducted. The best estimate of current abundance of humpback whales along the U.S. West Coast, based on mark-recapture data from 2015-2018 and a Chao model that accounts for heterogeneity of capture probabilities, is 4,973 (CV=0.048) whales (Calambokidis and Barlow 2020; Carretta 2022). The growth rate of the California/Oregon/Washington stock is estimated to be around 8% since the late 1980s (Calambokidis and Barlow 2020). The California/Oregon/Washington stock showed a long-term increase in abundance from 1990 through approximately 2008, but more recent estimates through 2014 indicate a leveling-off of the population size (Calambokidis et al. 2017).

Curtis et al. (2022) estimated the population size of whales wintering in southern Mexico and Central America using spatial capture-recapture methods based on photographic data collected between 2019 and 2021. Their estimate of abundance is 1,496 (CV=0.171) whales, which represents the best estimate of abundance for the Central America / Southern Mexico - CA-OR-WA stock of humpback whales (Carretta et al. 2023b). The minimum population estimate for this stock is taken as the lower 20th percentile of the capture-recapture estimate from Curtis et al. (2022), or 1,284 whales (Carretta et al. 2023b). Curtis et al. (2022) derived a population growth rate for Central America / Southern Mexico whales based on differences between 2004-2006 estimates and the current estimate by excluding whales from southern Mexico, yielding an annual growth rate of 1.6% (SD = 2.0%) for this stock of humpback whales (Carretta et al. 2023b). There is currently no abundance estimate for the Mexico DPS, although an estimated 3,477 (CV=0.101) whales from the Mexico DPS feed off the U.S. West Coast (Calambokidis and Barlow 2020; Curtis et al. 2022).

Humpback whales 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.

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.

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, with only a few individuals identified at the northern Washington – southern British Columbia feeding grounds (81 Fed. Reg. 62259). 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 Fed. Reg. 62259). The proportion of whales heading to Central America or mainland Mexico generally corresponds to the latitude at which they feed; those feeding at lower latitudes will likely migrate further south to breed. Whales that feed off Southern California are more likely to go to Central America than those found further north along the coast (Calambokidis et al. 2017). In contrast, whales that feed off Central California to Washington migrate predominantly to northern Mexico (Calambokidis et al. 2017). Although there is not typically an exchange of individuals between groups of whales that share the same wintering and feeding area (migratory group), interbreeding and exchange of genetic material between groups likely occurs (Martien et al. 2021; Darling et al. 2022). Martínez-Loustalot et al. (2022) studied the relationship between humpback whales of southern Mexico and those of the surrounding population units. Their results showed higher movements among the southern sampling areas, suggesting that whales from southern Mexico belong to the Central American population unit. This population unit migrates north to the feeding areas of the U.S. West Coast using a migratory corridor along mainland Mexico to the mouth of the Gulf of California along the Baja California Peninsula (Martínez-Loustalot et al. 2022). The degree of interaction with humpback whales from other population units is unknown. De Weerdt Joelle et al. (2023) found evidence of fine-scale site affinity of individual humpback whales within Nicaraguan coastal waters and with other breeding and feeding areas, suggesting potential spatial structuring within the Central American DPS.

Critical Habitat

Designated critical habitat was found to not likely be adversely affected by the action, as described in Section 4.1.3.

Recovery Planning

In response to the threats facing the species, NMFS originally developed a recovery plan for humpback whales in 1991 (NMFS 1991). The major actions recommended in the existing species-wide recovery plan were:

- Reduce or eliminate injury and mortality caused by fisheries, fishing gear, and vessel collisions
- Minimize effects of vessel disturbance
- Continue the international moratorium on commercial whaling
- Collect as much data as possible from dead whales through our Marine Mammal Health and Stranding Program

Following the 2016 change in the listing status of humpback whales, NMFS is currently developing an updated, DPS-specific recovery plan for the ESA-listed Central America, Mexico, and Western North Pacific DPSs. In the interim, the recovery strategy for these DPSs will be guided by the existing species-wide plan and a new DPS-specific recovery outline for the Western North Pacific, Central America, and Mexico DPSs of humpback whale (NMFS 2022d). The recovery outline serves as an interim guidance document to direct recovery efforts until a revised full recovery plan is developed and approved, thus replacing the 1991 recovery plan. The interim recovery program will focus on the following goals:

- Management activities that continue to protect humpback whales and their critical habitat
- Management activities that reduce medium and high risk threats to humpback whales, including vessel strike and entanglement in fishing gear
- Research activities to fill critical information gaps necessary to inform management actions
- Education and outreach activities to engage ocean users and to promote public involvement in humpback whale research and recovery

4.2.5 Sei Whale

The sei whale was originally listed as endangered on December 2, 1970 under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained listed after the passage of the ESA in 1973. The sei whale is a widely distributed baleen whale found in all major oceans (Figure 27).

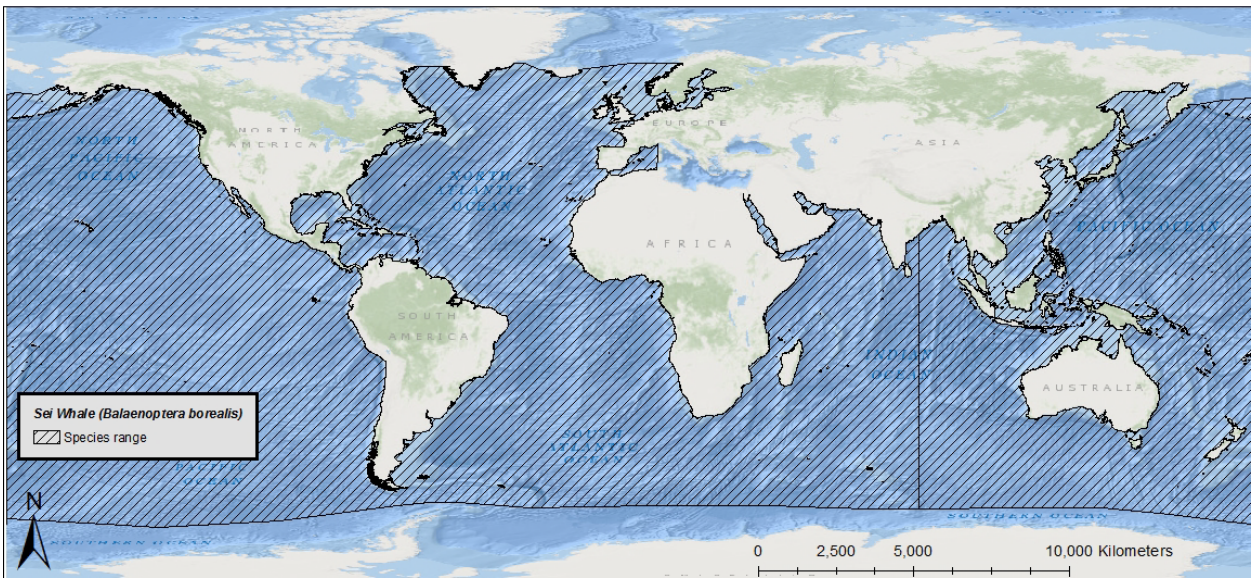


Figure 27. Map Identifying the Range of the Endangered Sei Whale

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 (Wiles 2017). Sei whales filter prey through their baleen, and have the unique ability to capture prey via engulfment (like other rorquals), or via skimming on lower prey concentrations (Prieto et al. 2012).

Vocalizations 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 kHz (Richardson et al. 1995b). Tremblay et al. (2019) recorded 50 to 30-Hz triplet and singlet downsweeps and 82 to 34-Hz downsweeps from sei whales in the western North Atlantic, suggesting that sei whales may produce songs. Source levels of 189 ±5.8 decibels re: 1 µPa at 1 meter have been established for sei whales in the northeastern Pacific Ocean (Weirathmueller et al. 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 (Richardson et al. 1995b; 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 36 kHz (NMFS 2024g).

Population Dynamics

The central and eastern North Pacific Ocean sei whale population was estimated to be 29,632 (95% confidence interval from 18,576 to 47,267) between 2010 and 2012 (IWC 2016; Thomas et al. 2016; Hakamada et al. 2017). To this can be added an estimate of 5,086 whales from surveys in the western North Pacific (Hakamada and Matsuoka 2016), giving a total estimate of about 35,000 sei whales in the North Pacific Ocean (NMFS 2021g).

The most recent estimate of sei whale abundance in the California Current is 864 (CV = 0.40) based on the 2014 survey data (Carretta et al. 2025). The current minimum population estimate is 625 whales (Carretta 2022). Although there was no formal assessment of an abundance trend for sei whales, estimates reported in Barlow (2016) showed an increasing trend from 1991-2014, with the 2014 estimate being the highest estimated (Carretta et al. 2025).

Encounter data from a 2010 shipboard line-transect survey of the entire Hawaiian Islands EEZ was evaluated using Beaufort sea-state-specific trackline detection probabilities for sei whales, resulting in an abundance estimate of 391 (CV = 0.9) sei whales (Bradford et al. 2017) in the Hawaii stock; the current minimum population estimate is 204 (Carretta 2022). No data are available on the current population trend for the Hawaii stock of sei whales. Scientifically reliable abundance estimates are not available for the North Atlantic and Southern Hemisphere, due to inherent uncertainties in sampling design, data collection methodologies, and use of outdated abundance estimates (NMFS 2021g).

Two sub-species 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). Taguchi et al. (2021) conducted the first population genetic study of sei whales worldwide using microsatellite DNA (msDNA). He found that sei whales appear to be significantly differentiated among oceanic regions hierarchically, and sei whales in the Southern Hemisphere are more closely related to sei whales in the North Pacific than sei whales in the North Atlantic (Taguchi et al. 2021). Other genetic studies have found that there is high haplotype diversity in the North Pacific sei whale population and moderate haplotype diversity in the North Atlantic population, indicating greater diversity in the Pacific Ocean (Pastene et al. 2016; Huijser et al. 2018). A sei whale genetics

study by Tamura et al. (2019) showed no evidence of genetic heterogeneity and a lack of genetic structuring, further supporting the single stock hypothesis for North Pacific sei whales(Taguchi et al. 2021). Overall, these studies indicate that the sei whale has moderate to high genetic diversity, providing the raw genetic material required to adapt to changes in its environment, and thus some resilience to such perturbations (NMFS 2021g).

Status

The sei whale is endangered as a result of past commercial whaling. Due to a lack of comprehensive abundance and distribution data for all three ocean basins, and absence of dedicated systematic surveys, there is no scientifically rigorous estimate of global abundance. As such, there are insufficient data to determine global population trends for sei whales. A crude estimate of global decline from approximately 250,000 whales before whaling to perhaps 32,000 whales by the 1970s to 1980s is reported (Wiles 2017). Sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). Currently, only a few individuals are taken each year by Japan; however, Iceland has expressed an interest in targeting sei whales.

Although the population in the North Pacific is expected to have grown since being given protected status in 1976, the possible effects of continued unauthorized takes and ongoing threats make this uncertain (Carretta 2021b). Ongoing threats include vessel strikes, fisheries interactions (including entanglement), changing environmental conditions (habitat loss and reduced prey availability), and anthropogenic sound. Given the species' overall abundance, they may be somewhat resilient to current threats. However, trends are largely unknown, especially for individual stocks, many of which have relatively low abundance estimates

Critical Habitat

No critical habitat has been designated for the sei whale.

Recovery Planning

In response to the current threats facing the species, NMFS developed goals to recover sei whale populations. These threats will be discussed in further detail in the Environmental Baseline section of this consultation. See the 2011 Final Recovery Plan (National Marine Fisheries Service 2011) for the sei whale for complete downlisting/delisting criteria for both of the following recovery goals:

- Achieve sufficient and viable populations in all ocean basins.
- Ensure significant threats are addressed.

Threats to sei whales are discussed in further detail in the environmental baseline section of this opinion (Section 5). Downlisting criteria for sei whales include:

- Factors that may limit population growth (i.e., those that are identified as either high, medium, or unknown impact to recovery) have been identified and are being or have been addressed to the extent that they allow for continued growth of populations in each ocean basin in which it occurs (North Atlantic, North Pacific and Southern Hemisphere).

- The population in each ocean basin satisfies the risk analysis standard for threatened status (has no more than a 1% chance of extinction in 100 years) and has at least 1,500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males) (National Marine Fisheries Service 2011).

4.2.6 Sperm Whale

The sperm whale was originally listed as endangered on December 2, 1970 under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained listed after the passage of the ESA in 1973. The sperm whale is widely distributed and found in all major oceans (Figure 28).

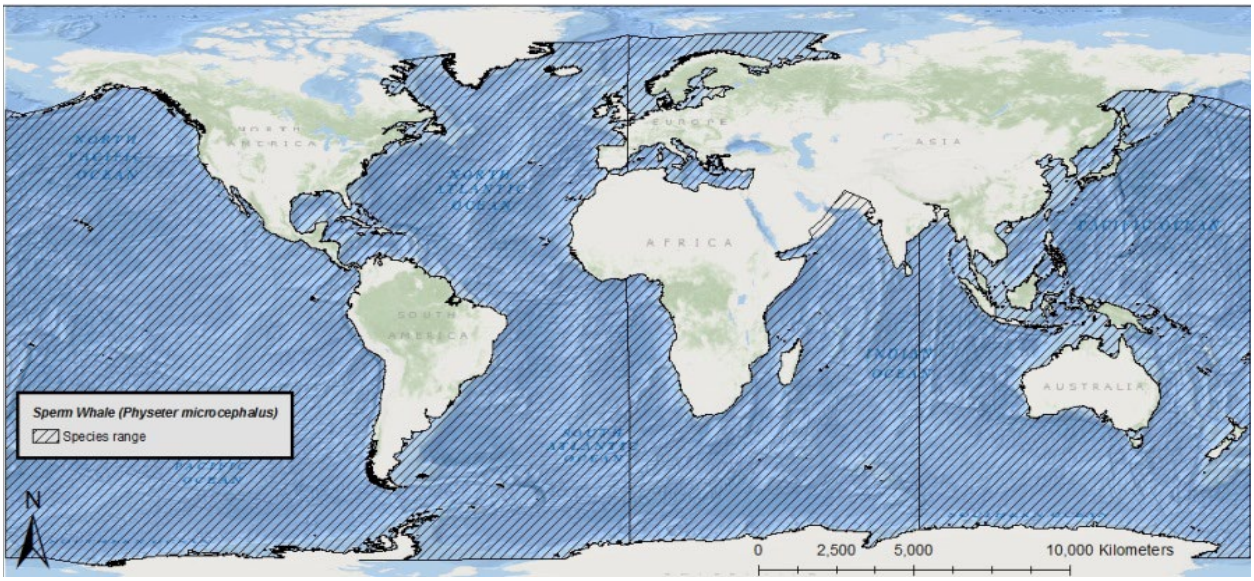


Figure 28. Map Identifying the Range of the Endangered Sperm Whale

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 seven 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 m (1,968 ft) or more, and are uncommon in waters less than 300 m (984 ft) 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 giant squid. Other prey includes octopus and demersal fish (including teleosts and elasmobranchs).

Sperm whales are one of the deepest and longest diving marine mammals, with recorded dives up to 3 km down and durations in excess of two hours (Clarke 1976; Watkins 1985; Watkins et al. 1993). However, typical dives are generally shorter (25 to 45 min) and shallower (400 to 1,000 meters). Dives are separated by 8 to 11 minutes rests at the surface (Gordon 1987; Papastavrou et al. 1989; Jochens et al. 2006; Watwood et al. 2006). Sperm whales typically travel about 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.

Sperm whales have a strong preference for waters deeper than 1,000 meters (Watkins and Schevill 1977; Reeves and Whitehead 1997), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. Sperm whales are rarely found in waters less than 300 m in depth, and these shallower areas are often associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956; Rice 1989). 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).

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 six 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).

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 kHz (Watkins 1977) and dominant frequencies between 1 to 6 kHz and 10 to 16 kHz. Another class of sound, "squeals," are produced with frequencies of 100 Hz to 20 kHz (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 (Weilgart and Whitehead 1993; Goold and Jones 1995; Weilgart and Whitehead 1997; Mohl et al. 2003). Most of the

energy in sperm whale clicks is concentrated at around 2 to 4 kHz and 10 to 16 kHz (Weilgart and Whitehead 1993; Goold and Jones 1995). 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 kHz) 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 (Whitehead and Weilgart 1991; Weilgart and Whitehead 1993; Goold and Jones 1995; Weilgart and Whitehead 1997; Miller et al. 2004). While foraging, sperm whales typically produce clicks in a 10 Hz–30 kHz frequency band, at a rate defining an inter-click interval that varies from 0.5 to 2.0 seconds during descent from the surface until the whale begins its ascent to the surface (André et al. 2017). 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 (Miller et al. 2004; Laplanche et al. 2005). 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 (Weilgart and Whitehead 1997; Rendell and Whitehead 2004).

Sperm whales are considered to be part of the high-frequency marine mammal hearing group, with a hearing range between 150 Hz and 160 kHz (NOAA 2016). 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 kHz and highest sensitivity to frequencies between 5 to 20 kHz. 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 1992b). The sperm whale may also possess better low-frequency hearing than other odontocetes, although not as low as many baleen whales (Ketten 1992b). 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 and Schevill 1975; Watkins et al. 1985).

Population Dynamics

The sperm whale is the most abundant of the large whale species. The current global sperm whale estimate is about 850,000 whales (Whitehead and Shin 2022). In the northeast Pacific Ocean, the abundance of sperm whales was estimated to be between 26,300 and 32,100 in 1997. In the eastern tropical Pacific Ocean, the abundance of sperm whales was estimated to be 22,700 (95% confidence intervals 14,800 to 34,600) in 1993 (Carretta 2022). Population estimates are

also available for two of three U.S. stocks that occur in the Pacific. The most-recent estimate of sperm whale abundance for the California/Oregon/Washington stock, based on a 2018 survey and a habitat density model that is informed by 1991-2018 data (Becker et al. 2020), is 2,606 individuals (N_{min}=2,011)(Carretta et al. 2025). The Hawaii stock is estimated to consist of 5,387 individuals (N_{min}=4,486) (Becker et al. 2021).

Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70 degrees North in the North Atlantic and 70 degrees South in the Southern Ocean (Reeves and Whitehead 1997; Perry et al. 1999), whereas mature females and immature individuals of both sexes are seldom found higher than 50 degrees North or South (Reeves and Whitehead 1997). Sperm whales distribute widely throughout the North Pacific Ocean, with movements over 5,000 km (3,107 mi), likely driven by changes in prey abundance. Males appear to range more broadly than females (Mizroch and Rice 2013). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring 1993) where adult males join them to breed.

In the Pacific, sperm whales are primarily found in temperate and tropical waters (Rice 1989; Merkens et al. 2019). This species appears to have a preference for deep waters and the continental shelf break and slope (Rice 1989; Whitehead 2003b; Whitehead et al. 2008; Baird 2013; Jefferson et al. 2015). Typically, sperm whale concentrations also correlate with areas of high productivity generally near drop offs and areas with strong currents and steep topography (Gannier and Praca 2007; Jefferson et al. 2015). Using survey data from 1991 to 2018, high densities of sperm whales have been predicted to occur along the 2,000 m isobath in the California Current Ecosystem in summer (Becker et al. 2020).

Sperm whales are found year-round in California waters, but they reach peak abundance from April through mid-June and from the end of August through mid-November (Barlow 1995; Forney et al. 1995; Carretta 2021b). Sperm whales have been sighted throughout the Hawaiian EEZ, including nearshore waters of the main and Northwestern Hawaiian Islands (Mobley Jr et al. 2000; Baird 2016). In addition, the sounds of sperm whales have been recorded throughout the year within the main and Northwest Hawaiian Islands (Merkens et al. 2019).

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.

Status

The sperm whale is endangered as a result of past commercial whaling. 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. Current potential threats likely affecting sperm whale recovery include vessel strikes, entanglement in fishing gear, oil spills anthropogenic noise, exposure to contaminants, changing environmental conditions, and marine debris (NMFS 2015c). The species' large population size shows that it may be somewhat resilient to current threats. However, the magnitude of many of these threats is highly uncertain, and the impact of some threats (e.g., contaminant levels and changing environmental conditions) may be intensifying (NMFS 2015c).

Critical Habitat

No critical habitat has been designated for the sperm whale.

Recovery Planning

In response to the current threats facing the species, NMFS developed goals to recover sperm whale populations. These threats are discussed in further detail in the Environmental Baseline (Section 5) of this consultation. See the 2010 Final Recovery Plan (NMFS 2010a) for the sperm whale for complete downlisting/delisting criteria for both of the following recovery goals:

- Achieve sufficient and viable populations in all ocean basins; and
- Ensure significant threats are addressed.

The 2010 sperm whale recovery plan (NMFS 2010a) 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, changing environmental conditions, direct harvest, oil spills, anthropogenic noise, and ship collision. 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 1% chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10% chance of becoming endangered in 20 years.

4.2.7 False Killer Whale – Main Hawaiian Islands Insular DPS

NMFS recognizes three stocks of false killer whales in Hawaiian waters: the Main Hawaiian Islands insular, Hawaii pelagic, and the Northwestern Hawaiian Islands. All false killer whales found within 40 km (21.6 nm) of the Main Hawaiian Islands belong to the insular stock and all false killer whales beyond 140 km (75.6 nm) belong to the pelagic stock. Animals belonging to the Northwest Hawaiian Islands stock are insular to the Northwest Hawaiian Islands (Bradford et al. 2012), however, this stock was identified by animals encountered off Kauai.

The Main Hawaiian Islands insular DPS of false killer whale (MHI IFKW) was originally listed as endangered on November 28, 2012. As shown in Figure 29, this DPS of false killer whales has a limited range around the Main Hawaiian Islands.

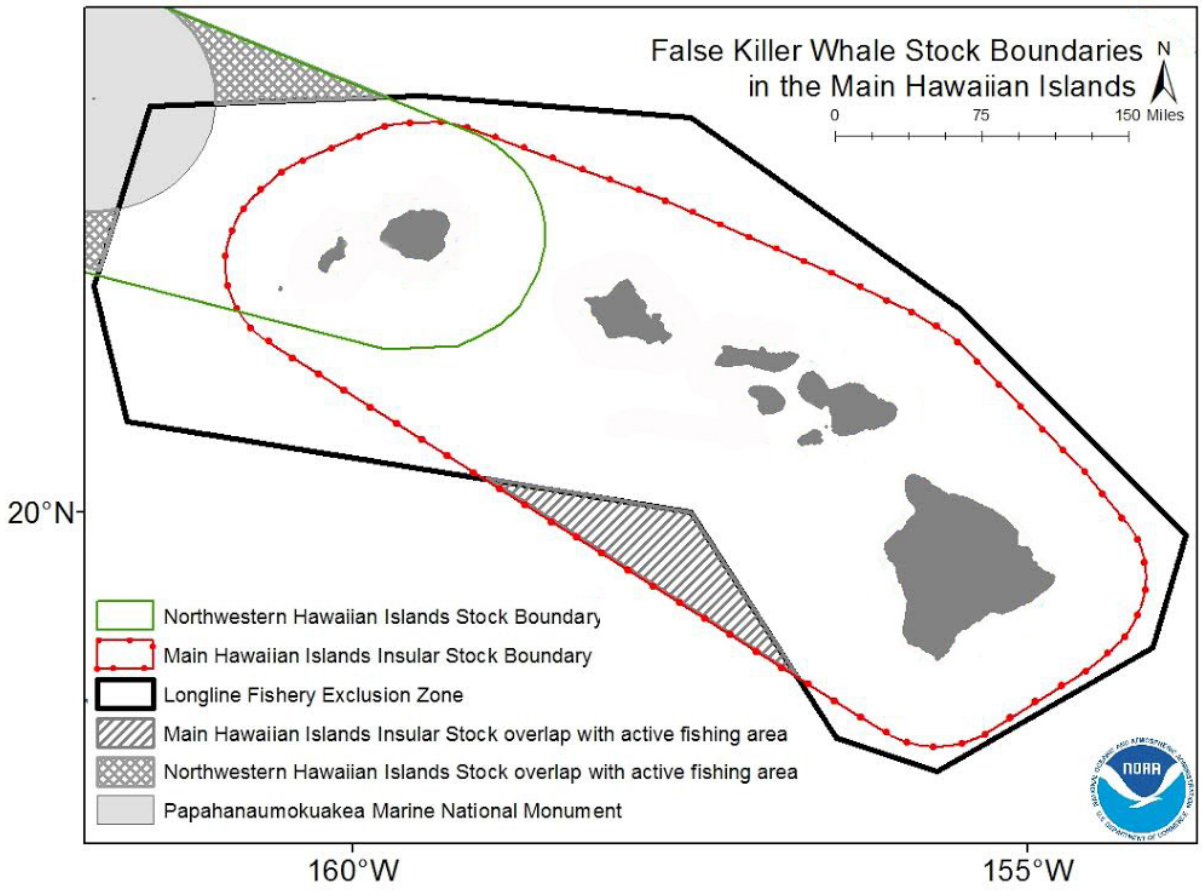


Figure 29. Map of the Main Hawaiian Islands Insular DPS of False Killer Whale Boundary (NMFS 2021d)

Life History

The false killer whale is a toothed whale and large member of the dolphin family. False killer whales can live, on average, for 60 years. They have a gestation period of 11 to 16 months, and calves nurse for 1.5 to two years. Females reach sexual maturity at around 8 to 11 years of age with a very low reproduction rate and calving interval of approximately seven years. Calving occurs within tropical waters year-round. False killer whales prefer tropical to temperate waters that are deeper than 1,000 meters (3,281 feet). They feed during the day and at night on fishes and cephalopods, and are known to attack other marine mammals, indicating they may occasionally feed on them.

Photo-identification data from 2020 and 2021 were combined with identifications available from 1999 through 2019 in an analysis to identify the number and membership of MHI IFKW social clusters (Baird et al. 2023). Results showed cluster specific spatial use indicating there are four discrete social clusters within the population. Clusters varied in their spatial use by distance from shore, in relation to seafloor depth, and in terms of the proportion of time they spent in windward or leeward waters around the islands (Baird et al. 2023).

Vocalization and Hearing

In general, odontocetes produce sounds across the widest band of frequencies. There are three categories of sounds that odontocetes make. The first includes echolocation sounds of high intensity, high-frequency, high repetition rate, and very short duration that are used to detect, localize and characterize underwater objects such as prey (Au et al. 1993; Au et al. 2000). The second category of odontocete sounds is comprised of pulsed sounds. Echolocation clicks have source levels that can be as high as 229 decibels re: 1 µPa peak-to-peak (Au et al. 1974). Burst pulses are generally very complex and fast, with frequency components sometimes above 100 kHz and average repetition rates of 300 per second (Yuen et al. 2007). The final category of odontocete sounds is the narrowband, lower frequency, tonal whistles (Caldwell et al. 1990; Au et al. 2000). With most of their energy below 20 kHz, whistles have been observed with an extensive variety of frequency patterns, durations, and source levels, each of which can be repeated or combined into more complex phrases (Tyack and Clark 2000; Yuen et al. 2007). Odontocete social vocalizations range from a few hundreds of Hertz to tens of kHz (Southall et al. 2007) with source levels in the range of 100 to 170 decibels re: 1 µPa (see (Richardson et al. 1995b).

False killer whales produce echolocation clicks, whistles, and burst pulses. Echolocation clicks have been recorded over a wide range of frequencies from 17-104 kHz (Au et al. 1995; Madsen et al. 2004; Thode et al. 2016). Studies suggest that the peak frequency of false killer whale echolocation signals may be dependent on the source level, being low at low source levels and high at high source levels (Thomas et al. 1988; Thomas and Turl 1990; Au et al. 1995). The production and perception of large bandwidth signals utilized by delphinid odontocetes, including false killer whales, may provide the opportunity for greater target information, including more echo highlights of prey (i.e., finer detail) and greater range resolution (Kloepper et al. 2010). Lower frequency whistles, used by false killer whales for communication, are typically between 4-10 kHz (Murray et al. 1998; Oswald et al. 2003; Thode et al. 2016).

Nachtigall and Supin (2008) investigated the signals from an echolocating false killer whale and found that the majority of clicks had a single-lobed structure with peak energy between 20 and 80 kHz false rather than dual-lobed clicks, as has been demonstrated in the bottlenose dolphin. U.S. Navy researchers measured the hearing of a false killer whale and demonstrated the ability of this species to change its hearing during echolocation (Nachtigall and Supin. 2008). They found that there are at least three mechanisms of automatic gain control in odontocete echolocation, suggesting that echolocation and hearing are a very dynamic process (Nachtigall and Supin. 2008). For instance, false killer whales change the focus of the echolocation beam based on the difficulty of the task and the distance to the target. The echo from an outgoing signal can change by as much as 40 decibels, but the departing and returning signal are the same strength entering the brain (Nachtigall and Supin. 2008). The U.S. Navy demonstrated that with a warning signal, the false killer whale can adjust hearing by 15 decibels prior to sound exposure (Nachtigall and Supin. 2008).

Functional hearing in high-frequency cetaceans, including MHI IFKWs, is conservatively estimated to be between approximately 150 Hertz and 160 kHz (Southall et al. 2007). Hearing measurements of a false killer whale show a hearing range between 4-50 kHz with best sensitivity between 16-24 kHz (Yuen et al. 2007).

Population Dynamics

Using methods from Oleson et al. (2010), NMFS (2021e) came up with a revised plausible historical abundance estimate of 655 animals. Aerial survey sightings from 1989 to 2003 suggest that the population declined until the early to mid-2000s (NMFS 2022c; Carretta et al. 2023a). In 2018, published estimates of abundance ranged between 144 and 187. However, Badger et al. (2025) estimate 139 individuals (95% CRI: [120, 160]) in the population in 2021. The population appears to be in decline, with an estimated trend of -3.5% over the last 10 years (Badger et al. 2025).

The Main Hawaiian Islands insular DPS of false killer whale is considered resident to the Main Hawaiian Islands and is genetically and behaviorally distinct compared to other stocks. Genetic data suggest little immigration into the Main Hawaiian Islands insular DPS of false killer whale. Genetic analyses indicated restricted gene flow between false killer whales sampled near the Main Hawaiian Islands, the Northwestern Hawaiian Islands, and pelagic waters of the Eastern and Central North Pacific Ocean. Density based on information from 38 satellite-tagged individuals is shown in Figure 30.

Martien et al. (2019) investigated mating patterns of MHI IFKWs, using both genetic and photo-ID data. Their research indicates that between 36% and 64% of matings involved individuals from the same social group or cluster, which is a sign that inbreeding is occurring within this small resident population. Results suggest that because the population declined between the 1980s and early 2000s, the intra-group matings may be the result of reduced opportunities for inter-group matings since the decline (Martien et al. 2019). When inbreeding persists over multiple generations, it can lead to a decline in genetic diversity and increase the chance that individuals inherit harmful recessive alleles from both parents (Martien 2025). Prior to the population decline, social groups may have been sufficiently large that selective pressure to develop inbreeding avoidance mechanisms was low, or MHI IFKWs may have evolved alternate inbreeding avoidance mechanisms such as kin recognition (Martien et al. 2019).

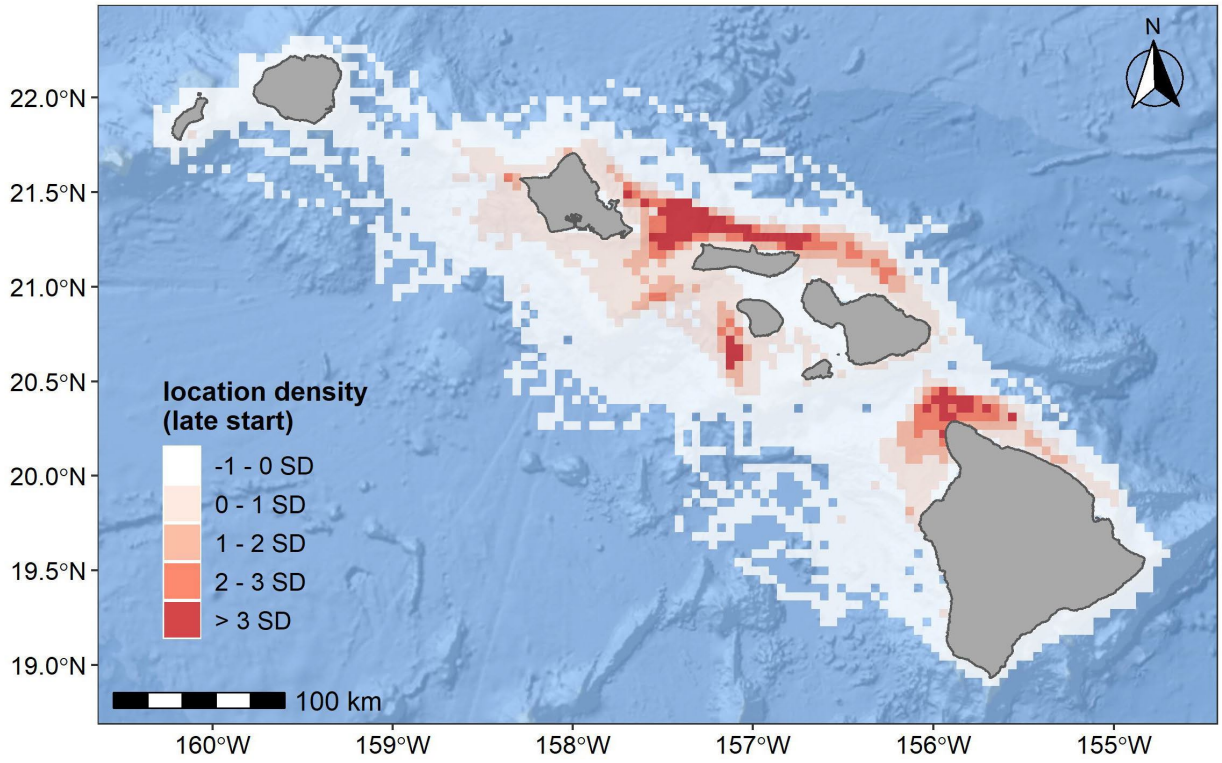


Figure 30. MHI IFKW Density by All Five Clusters Representing 38 Satellite-tagged Individuals Deployed through August 2021 (updated by Dr. Robin Baird, Cascadia Research Collective, pers. comm. August 2021 with the original source as Baird et al. (2012) (NMFS 2021e).

Mahaffy et al. (2023) used photo-identification data over a 23-year period to reassess the number and membership of social clusters in the MHI IFKW population. Their results indicate that the four social clusters identified have significant differences in association patterns, spatial use, and genetics, including significant differentiation in nuclear DNA. All individuals in two of the clusters possess the same mitochondrial DNA haplotype, while in the other two clusters, approximately 40% of animals possess a second haplotype (Mahaffy et al. 2023). Mahaffy et al. (2023) found the following spatial use patterns by cluster (Figure 31): Cluster 1 (n=62 individuals) was seen primarily off the northwest side of Hawaii, including the Alenuihaha Channel, but also in the 4-islands region; Cluster 2 (n=15 individuals) was primarily seen between Hawaii and Maui, including the Alenuihaha Channel; Cluster 3 (n=60 individuals) was seen across all Main Hawaiian Islands but primarily in the area between Oahu and Molokai, including the Kaiwi Channel; and Cluster 4 (n=37 individuals) was seen primarily off the western sides of Molokai and Lanai, including the Kaiwi Channel.

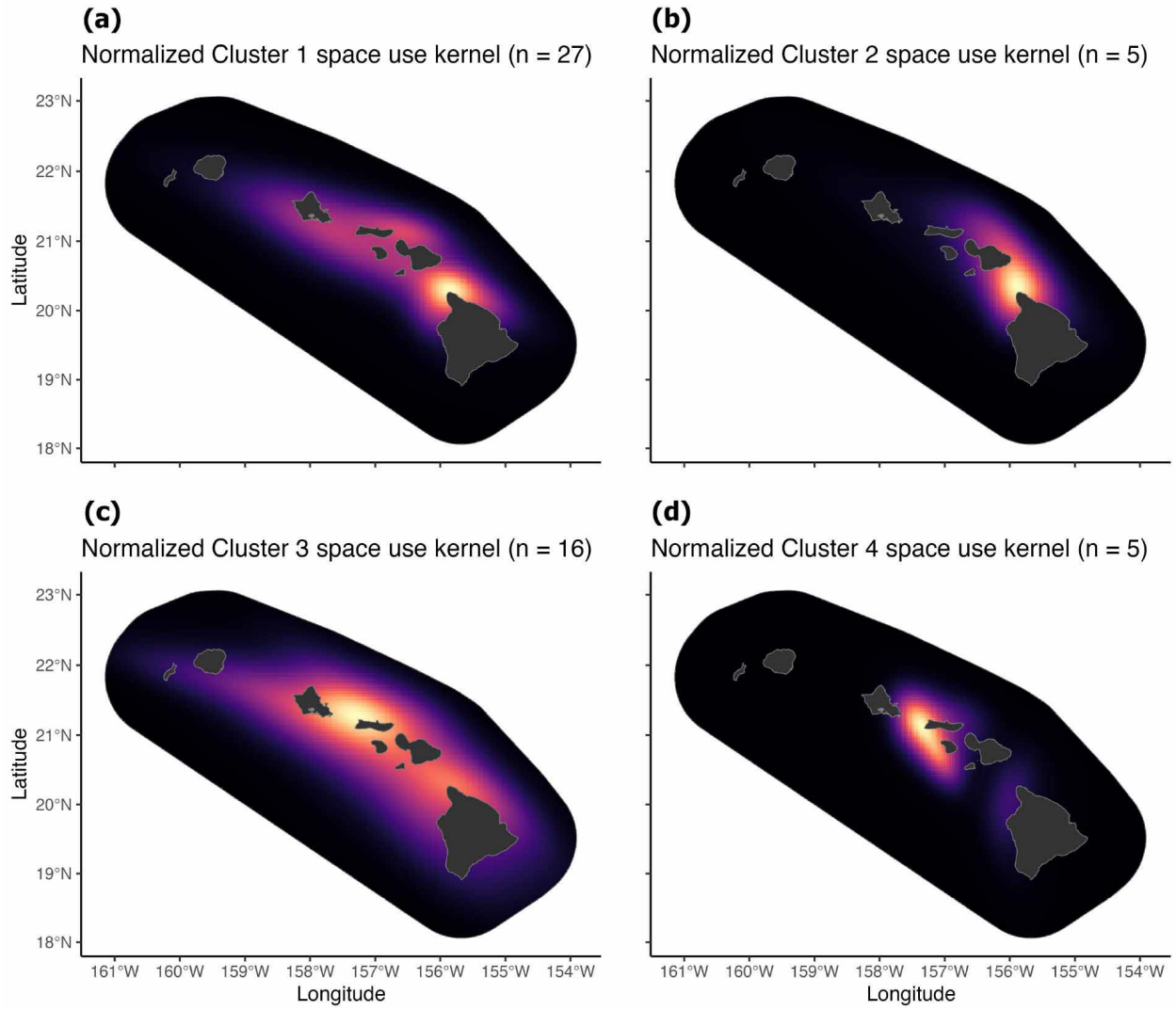


Figure 31. Cluster-level Space Use of Main Hawaiian Islands Insular False Killer Whales by Social Cluster Based on Satellite-tag Data from 2007 to 2022 (Mahaffy et al. 2023).

Status

The most recent MHI IFKW status review (NMFS 2022c) identified the following threats of high concern or moderate-to-high concern: 1) incidental take in non-longline commercial and recreational fisheries, 2) inadequate regulatory mechanisms of non-longline commercial and recreational fisheries, 3) small population size, 4) competition with commercial non-longline fisheries, 5) competition with recreational fisheries, and 6) environmental contaminants (e.g., polychlorinated biphenyls [PCBs], heavy metals) and naturally occurring biotoxins. The population has likely declined until at least the early 2000s, although trends over the last few decades are less clear (NMFS 2022c).

Critical Habitat

Critical habitat for the MHI IFKW was designated on July 24, 2018, with an effective date of August 23, 2018 (83 Fed. Reg. 35062). The designation includes waters from the 45 m (148 ft) depth contour to the 3,200 m (10,499 ft) depth contour around the Main Hawaiian Islands (Figure 32). Parts of the designation were excluded for national security or economic reasons because the benefits of excluding these areas outweigh the benefits of designation. Of the approximately 56,821 km^2 (21,933 mi^2) of total critical habitat area considered, about 20% (11,317 km^2) were excluded for national security reasons.

The designated critical habitat includes one PBF essential for conservation of the species, with the following four characteristics:

1. Adequate space for movement and use within shelf and slope habitat;
2. Prey species of sufficient quantity, quality, and availability to support individual growth; reproduction, and development, as well as overall population growth;
3. Waters free of pollutants of a type and amount harmful to MHI IFKWs;
4. Sound levels that would not significantly impair false killer whales' use or occupancy.

MHI IFKW diet consists primarily of pelagic fish and squid (NMFS 2018c), but without further information on prey preferences, NMFS was unable to determine where prey resources of higher value exist for MHI IFKW within or outside designated critical habitat (NMFS 2018c). MHI IFKW prey may include various tuna species, marlin species, jack species, mahi mahi, wahoo, moonfish, and squid (NMFS 2018c). Recent stomach content analysis from MHI IFKWs that stranded from 2010-2016 has detected seven Genus of fish and four species of cephalopods. Of those, diamondback squid (*Thysanoteuthis rhombus*) were the most common prey item.

Regarding the characteristic specific to sound levels, the final rule to designate critical habitat defined these sound levels as those that inhibit MHI IFKW's "…*ability to receive and interpret sound for the purposes of navigation, communication, and detection of predators and prey. Such noises are likely to be long-lasting, continuous, and/or persistent in the marine environment and, either alone or added to other ambient noises, significantly raise local sound levels over a significant portion of an area*" (83 Fed. Reg. 35062). The final biological report developed in support of the final rule discussed the complexity of analyzing how human activities may change an animal's use of an area (NMFS 2018c). The biological report emphasized that "…*the duration of the offending or masking noise will determine whether the effects or degradation to the habitat may be temporary or chronic, and whether such alterations to the soundscape may alter the conservation value of that habitat*" (NMFS 2018c).

The final rule to designate critical habitat identified several activities that may threaten the PBF essential to conservation such that species management considerations or protections may be required. Major categories of activities included in the final rule were (1) in-water construction (including dredging); (2) energy development (including renewable energy projects); (3) activities that affect water quality; (4) aquaculture/mariculture; (5) fisheries; (6) environmental

restoration and response activities (including responses to oil spills and vessel groundings, and marine debris clean-up activities); and (7) some military readiness activities.

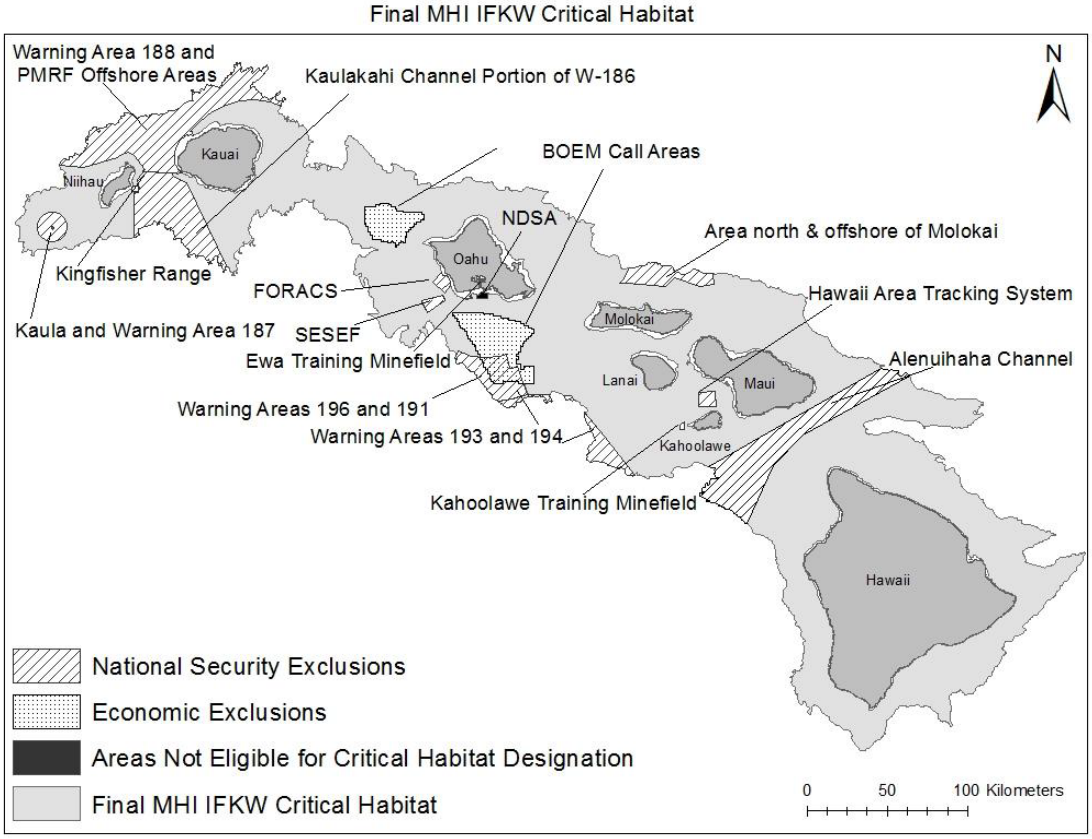


Figure 32. Designated Critical Habitat for Main Hawaiian Islands Insular DPS False Killer Whale

Recovery Planning

See the 2021 Recovery Plan for the MHI IFKW DPS for complete down listing/delisting criteria for each of their respective recovery goals (NMFS 2021c). Considering the small population size, life history characteristics, and the threats affecting this DPS, the Recovery Plan lists the following strategy:

1. Increase the population size through management actions that increase survival and decrease mortality due to known threats
2. Address threats from fisheries, including incidental take and competition with fisheries for prey
3. Protect, maintain, and enhance habitat by identifying and minimizing environmental contaminants, biotoxins, anthropogenic noise, and the effects of changing environmental conditions, and planning for other rare threats such as oil spills
4. Ensure that other regulatory mechanisms such as state and Federal laws and a post-delisting monitoring plan are in place to successfully manage threats and ensure that the population remains stable or increases after it is delisted

5. Continue research and monitoring to understand secondary threats and how they interact; based on the results, improve our ability to address multiple threats acting concurrently with feasible and effective management actions

4.2.8 Guadalupe Fur Seal

Guadalupe fur seals were listed as threatened under the ESA on December 16, 1985 (50 Fed. Reg. 51252). Guadalupe fur seals were once found throughout Baja California, Mexico and along the California coast. Currently, the species breeds mainly on Guadalupe Island, Mexico, off the coast of Baja California (Figure 33). A smaller breeding colony, discovered in 1997, appears to have been established at Isla Benito del Este in the San Benito Archipelago, Baja California, Mexico (Belcher and Lee Jr 2002).

Life History

Guadalupe fur seals are medium-sized, sexually dimorphic otariids (eared seal family) that 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 24 days (average of 14 days). Tracking data show that adult females spend 75% of their time at sea, and 25% 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 kHz 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 (Moore and Schusterman 1987; Babushina et al. 1991). Based on these studies, Guadalupe fur seals would be expected to hear sounds within the ranges of 50 Hz–75 kHz in air and 50 Hz–50 kHz in water.

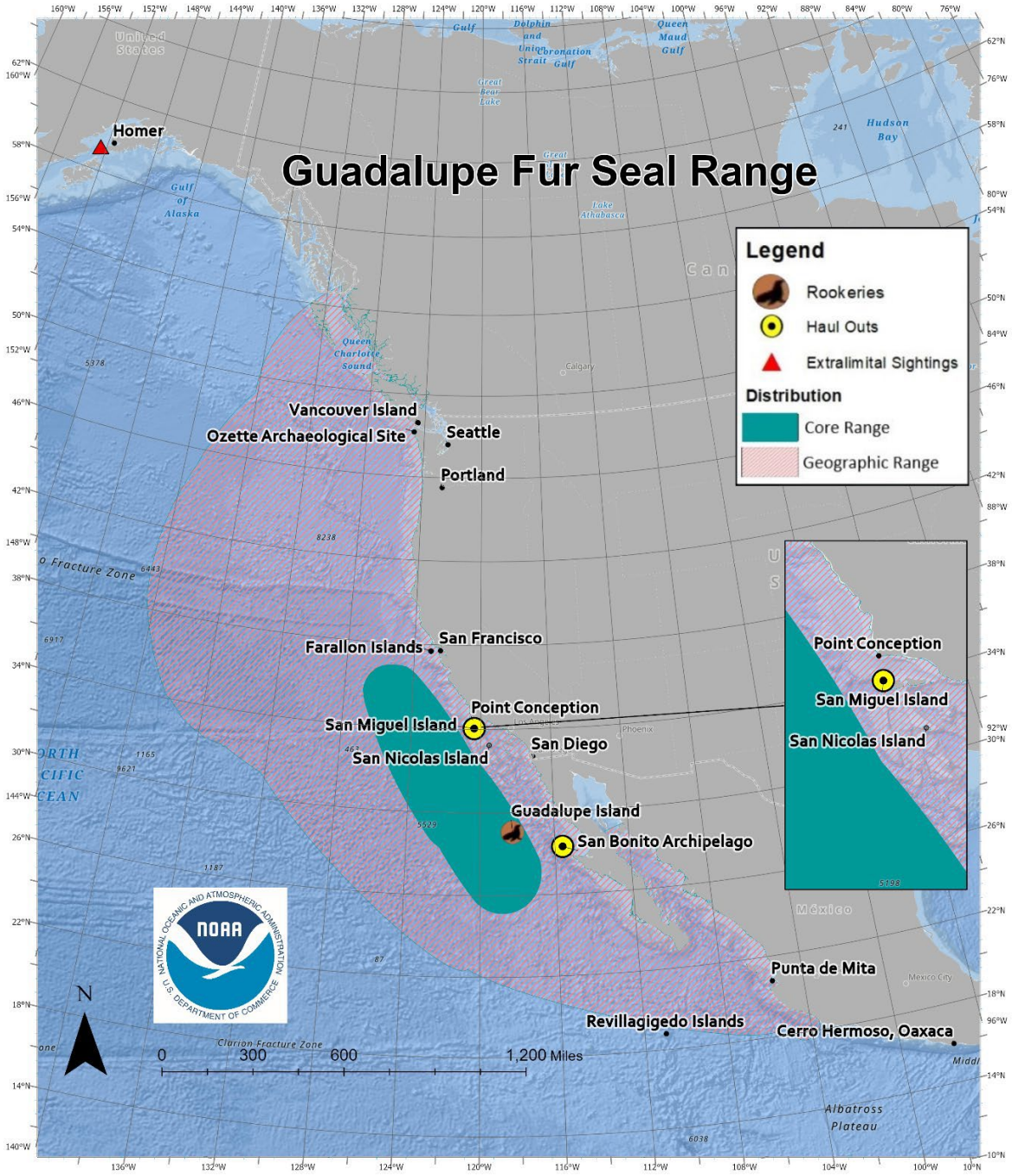


Figure 33. Guadalupe Fur Seal Range off the West Coast of the United States and Mexico (McCue 2021)

Population Dynamics

The best estimate of Guadalupe fur seal abundance is the Juárez-Ruiz et al. (2022) 2019 estimate of 63,850 animals (Carretta et al. 2025). The annual growth rate of 8.4% estimated for the 1991–2019 period is higher than the 5.9% estimated for 1984–2013 (García-Aguilar et al. 2018), indicating a steep increase in pup production in recent years (Lotze et al. 2011; Juárez-Ruiz et al.

2022). The status of the Guadalupe fur seal population has improved significantly since the species was originally protected under the ESA in 1985 (McCue 2021). Their breeding grounds are almost entirely on Guadalupe Island, off the Pacific coast of Mexico (Figure 33), which lies just outside of SOCAL RC. Guadalupe fur seals have recently re-colonization the islands comprising the San Benito Archipelago, and a small number of Guadalupe fur seals have also been reported on the northern Channel Islands off California. Existing legal protections and likely favorable environmental conditions have supported the species growth and productivity at the primary rookery at Guadalupe Island through at least 2010, and the species has expanded into its historical range (McCue 2021).

Production of Guadalupe fur seal pups has increased exponentially during the last three decades without reaching an asymptote, suggesting that the species is still in a recovery phase. Juárez-Ruiz et al. (2022) evaluated the effects of sea surface temperature anomalies in the southern California Current Ecosystem on annual Guadalupe fur seal pup production. Their ecological model shows that the difference between the expected number of pups, based on the 29-year growth rate trend, and the observed counts responds to interannual changes in the oceanic environment within the pregnant females' foraging habitat, indicated by sea surface temperature anomalies. Sea surface temperature anomalies of less than −0.2 ◦C and between 0.6 and 1.38 ◦C increased pup production in the subsequent breeding season, whereas normal to slightly warm (−0.17 to 0.6◦C) and extreme anomalies (>1.4◦C) decreased pup counts (Juárez-Ruiz et al. 2022).

However, since 2010, a number of potential impacts to the population have occurred, including dramatic shifts in species distribution and abundance and the mass strandings events along the U.S. West Coast described above (McCue 2021). In 2015-2016, Guadalupe fur seal strandings totaled approximately 175 animals along the coast of California (compared with approximately 10 animals annually in prior years), and NMFS declared an Unusual Mortality Event (Carretta 2021b). Starting in 2005 through 2016, 169 Guadalupe fur seals stranded in Washington and Oregon, involving two designated UMEs. Detailed necropsies, histopathology (n = 93), and epidemiological analysis found three main causes of death: emaciation (44%), trauma (29%), and infectious disease (19%) and the factors associated with overall strandings and emaciation (D'Agnese et al. 2020). In July 2019, NMFS also declared a UME off the coasts of Oregon and Washington due to unusually high stranding events. The Guadalupe fur seal UME, which was in place from January 1, 2015 to September 2, 2021, resulted in a total of 715 seal strandings in California, Oregon, and Washington combined. Most stranding were weaned pups and yearlings (9–15 months old), with both live and dead strandings occurring.

Guadalupe fur seal pup mortality rates from 2013-2016 fluctuated between 5.6-14.7% (Gálvez et al. 2023). The leading causes of pup mortality during this period included emaciation (49%), trauma (24%), and infectious disease (8%); 11% were undetermined (Gálvez et al. 2023). Results from Gálvez et al. (2023) suggest that the northeast Pacific marine heatwave in 2014-2016 contributed to higher rates of pup emaciation mortality (i.e., 54% in 2015 as compared to 9% in 2013).

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 2021). 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).

Guadalupe fur seals can be expected to occur in both deeper waters of the open ocean and coastal waters within the action area (Hanni et al. 1997b; Norris 2019). Satellite tagging data from animals leaving Guadalupe Island indicate that Guadalupe fur seals primarily use habitats offshore of the continental shelf between 50 and 300 km from the U.S. West Coast, with approximately one-quarter of the population foraging farther out and up to 700 km offshore (Norris 2017; Norris 2019). Females with pups are generally restricted to rookery areas because they must return to nurse their pups (Gallo-Reynoso et al. 2008). Satellite tags have documented the movement of females without pups at least as far as 1,300 km north of Guadalupe Island (approximately Point Cabrillo in Mendocino County, California; (Norris 2019). Adult males have not been tagged but typically undertake some form of seasonal movement either after the breeding season or during the winter, when prey availability is reduced (Arnould 2009). Satellite-tagged juvenile males appear to have more variable movement patterns than females. Although most remained within 600 km of Guadalupe Island, only one of 10 satellite tagged males traveled north of Point Cabrillo, California (Norris 2017). The recent sighting of a juvenile male Guadalupe fur seal in the Galapagos Islands (Páez-Rosas et al. 2020), is also consistent with the wide range of the species from the rookeries off Mexico.

Status

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. In many respects, the Guadalupe fur seal population has significantly recovered from the population status it had when the species was originally listed (McCue 2021). Existing legal protections and likely favorable environmental conditions have supported the species growth and productivity at the primary rookery at Guadalupe Island through at least 2010, and the species has expanded into its historical range (McCue 2021). However, since 2010, a number of potential impacts to the population have occurred, including dramatic shifts in species distribution and abundance and mass strandings and mortalities of Guadalupe fur seals and other species related to prey limitation (McCue 2021). Since Guadalupe fur seals have only one rookery where greater than 99% of the pups are born, they may be especially vulnerable to threats if they were to occur at this rookery, with potentially catastrophic impacts on the population (McCue 2021).

Critical Habitat

NMFS has not designated critical habitat for Guadalupe fur seals.

Recovery Planning

NMFS has not prepared a Recovery Plan for Guadalupe fur seals.

4.2.9 Hawaiian Monk Seal

The Hawaiian monk seal was listed as endangered on November 23, 1976. This species is a large phocid ("true seal") that inhabits the Northwestern Hawaiian Islands and Main Hawaiian Islands (Figure 34).

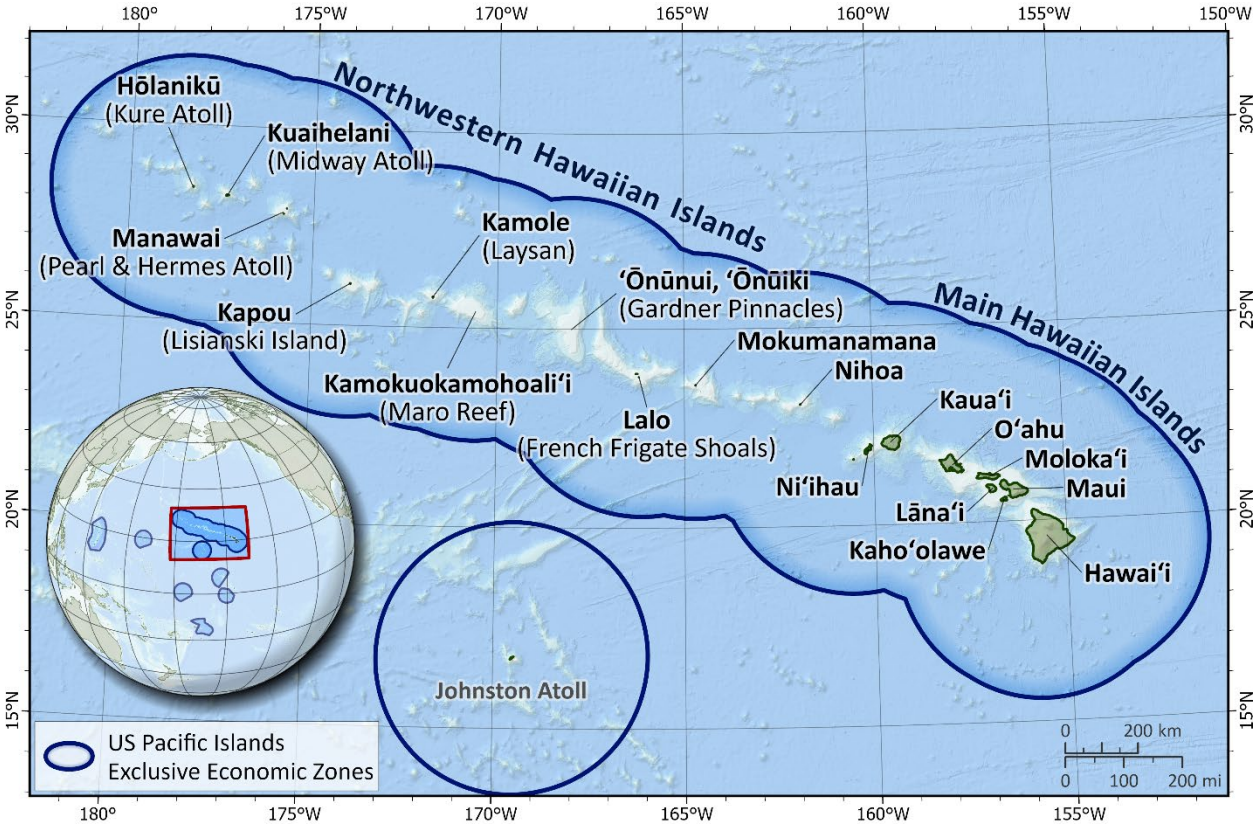


Figure 34. Hawaiian Archipelago and Range of the Hawaiian Monk Seal (Source: NMFS Pacific Islands Regional Office, Protected Resources Division)

Life History

Hawaiian monk seals can live, on average, 25 to 30 years. Sexual maturity in females is reached around five years of age and it is thought to be similar for males but they do not gain access to females until they are older. They have a gestation period of ten to eleven months, and calves nurse for approximately one month while the mother fasts and remains on land. After nursing, the mother abandons her pup and returns to the sea for eight to ten weeks before returning to beaches to molt. Males compete in a dominance hierarchy to gain access to females (i.e., guarding them on shore). Mating occurs at sea, however, providing opportunity for female mate choice. Monk seals are considered foraging generalist that feed primarily on benthic and demersal prey such as fish, cephalopods, and crustaceans. They forage in subphotic zones either

because the areas host favorable prey items or because these areas are less accessible by competitors (Parrish et al. 2000).

Hearing

Sills et al. (2021) measured underwater auditory detection thresholds in a male Hawaiian monk seal, and the range of most sensitive hearing was between 0.2 and 33 kHz. Peak hearing sensitivity of 73 dB re: 1 µPa was observed at 1.6 kHz. The audiogram for this individual was similar but narrower and elevated compared to the hearing group (phocid carnivores in water, PCW) composite audiogram used to assess impacts to this species. Ruscher et al. (2021) studied in-air hearing abilities of a Hawaiian monk seal. The behavioral audiogram indicates a functional hearing range from 0.1-33 kHz and poor sensitivity, with detection thresholds above 40 dB re: 20 µPa. The reduced terrestrial hearing ability of this individual, as compared to available auditory data for a northern elephant seal (*Mirounga angustirostris*), suggests that hearing in Monachinae seals differs from that of the more sensitive Phocinae seals (Ruscher et al. 2021; Ruscher et al. 2025).

Population Dynamics

The entire range of the Hawaiian monk seal is located within U.S. waters. In addition to a small but growing population found on the main Hawaiian Islands, there are six main breeding subpopulations in the northwestern Hawaiian Islands identified as: French Frigate Shoals (Lalo), Laysan Island (Kamole), Lisianski Island (Kapou), Pearl and Hermes Atoll (Manawai), Midway Atoll (Kuaihelani), and Kure Atoll (Hōlanikū). To a lesser extent, some seals also occur at Necker Island (Mokumanamana), and Nihoa Island.

The best official estimate of the total Hawaiian monk seal population size is 1,605; 95% confidence interval from 1,512-1,743 (NOAA 2023; Carretta et al. 2025). About 75% of these seals are from the northwestern islands that make up the Papahānaumokuākea Marine National Monument, with the remaining 25% from the main Hawaiian Islands. The population has grown at an average annual rate of about 2% from 2013-2022 (Carretta et al. 2023b; NMFS 2023e; NOAA 2023).

Genetic analysis indicates the species is a single panmictic population, thus warranting a single stock designation (Schultz et al. 2011). Genetic variation among monk seals is extremely low and may reflect a long-term history at low population levels and more recent human influences (Kretzmann et al. 2001; Schultz 2009). In addition to low genetic variability, studies by (Kretzmann et al. 1997) suggest the species is characterized by minimal genetic differentiation among sub-populations and, perhaps some naturally occurring local inbreeding. The potential for genetic drift should have increased when Hawaiian monk seal numbers were reduced by European harvest in the nineteenth century, but any tendency for genetic divergence among sub-populations is probably mitigated by the inter-island movements of seals.

Status

Hawaiian monk seals were once harvested for their meat, oil, and skins, leading to extirpation in the main Hawaiian Islands and near-extinction of the species by the twentieth century (Hiruki and Ragen 1992; Ragen 1999). The species partially recovered by 1960, when hundreds of seals were counted on northwestern Hawaiian Islands beaches. Since then, however, the species has declined in abundance. Though the ultimate cause(s) for the decline remain unknown threats include: food limitations in northwestern Hawaiian Islands, entanglement in marine debris, human interactions, loss of haul-out and pupping beaches due to erosion in northwestern Hawaiian Islands, disease outbreaks, shark predation, male aggression towards females, and low genetic diversity. Annual human-caused mortality or serious injury, based on reported strandings, for the 5-year period 2018-2022 was at least 4.8 animals, including fishery-related mortality in nearshore gillnets, hook-and-line gear, and mariculture (≥ 2.2/year), intentional killings and other human-caused mortalities (≥ 0.8/year), entanglement in marine debris (≥ 0.4/year), and deaths due to toxoplasmosis (≥ 1.4/year)(Carretta 2021b) (Carretta et al. 2025). These are likely underestimates since many strandings are not observed or reported. With only approximately 1,605 individuals remaining, the species' resilience to further perturbation is low. Since the population is so small there is concern about long-term maintenance of genetic diversity making it quite likely that this species will remain endangered for the foreseeable future.

Critical Habitat

Designated critical habitat was found to not likely be adversely affected by the action, as described in Section 4.1.3.

Recovery Planning

See the 2007 Final Recovery Plan (NMFS 2007b) for the Hawaiian monk seal for complete down listing/delisting criteria for each of the four following recovery goals:

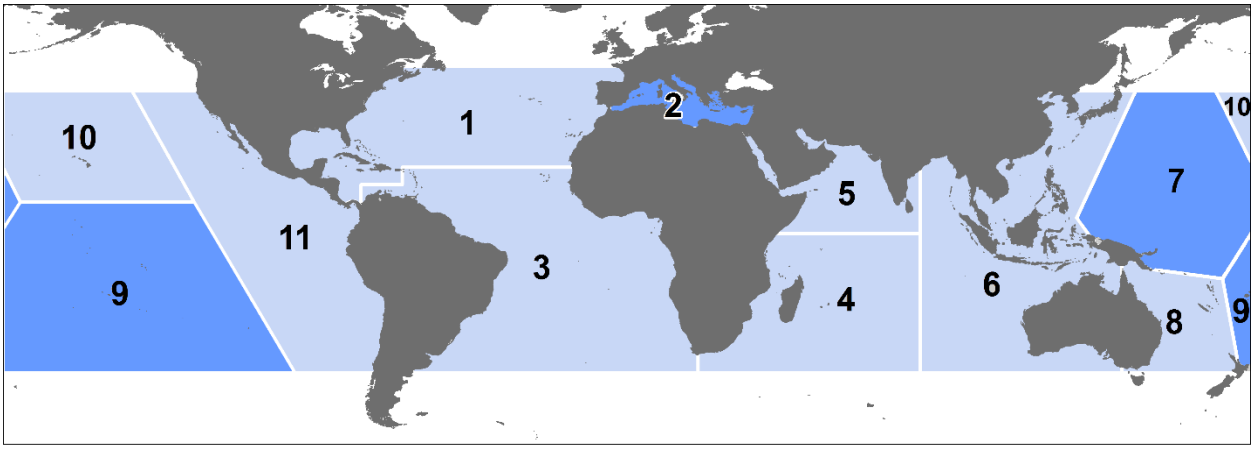
1. Improve the survivorship of females, particularly juveniles, in sub-populations of the northwestern Hawaiian Islands.
2. Maintain the extensive field presence during the breeding season in the northwestern Hawaiian Islands.
3. Ensure the continued natural growth of the Hawaiian monk seal in the main Hawaiian Islands by reducing threats including interactions with recreational fisheries, disturbance of mother-pup pairs, disturbance of hauled out seals, and exposure to human domestic animal diseases.
4. Reduce the probability of the introduction of infectious diseases into the Hawaiian monk seal population.

4.2.10 Green Sea Turtle – Central North Pacific and East Pacific DPSs

The green sea turtle was listed range-wide under the ESA on July 28, 1978. On April 6, 2016, NMFS listed 11 distinct population segments (DPSs) of green sea turtles as threatened or endangered under the ESA. Eight DPSs are listed as threatened: Central North Pacific, East

Indian-West Pacific, East Pacific, North Atlantic, North Indian, South Atlantic, Southwest Indian, and Southwest Pacific. Three DPSs are listed as endangered: Central South Pacific, Central West Pacific, and Mediterranean. The Central North Pacific and East Pacific DPSs occur within the action area.

The green sea turtle is globally distributed and commonly inhabits nearshore and inshore waters, occurring throughout tropical, subtropical and, to a lesser extent, temperate waters (Figure 35).



Threatened (light blue) and endangered (dark blue) green turtle DPSs:
1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. East Pacific.

Figure 35. Map Depicting Range and DPS Boundaries for Green Sea Turtles

Life History

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 pounds (159 kg). Age at first reproduction for females is twenty to forty years. Green sea turtles lay an average of three nests per season with an average of 100 eggs per nest. The remigration interval (i.e., return to natal beaches) is two to five years. Nesting occurs primarily on beaches with intact dune structure, native vegetation and appropriate incubation temperatures during summer months. After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat jellyfish, sponges and other invertebrate prey.

Vocalization and Hearing

Sea turtles lack 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. 1969b). The functional morphology of the sea turtle ear is poorly understood, however it has been suggested that aquatic turtle ears are adapted for hearing underwater sounds and vibration (Lenhardt 1982; Ketten 2008; Christensen-Dalsgaard et al. 2012).

The role of hearing in sea turtles remains unclear, however they likely use sound for navigation, locating prey, avoiding predators, and environmental awareness (Piniak et al. 2016). Electrophysiological and behavioral studies of sea turtle hearing have documented that most species of sea turtles can detect low-frequency acoustic and/or vibratory stimuli underwater and in air (Ridgway et al. 1969b; Bartol et al. 1999; Bartol and Ketten 2006; Dow Piniak et al. 2012; Martin et al. 2012; Piniak 2012; Lavender et al. 2014; Piniak et al. 2016). Sea turtles generally are most sensitive to underwater and aerial sounds below 1,000 Hz, though variation in sensitivity and frequencies of maximum sensitivity exist between species and age classes (see Piniak, 2012 for species comparisons). Dow Piniak et al. (2012) measured hearing of leatherback sea turtle hatchlings in water and in air using an electrophysiological method (i.e., auditory evoked potentials or responses), and determined that hatchlings are able to detect low-frequency sounds between 50 Hz and 1200 Hz in water and 50 Hz and 1600 Hz in air, with best sensitivity between 100 and 400 Hz in water (lowest sound pressure level detected: 84 dB re: 1 µPa-rms at 300 Hz) and 50 and 400 Hz in air (lowest sound pressure level detected: 62 dB re: 20 µPa-rms at 300 Hz).

Sea turtles are also known to behaviorally respond to low-frequency sounds, lending supporting evidence that sea turtles detect and respond to these sounds when present in their environment. Studies focused on responses of sea turtles to impulsive sounds (*e.g.*, seismic airguns) documented behavioral responses such as increased swimming, diving, and aversion (O'Hara and Wilcox 1990; McCauley et al. 2000; DeRuiter and Doukara 2012). For example, McCauley et al. (2000) observed that one loggerhead sea turtle and one green turtle changed their behavior in the presence of sounds produced by airguns at a distance of 2 km and at 1 km, with received sound pressure levels of 166 dB re: 1 µPa-m and 175 dB re: 1 µPa, respectively. The sea turtles' response was consistent and above the sound pressure level of 166 dB re: 1 µPa, with the sea turtles noticeably increasing their swimming activity. Above 175 dB re: 1 µPa, their behavior became more erratic, possibly indicating agitation (McCauley et al. 2000).

Limited information is available on sea turtle vocalizations; however recent studies suggest that some sea turtle species and age classes may produce sounds in air. Whether these sounds are produced and used for communication or are produced as a by-product of physical activity (air movement or respiration during nesting or emerging from the nest) remains unclear (Mrosovsky 1972; Cook and Forrest 2005; Ferrara et al. 2014; McKenna et al. 2019; Monteiro et al. 2019).

Population Dynamics

Our ability to estimate sea turtle population abundance, in general, is complicated by the life history of these species. Data collected at nesting beaches are often the best available data but do not provide information on life stages away from the nesting beaches (i.e., immature and mature males and immature females). There is general agreement that the number of nesting females

provides a useful index of the species' population size and stability at this life stage, For example, Vandeperre et al. (2019) found that relative abundance of loggerheads in the Azores tracked annual nest abundance at source rookeries in Florida when adjusted for a 3-year lag. This concurrence of abundance patterns indicates that recruitment to the oceanic stage is more dependent on nest abundance at source rookeries than on stochastic processes derived from short term climatic variability (Vandeperre et al. 2019). However, adult nesting females often account for less than 1% of total sea turtle population numbers, and there are doubts about the ability to estimate the overall population size from this index (Bjorndal et al. 2005).

Additionally, standardized nesting surveys are difficult to maintain over many, consecutive years and at all nesting beaches. Here we provide data that have been consistently collected using a standardized monitoring approach over the most recent available remigration interval, providing reasonable certainty that such data are representative of recent nesting at the identified beach. Although some data may have been collected at other nesting beaches, monitoring has not been recent, consistent, or standardized, thus limiting our certainty of these data; therefore data from those nesting beaches cannot be used to calculate abundance.

The NMFS 2015 5-year status review estimated the Central North Pacific DPS to have 3,846 nesting females, which nest across 12 sites (Figure 36), with 96% nesting at the largest site of East Island, French Frigate Shoals, Hawaii (Seminoff et al. 2015). Nesting abundance at East Island, French Frigate Shoals increased at an estimated 4.8% annually from the mid-1970's through 2012 (Seminoff et al. 2015). In 2018, Hurricane Walaka demolished East Island, which was the index beach for this DPS. Given the instability of East Island, in 2019 Tern Island was officially identified as the new index site at French Frigate Shoals for the Central North Pacific green sea turtle population. From 2017 – 2024, green sea turtle nesting on Tern Island fluctuated between 86 females in 2018 and 645 females in 2021, with an average of 367 females nesting annually (NMFS unpublished). The most recent estimate of the Central North Pacific green sea turtle population is expected to continue increasing at a rate of 5.4% annually for the foreseeable future (Balazs et al. 2015). The estimated mean total population for this DPS is 682,296 (range 60,965 to 1,145,988) (NMFS 2023f). Overall, the Central North Pacific DPS has a relatively low level of genetic diversity and stock sub-structuring (Seminoff et al. 2015).

Green turtles in the East Pacific DPS are found from the California/Oregon border south to central Chile. Major nesting sites occur at Michoacán, Mexico, and the Galápagos Islands, Ecuador. Smaller nesting sites are found on the Pacific coast of Costa Rica, and in the Revillagigedos Archipelago, Mexico (Figure 37). Scattered nesting occurs in Columbia, Ecuador, Guatemala and Peru (Seminoff et al. 2015). There are thirty-nine nesting sites for the East Pacific DPS, with an estimated 20,062 nesting females. The largest nesting site is at Colola, Mexico, which hosts 58% of the nesting females for the DPS (Seminoff et al. 2015). There are no estimates of population growth for the DPS as a whole. Only one nesting site in the East Pacific DPS at Colola, Mexico, has sufficient long-term data to determine population trends. Data analysis indicates that the nesting population at Colola is increasing and is likely to

continue to do so (Seminoff et al. 2015). The estimated mean total population for the East Pacific DPS, individuals greater than one year old, is 3,580,207 (range 2,473,546 to 5,814,512) (NMFS 2023f).

In southern California, East Pacific DPS green turtles have been seen in more areas and in greater numbers since 2014 than before as a result of population recovery over the last two decades attributed to protection at nesting beaches and foraging areas (Hanna et al. 2021; Massey et al. 2023). Hanna et al. (2021) used citizen-based science to study green turtles near La Jolla Shores, California. Based on underwater photographs from members of the local dive/snorkel community, the authors calculated a mean minimum residency duration of 424 days, confirming the presence of a recently established resident aggregation of green turtles near La Jolla Shores. In this study, green turtles were seen foraging in water temperatures as low as 15.8°C, the lowest recorded temperature for foraging green turtles documented in the literature (Hanna et al. 2021). Based on citizen-based science from 2013-2021, Massey et al. (2023) documented the northernmost resident population of East Pacific DPS green sea turtles in the near the mouth of the San Gabriel River in Long Beach, California. This study established a year-round presence of a green sea turtle population along a 2.4-km (1.5 mile) stretch of the river, with the most green sea turtle activity near the Los Cerritos Wetlands and a power plant warm water effluent area (Massey et al. 2023).

Eguchi et al. (2020) investigated the effects of a power plant closure, on green turtle home ranges in South San Diego Bay using GPS-based telemetry. Core use areas (50% utilization distribution) nearly doubled in size after the power plant decommissioning, mostly due to increased core use areas during nighttime as turtles moved from the effluent channel to an area closer to seagrass pastures, a presumed foraging habitat. The authors note that the observed expansion of green turtle home ranges in San Diego may increase turtle−human interactions, such as boat strikes, within the foraging ground.

Maurer et al. (2024) found that green turtles in San Diego Bay select for eelgrass meadows (*Zostera marina*), particularly during the warmest months of the year and during daytime hours. The bulk of eelgrass beds in San Diego Bay are in the shallow water, southern portion of the bay where green sea turtles are typically found. Eelgrass areal coverage in south San Diego Bay increased from 2014 to 2020 (Associates 2020).

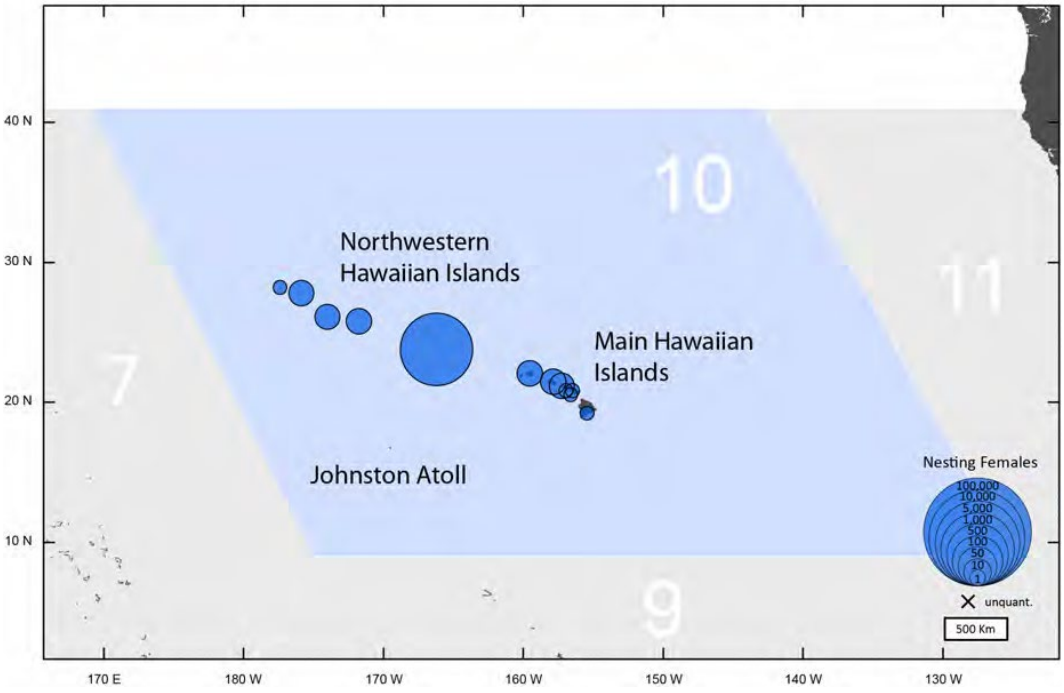


Figure 36. Nesting Distribution of Green Turtles in the Central North Pacific DPS (water body labeled '10'). Size of circles indicates estimated nester abundance (Seminoff et al. 2015).

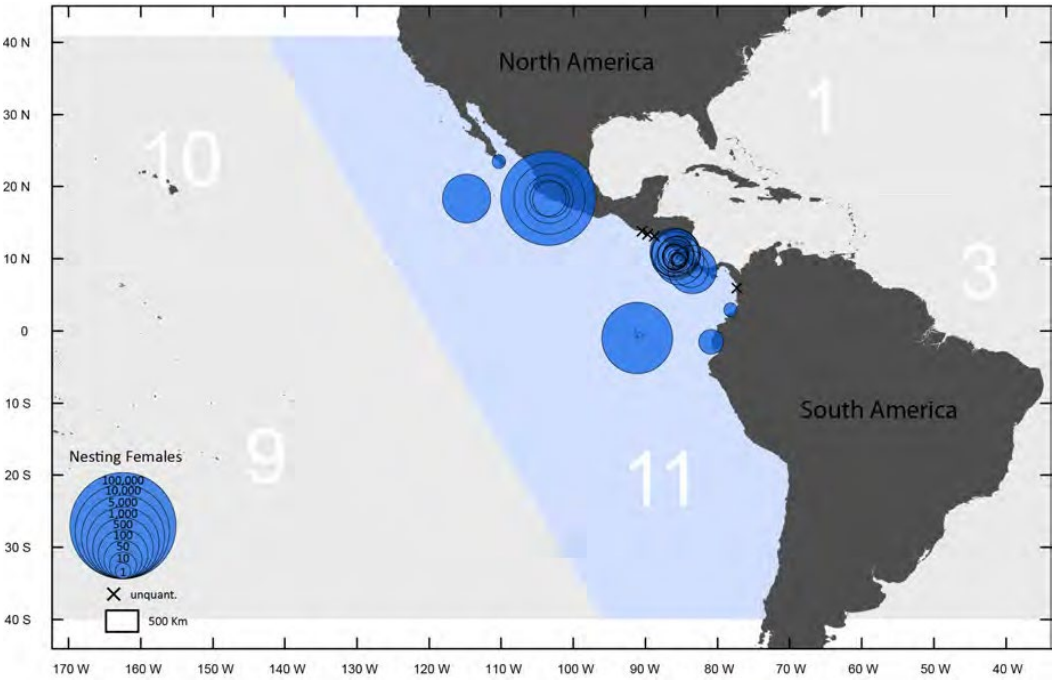


Figure 37. Nesting Distribution of Green Turtles in the East Pacific DPS (water body labeled '11'). Size of circles indicates estimated nester abundance (Seminoff et al. 2015).

Green turtle population genetic structure in the Pacific has largely been based on mitochondrial DNA. Based on examination of nuclear DNA across 20 Pacific green turtle rookeries, Roden et

al. (2023) found a defined East-West split consistent with previous studies based on mitochondrial DNA, except for some areas with evidence of possible male-mediated gene flow where breeding migration corridors of separate populations overlap with courtship areas. This may occur primarily along the margins of continents, including along Mexico and Central America in the East Pacific DPS (Roden et al. 2023).

Status

Once abundant in tropical and subtropical waters, green sea turtles worldwide exist at a fraction of their historical abundance, as a result of over-exploitation. Green turtles from the Central North Pacific DPS in the Hawaiian Archipelago were subjected to hunting pressure for subsistence and commercial trade, which was largely responsible for the decline in the region. Though the practice has been banned, there are still anecdotal reports of harvest. Incidental bycatch in fishing gear, ingestion of marine debris, and the loss of nesting habitat due to sea level rise are current threats to the population. Although these threats persist, the increase in annual nesting abundance, continuous scientific monitoring, legal enforcement and conservation programs are all factors that favor the resiliency of the DPS.

The population decline for the East Pacific DPS was primarily caused by commercial harvest of green turtles for subsistence and other uses (e.g., sea turtle oil as a cold remedy). Conservation laws are in place in several countries across the range of the DPS, but enforcement is inconsistent, limiting effectiveness. Incidental bycatch in commercial fishing gear, continued harvest, coastal development and beachfront lighting are all continuing threats for the DPS. The observed increases in nesting abundance for the largest nesting aggregation in the region (Michocán, Mexico), a stable trend at Galapagos, and record high numbers at sites in Costa Rica suggest that the population is resilient, particularly in Mexico.

Critical Habitat

On July 19, 2023, NMFS issued a proposed rule (88 Fed. Reg. 46376) to designate critical habitat for six DPSs of green sea turtle, including the Central North Pacific and East Pacific DPSs that occur within the HCTT action area.

The PBFs essential to the conservation of the Central North Pacific and/or East Pacific DPSs of green sea turtle are:

- Reproductive essential feature: From the mean high water line to 20 m depth, sufficiently dark and unobstructed nearshore waters adjacent to nesting beaches proposed as critical habitat by USFWS (see https://www.regulations.gov, Docket No. FWS-R4-ES-2022-0164), to allow for the transit, mating, and internesting of reproductive individuals and the transit of post-hatchlings. (Note: this feature does not apply to East Pacific DPS since no nesting occurs within U.S. jurisdiction for the population).
- Migratory essential feature: From the mean high water line to 10 km offshore, sufficiently unobstructed corridors that allow for unrestricted transit between foraging and nesting areas for reproductive individuals (Note: this feature does not apply to the

Central North Pacific DPS since NMFS did not identify a narrow, constricted migratory corridor for this population).

- Benthic foraging/resting essential feature: From the mean high water line to 20 m depth, underwater refugia and food resources (i.e., seagrasses, macroalgae, and/or invertebrates) of sufficient condition, distribution, diversity, abundance, and density necessary to support survival, development, growth, and/or reproduction (88 Fed. Reg. 46376).

Within the HCTT Action Area, NMFS is proposing critical habitat for the Central North Pacific DPS green sea turtle within waters under the jurisdiction of Joint Base Pearl Harbor (JBPHH) (Pearl Harbor Naval Defensive Sea Area, Naval Special Warfare training areas at Pearl City Peninsula and Lima Landing, Puuloa Underwater Range, And Barbers Point Underwater Range and Ewa Training Minefield), PMRF, and Navy Training Ranges in Southern California (SSTC and Coastal San Diego Training Ranges, select areas within San Diego Bay, and Naval Weapons Station Seal Beach).

The migratory corridor between San Diego Bay and the U.S./Mexico border provides a high value to the DPS because reproductive individuals (i.e., the life stage most directly linked to population growth and recovery) must use it to migrate between nesting beaches in Mexico and foraging areas in California (NMFS 2023c). Within the action area, Naval Weapons Station Seal Beach and San Diego Bay are considered high value foraging/resting critical habitat areas for the East Pacific DPS due the year-round presence of resident turtles in these areas (NMFS 2023c).

Recovery Planning

See the 1991 and 1998 recovery plans (NMFS and USFWS 1991; NMFS and USFWS 1998b) for the Pacific and East Pacific populations of green turtles for complete down-listing/delisting criteria for recovery goals for the species. Broadly, recovery plan goals emphasize the need to protect and manage nesting and marine habitat, protect and manage populations on nesting beaches and in the marine environment, increase public education, and promote international cooperation on sea turtle conservation topics.

4.2.11 Hawksbill Turtle

The hawksbill turtle was first listed under the Endangered Species Conservation Act and has been listed as endangered under the ESA since 1973. The species has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans (Figure 38).

Life History

Hawksbill turtles reach sexual maturity at 20 to 40 years of age. Based on skeletochronological analysis for hawksbills stranded along U.S. coastlines, Avens et al. (2021) predicted an adult sexual maturity age ranging from 15-25 years old, suggesting a younger average age at maturity for hawksbills in U.S. waters. Females return to their natal beaches every two to five years to nest and nest an average of three to five times per season. Clutch sizes are large (up to 250 eggs). Sex determination is temperature dependent, with warmer incubation producing more females. Brunson et al. (2022) analyzed hawksbill sea turtle strandings data collected between 1984 and

2018 in the Hawaiian Islands. Stranding events were biased towards females (female to male sex ratio of nearly 5:1), which may be indicative of the population as a whole (Brunson et al. 2022).

Hatchlings migrate to and remain in pelagic habitats until they reach approximately twenty two to twenty five centimeters in straight carapace length. As juveniles, they take up residency in coastal waters to forage and grow. As adults, hawksbills use their sharp beak-like mouths to feed on sponges and corals. Hawksbill sea turtles are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Satellite tagged turtles have shown significant variation in movement and migration patterns. Distance traveled between nesting and foraging locations ranges from a few hundred to a few thousand kilometers (Miller et al. 1998; Horrocks et al. 2001).

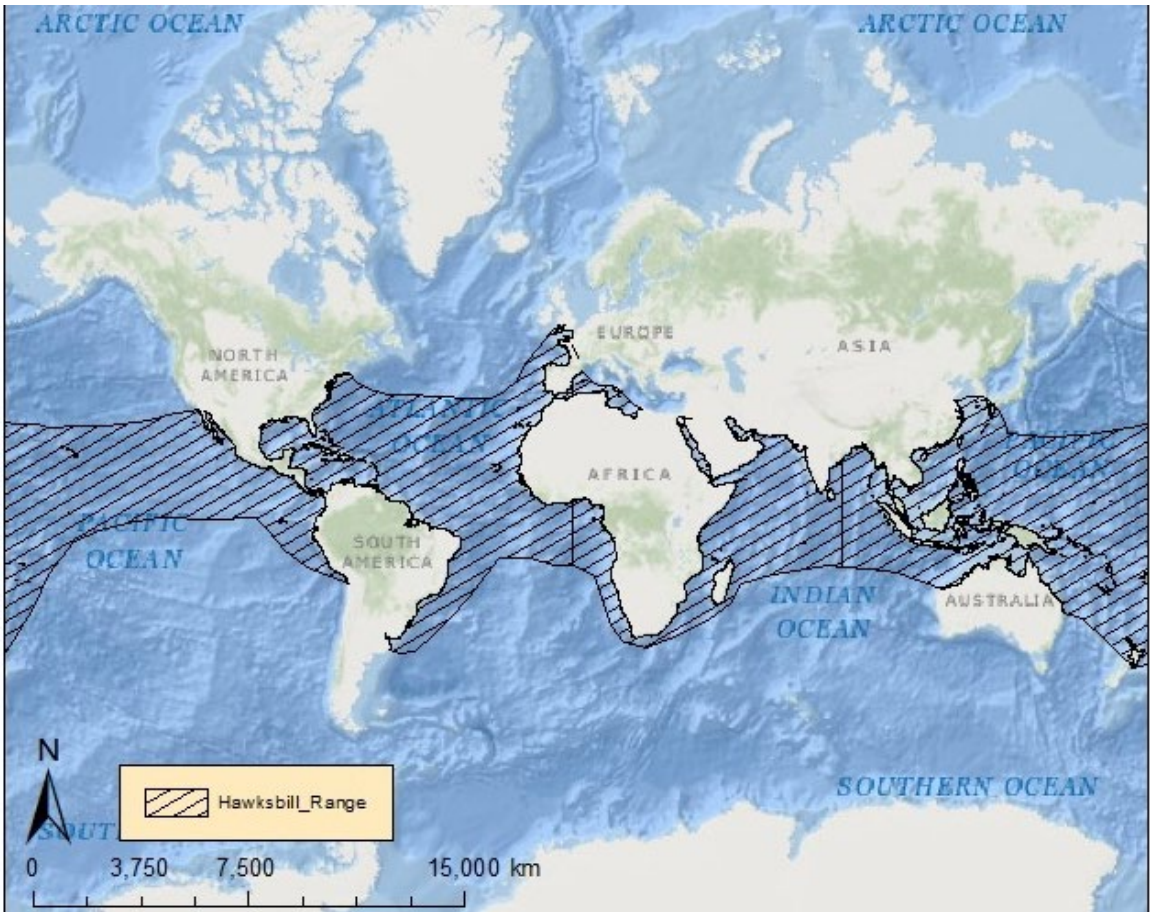


Figure 38. Map Identifying the Range of the Endangered Hawksbill Sea Turtle

Vocalization and Hearing

See Section 4.2.10 (green sea turtle status of species) above for a general discussion of sea turtle vocalization and hearing.

Population Dynamics

Surveys at 88 nesting sites worldwide indicate that 22,004 to 29,035 females nest annually (NMFS and USFWS 2013). In general, hawksbills are doing better in the Atlantic and Indian Ocean than in the Pacific Ocean, where despite greater overall abundance, a greater proportion of the nesting sites are declining. Long-term data on hawksbill turtle nesting indicate that among the sites for which historic trends could be assessed (n=63), all (100%) showed a decline during the long-term period from greater than 20 to 100 years ago. Among the 41 sites for which recent trend data are available, 10 (24%) were increasing, 3 (7%) were stable, and 28 (68%) were decreasing at the time of the last hawksbill 5-year review (NMFS 2013b).

From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased 15% annually; however, due to recent declines in nest counts, decreased survival at other life stages, and updated population modeling, this rate is not expected to continue (NMFS and USFWS 2013).

Nesting has been documented on four Hawaiian Islands, with the overwhelming majority of nesting females (78.4%) and nests (86.5%) recorded at four beaches along the southern coast of Hawaii Island (Gaos et al. 2021). Monitoring at a beach on Molokai Island in 2018 revealed numbers similar to the most important beaches on Hawaii Island (Gaos et al. 2021). Field monitoring conducted on the Hawaiian Islands between 1988 and 2018 documented an annual average of 14 nesting hawksbill females and 48 nests (Gaos et al. 2021). Both the number of hawksbill nesting females and nests in Hawaii have been positively trending since 2006 (Gaos et al. 2021).

Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. Gaos et al. (2020) used mitochondrial DNA to provide the first genetic characterization of nesting (n = 108) and foraging (n = 29) hawksbills in the Hawaiian Islands. They found strong differentiation ($F_{ST} > 0.238$, P-value < 0.001) between the Hawaii hawksbill nesting colony and those in the West and East Pacific, indicating the Hawaii nesting colony is demographically isolated and warrants recognition as a distinct management unit (Gaos et al. 2020). Genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the western Atlantic, where the vast majority of nesting has been documented (McClellan et al. 2010; Monzón-Argüello et al. 2010). Hawksbills in the Caribbean seem to have dispersed into separate populations (rookeries) after a bottleneck roughly 100,000 to 300,000 years ago (Leroux et al. 2012). All populations have declined historically, and the majority had declined recently; however, the exceptions are in the Sabah Turtle Islands in Malaysia where the population is stable and, as discussed above in Hawaii where data suggests the nesting populations are trending positive (Gaos et al. 2021).

The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific Oceans. In their oceanic phase, juvenile hawksbills can be found in Sargassum mats; post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Musick and Limpus 1997; Bjorndal and Bolten 2010).

Status

The greatest threats to hawksbill turtles are overharvesting of sea turtles and eggs, degradation of nesting habitat, and fisheries interactions. Adult hawksbill turtles are harvested for their meat and carapace, which is sold as tortoiseshell. Eggs are taken at high levels, especially in Southeast Asia where collection approaches 100% in some areas. In addition, lights on or adjacent to nesting beaches are often fatal to emerging hatchlings and alter the behavior of nesting adults. The species' resilience to additional perturbation is low.

Critical Habitat

On September 2, 1998, NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 Fed. Reg. 46693). Hawksbill sea turtle designated critical habitat does not overlap with the action area or any stressors resulting from the proposed action.

Recovery Planning

See the 1998 Recovery Plan for the U.S. Pacific populations (NMFS and USFWS 1998) of hawksbill sea turtles, for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the recovery plan:

- Identify important nesting beaches;
- Ensure long-term protection and management of important nesting beaches;
- Protect and manage nesting habitat; prevent the degradation of nesting habitat caused by seawalls, revetments, sand bags, other erosion-control measures, jetties and breakwaters;
- Identify important marine habitats; protect and manage populations in marine habitat;
- Protect and manage marine habitat; prevent the degradation or destruction of important [marine] habitats caused by upland and coastal erosion;
- Prevent the degradation of reef habitat caused by sewage and other pollutants;
- Monitor nesting activity on important nesting beaches with standardized index surveys;
- Evaluate nest success and implement appropriate nest-protection on important nesting beaches;
- Ensure that law-enforcement activities prevent the illegal exploitation and harassment of sea turtles and increase law-enforcement efforts to reduce illegal exploitation; and
- Determine nesting beach origins for juvenile and sub adult populations.

4.2.12 Leatherback Turtle

Leatherback sea turtles are listed as endangered under the ESA throughout their global range. The leatherback turtle has the most extensive global distribution of any reptile and is distributed throughout the oceans of the world from the equator to subpolar regions in both hemispheres Figure 39. Leatherback turtles spend the majority of their lives at sea, where they develop, forage, migrate, and mate, nesting on beaches on every continent except Europe and Antarctica, and several islands of the Caribbean and the Indo-Pacific (Eckert et al. 2012; NMFS and USFWS 2020a). Seven populations are currently recognized: (1) Northwest Atlantic; (2) Southeast Atlantic; (3) Southwest Atlantic; (4) Northeast Indian; (5) Southwest Indian; (6) West Pacific; and (7) East Pacific Ocean populations (NMFS and USFWS 2020a). The marine distribution for Pacific leatherback sea turtles extends north into the Sea of Japan, northeast and east across the North Pacific to the west coast of North America (predominantly off California), west to the South China Sea and Indonesian Seas, and south into the high latitude waters of the western South Pacific Ocean and Tasman Sea (Benson et al. 2011). There are two leatherback populations within the Pacific Ocean basin: West Pacific and East Pacific (Figure 40). For purposes of this opinion, we focus on the West Pacific population which occurs within the HCTT action area.

Life History

Avens et al. (2020) estimate a mean age at maturity of 17 years (range of 12 to 28 years) for Pacific leatherbacks (East and West populations combined). In Indonesia, the mean size of nesting females is 161 cm curved carapace length (CCL) with an observed minimum of 138 cm CCL (Hitipeuw and Maturbongs 2002; Lontoh 2014). The most comprehensive information on clutch frequencies for West Pacific leatherback nesting females is from Jamursba Medi and Wermon beaches (Indonesia), where Tapilatu et al. (2013) estimated an average clutch frequency of 5.5 (+/- 1.6, range 3-10). For the overall West Pacific population, Lontoh (2014) found evidence that clutch frequency and remigration interval may vary with foraging life history. Mean clutch frequency varied among individuals, but was higher for the North Pacific Transition Zone foraging group (6.1) than the Northeast Pacific (5.6) and South China Sea (4.8) foraging groups and increased with turtle size (curved carapace width) (Lontoh 2014). The range of remigration intervals was 1-6 years, and the mean (± SE) remigration interval was greater for the Northeast Pacific foraging group (4 ± 0.3 years) than the North Pacific Transition Zone foraging group (2.3 ± 0.4 years) and the South China Sea foraging group (2.9 ± 0.4 years) (Lontoh 2014). Mean clutch sizes for West Pacific leatherbacks range from 76 to 94 eggs per nest (Tapilatu and Tiwari 2007; Pilcher 2011; Jino et al. 2018).

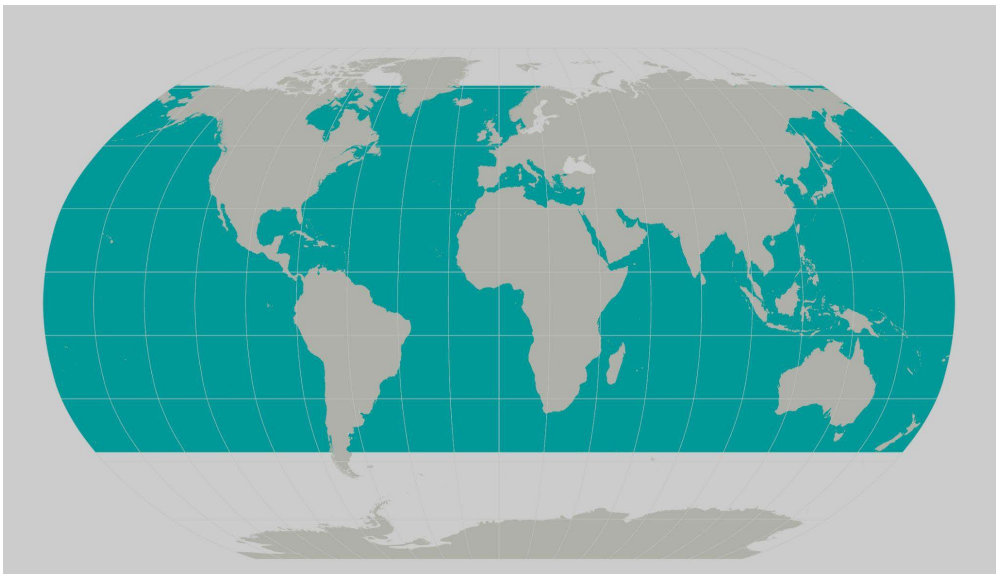


Figure 39. Map identifying the range of the endangered leatherback sea turtle. From NMFS https://www.fisheries.noaa.gov/species/leatherback-turtle#overview

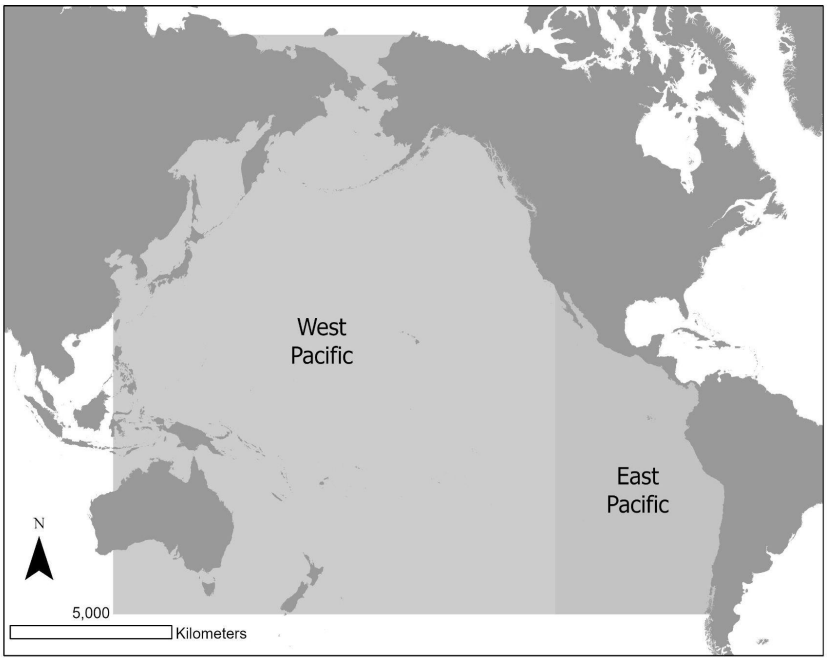


Figure 40. Map Identifying the Ranges of the East and West Pacific Leatherback Sea Turtle Populations

Leatherback turtles primarily feed on gelatinous zooplankton such as cnidarians (jellyfish and siphonophores) and tunicates (salps and pyrosomas) (Bjorndal 1997; NMFS and USFWS 1998a). These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight and energetic needs. Adult female leatherbacks weigh about 33% more on their foraging grounds than at nesting grounds, indicating that they probably catabolize fat reserves to fuel migration and subsequent reproduction (James et al. 2005; Wallace

et al. 2006). Adult female sea turtles must replenish their energy stores after nesting. Therefore, their remigration intervals (the time between nesting years) are dependent upon foraging success and duration (Hays 2000; Price et al. 2004).

Leatherback sea turtles feed from near the surface to depths exceeding 1,000 m, including nocturnal feeding on tunicate colonies within the deep scattering layer (Spotila 2004). Although leatherback sea turtles can dive deeper than any other reptile, most foraging dives are less than 80 m (Shillinger et al. 2011). Migrating leatherback sea turtles spend a majority of their time submerged and display a pattern of continual diving. They appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting continual foraging along the entire depth profile (Eckert et al. 1989). Typical dive durations averaged 6.9 to 14.5 minutes per dive, with a maximum of 42 minutes (Eckert et al. 1986).

Foraging areas are generally characterized by zones of upwelling, including off the edges of continents, where major currents converge, and in deep-water eddies (Saba 2013). Western Pacific winter-nesting females (described below) either travel southward into waters off Australia and New Zealand or remain in tropical island-associated Indonesian waters to forage post-nesting. Western Pacific summer-nesting females either move westward into the Sulawesi, Sulu, and South China Seas to forage primarily in Indonesian, Malaysian and Philippine waters or travel northeastward into waters of the western, central, or eastern temperate North Pacific, including the most distant foraging grounds off California, Oregon and Washington.

Vocalization and Hearing

See Section 4.2.10 (green sea turtle status of species) above for a general discussion of sea turtle vocalization and hearing.

Population Dynamics

Using the best available data for the West Pacific leatherback population (Fitry Pakiding, University of Papua, pers. comm. 2020) and a Bayesian steady-state model, Martin et al. (2020b) provided a median estimate of the total number of nesting females (i.e., over one, 3-year, remigration interval) at Jamursba Medi and Wermon beaches of 790 females, with a 95% credible interval of 666 to 942 females, as a snapshot of current abundance in 2017. We consider this the best available estimate of total adult female abundance at these two nesting beaches in 2017 (based on data from 2014 through 2017). To estimate the total number of nesting females from all nesting beaches in the West Pacific, we need to consider nesting at unmonitored or irregularly monitored beaches. As noted above, an estimated 50 to 75% of West Pacific leatherback nesting occurs at Jamursba-Medi and Wermon beaches (Dutton et al. 2007; NMFS and USFWS 2020a). Applying the conservative estimate of 75% to the Martin et al. (2020b) estimate of 790 nesting females at Jamursba Medi and Wermon beaches, the total number of nesting females in the West Pacific population would be 1,054 females with an overall 95% credible interval of 888 to 1,256 females. It should be noted that this estimate (i.e., 1,054) of nesting females for the West Pacific population based on more recent available information is an

update of the NMFS and USFWS (2020a) estimate (i.e., 1,277) which was based on a simple calculation that did not provide confidence or credible intervals.

Based on the estimates presented in Jones et al. (2012) for all Pacific populations, NMFS inferred an estimated West Pacific leatherback total population size (i.e., juveniles and adults) of 250,000 (95% confidence interval 97,000 to 535,000) in 2004. Based on the relative change in the estimates derived from Jones et al. (2012) and the more recent Martin et al. (2020b), NMFS estimates the current juvenile and adult population size of the West Pacific leatherback population is around 100,000 sea turtles (95% confidence interval 47,000 to 195,000 individuals).

The Western Pacific population exhibits low hatching success and decreasing nesting population trends due to past and current threats, which are likely to further lower abundance and increase the risk of extinction (NMFS and USFWS 2020a). The low estimated nesting female abundance of the West Pacific population places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (NMFS and USFWS 2020a). 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 and USFWS 2020a). Low site fidelity, which is characteristic of the species, results in the dispersal of nests among various beaches (NMFS and USFWS 2020a). This may help to reduce population level impacts from threats which may disproportionately affect one area over another, but may also place nests in locations that are likely unmonitored and not protected from human poaching or predation, thereby increasing threats to the population. Due to its small size, this population has restricted capacity to buffer such losses (NMFS and USFWS 2020a).

The median trend in annual nest counts estimated for Jamursba Medi nesting beaches from data collected from 2001-2017 was -5.7% annually (NMFS and USFWS 2020a). The median trend in annual nest counts estimated for Wermon nesting beaches from data collected from 2006 to 2017 (excluding 2013–2015 due to low or insufficient effort) was -2.3% annually (NMFS and USFWS 2020a). In the absence of population trend data on other leatherback life history stages, we consider these trends in annual nest counts an index of the population's growth rate. Results from Toha et al. (2025) indicate that Indonesian leatherbacks have low genetic diversity, which may limit the populations adaptive capacity for future environmental change.

Martin et al. (2020b) estimated the mean and median time until the West Pacific population declines to 50%, 25%, and 12.5% of its current estimated abundance. Results of this modeling effort indicate that the adult female portion of West Pacific leatherbacks nesting at Jamursba-Medi and Wermon beaches are predicted to decline to 50% of their current abundance in a mean of about 13 years (95% CI from 5 to 26 years) and to 25% of their current abundance in a mean of about 24 years (95% CI from 13 to 42 years).

Status

The leatherback turtle is an endangered species whose once large nesting populations have experienced steep declines in recent decades. The primary threats to leatherback turtles include fisheries bycatch, harvest of nesting females, and egg harvesting (Martínez et al. 2007). Because of these threats, once large rookeries are now functionally extinct, and there have been range-wide reductions in population abundance. Other threats include loss of nesting habitat due to development, tourism, and sand extraction. Lights on or adjacent to nesting beaches alter nesting adult behavior and are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea. Plastic ingestion is common in leatherback turtles and can block gastrointestinal tracts leading to death. Changes in environmental conditions may alter sex ratios (as temperature determines hatchling sex), range (through expansion of foraging habitat), and habitat (through the loss of nesting beaches, because of sea-level rise). The species' resilience to additional perturbation is low.

NMFS and USFWS (2020) found that all seven leatherback populations met the definition of high risk of extinction as a result of reduced nesting female abundance, declining nest trends (for all but the Southwest Atlantic DPS, which exhibits extremely low abundance), and numerous, severe threats. Based on the best available information presented above, the Pacific leatherback population is characterized by low resiliency and redundancy. With low abundance estimates in all countries where the species nests in both the Western Pacific and Eastern Pacific, leatherback sea turtles are at an extremely high risk of being extirpated from the Pacific Ocean. While leatherback abundance estimates are higher for other portions of its global range (e.g., Northwest Atlantic), all seven leatherback populations are currently at a high risk of extinction (NMFS and USFWS 2020a). Extirpation of Pacific leatherbacks would significantly contract the species' range, thus increasing the already high risk of extinction for the globally-listed entity.

Critical Habitat

Designated critical habitat was found to not likely be adversely affected by the action, as described in Section 4.1.3.

Recovery Planning

The 1998 Recovery Plan for U.S. Pacific Populations of the Leatherback (NMFS and USFWS 1998a) identifies the following criteria (all of which must be met) to consider de-listing:

- All regional stocks that use U.S. waters have been identified to source beaches based on reasonable geographic parameters;
- Each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over six years;
- Nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period;
- Existing foraging areas are maintained as healthy environments;

- Foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region;
- All Priority #1 tasks have been implemented (see 1998 Recovery Plan for details); and
- A management plan designed to maintain sustained populations of turtles is in place.

The primary recovery actions identified in the Pacific Leatherback Species in the Spotlight Five Year Action Plan (NOAA 2021) to support Pacific leatherback recovery include: reduce fisheries bycatch and in-water harvest, improve nesting beach protection and increase reproductive output, support in-water research and monitoring to inform conservation efforts, promote international cooperation, increase monitoring and research efforts, and encourage public engagement.

4.2.13 Loggerhead Turtle – North Pacific Ocean DPS

Loggerhead sea turtles are circumglobal, and are found in the temperate and tropical regions of the Indian, Pacific and Atlantic Oceans (Figure 41). The species was first listed as threatened under the ESA in 1978 (43 Fed. Reg. 32800). On September 22, 2011, NMFS designated nine DPSs of loggerhead turtles, including the North Pacific Ocean DPS, which is ESA-listed as endangered.

Life History

Mayne et al. (2020) estimated the average maximum lifespan of loggerhead turtles to be 62.8 years (± 3.7 years). Females make reproductive migrations every 3.3 years, laying 4.6 nests per year, with an average of 122 eggs per nest. Martin et al. (2020b) estimated the average age to maturity for North Pacific loggerheads at 37.9 years old. Nesting occurs on beaches, where warm, humid sand temperatures incubate the eggs. Temperature determines the sex of the turtle during the middle of the incubation period.

After emerging from their nests at night, hatchlings crawl seaward using visual cues, toward the brighter horizon of the open ocean and away from the dark, elevated silhouettes of dunes and vegetation (Pankaew and Milton 2018). After hatchlings enter the sea, they begin a 24 to 36-hour swim frenzy, orienting into waves to reach offshore currents (DuBois et al. 2021). Loggerhead turtles spend the post-hatchling stage in pelagic waters.

The juvenile stage is spent first in the oceanic zone and later in the neritic zone (i.e., coastal waters). While in their oceanic phase, loggerhead turtles undergo long migrations using ocean currents. Individuals from multiple nesting colonies can be found on a single feeding ground. Coastal waters provide important foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads. Loggerhead sea turtles are known to occur in areas where sea surface temperature ranges between 10°C and 28.7°C; however, mean sea surface temperatures, which are more indicative of preferred habitat, range between 16.3° and 24°C (Eguchi et al. 2018).

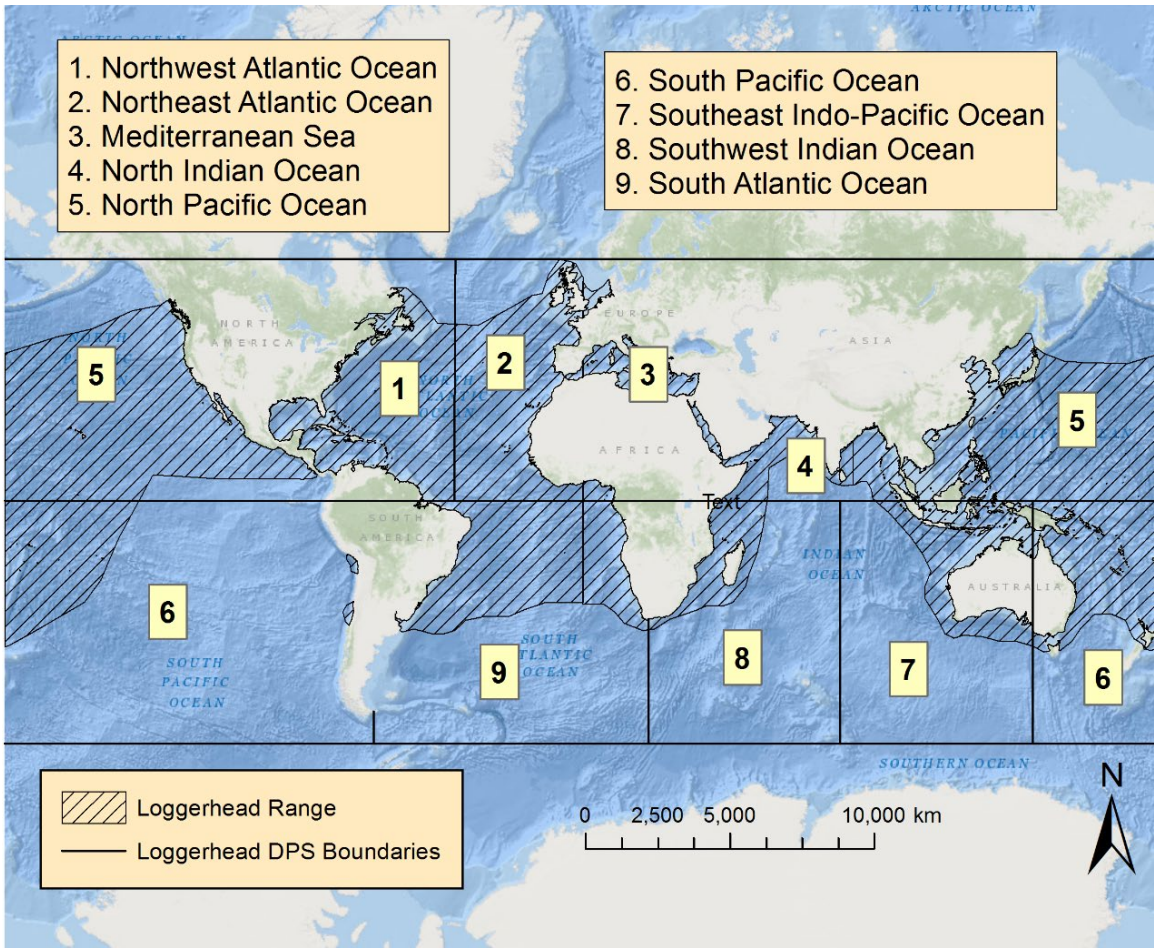


Figure 41. Map Identifying the Range and DPS Boundaries of the Loggerhead Sea Turtle

Vocalization and Hearing

See Section 4.2.10 (green sea turtle status of species) above for a general discussion of sea turtle vocalization and hearing.

Population Dynamics

Loggerheads return to their natal region for mating and nesting. All nesting for the North Pacific DPS occurs at sites in Japan (NMFS and USFWS 2020c). There is no loggerhead nesting on the Pacific coast of Mexico (Chapman and Seminoff 2016). Martin et al. (2020a) calculated a 2015 abundance "snapshot" of 4,541 nesting females for three nesting beaches (Inakahama, Maehama, and Yotsusehama on Yakushima) for which data were available. Because these beaches comprise approximately 52% of the total nesting population, the extrapolated 2015 total nesting abundance for the entire DPS is approximately 8,733 nesting females (USFWS 2020). North Pacific DPS loggerhead nesting likely increased between 1999 and 2012 at a rate of around 2.3% (95% CI: -1.1 to 15.6%) annually (Martin et al. 2020b). These trends will not necessarily represent the true growth rate of the population because annual nester counts, which represent the bulk of data on turtles, only represent a portion of the population, specifically, adult females.

However, as this is the only estimate of population growth available for this DPS, we consider it the best available scientific information to describe trends. Loggerhead abundance on foraging grounds off the Pacific Coast of the Baja California Peninsula, Mexico, was estimated to be 43,226 individuals (Seminoff et al. 2014).

Recent mitochondrial DNA analysis using longer sequences has revealed a more complex population sub-structure for the North Pacific Ocean DPS. Previously, five haplotypes were present, and now, nine haplotypes have been identified in the North Pacific Ocean DPS. This evidence supports the designation of three management units in the North Pacific Ocean DPS: 1) the Ryukyu management unit (Okinawa, Okinoerabu, and Amami), 2) Yakushima Island management unit and 3) Mainland management unit (Bousou, Enshu-nada, Shikoku, Kii and Eastern Kyushu) (Matsuzawa et al. 2016). Genetic analysis of loggerheads captured on the feeding grounds of Sanriku, Japan, found only haplotypes present in Japanese rookeries (Nishizawa et al. 2014).

Briscoe et al. (2021) analyzed data tracking loggerhead movement patterns in the North Pacific coupled with remotely sensed oceanographic conditions and novel methods in sequential sampling of annual bone growth layers. Based on their findings, they propose a new hypothesis regarding a possible physical mechanism for habitat connectivity and provide the first evidence of a "thermal corridor" that would allow for pulsed recruitment of loggerheads from the Central North Pacific to the North American coast. If a thermal corridor across the Eastern Pacific Barrier were to open more frequently in a warming ocean, it could result in an increase in the relative abundance of loggerheads and other highly migratory marine megafauna shifting from the Central North Pacific to foraging grounds along the eastern Pacific coast (Briscoe et al. 2021), including portions of the HCTT action area. Two major juvenile loggerhead foraging areas have been identified in the North Pacific Basin: Central North Pacific and off of Mexico's Baja California Peninsula.

In 2015, Eguchi et al. (2018) conducted an aerial survey of the southern California Bight, extending approximately from Pt. Conception to south of the U.S.-Mexico border and offshore as far as 123 N. Over 200 loggerheads were encountered during the survey, which coincided with anomalously high sea surface temperatures and a strong El Niño. Some sightings occurred within the Point Mugu Sea Range, while the majority were within Southern California Range Complex. Eguchi et al. (2018) estimated an offshore density of 0.24 loggerheads per km^2 for the large survey area that overlaps partially with the HCTT action area.

Loggerhead sea turtle presence off Southern California, including portions of the action area, has also been documented as bycatch in the California drift gillnet fishery. Observer data recorded 16 loggerhead captured as bycatch in this fishery from 1990-2006 during anomalously warm ocean conditions (Welch et al. 2019), but no observed captures from 2007-2019 (based on 1,474 observed sets). Thus, loggerhead bycatch in this fishery is extremely rare, particularly in recent years.

The higher density and abundance of loggerheads in the southern California Bight would only be expected during similar environmental conditions. A previous aerial survey (3,650 km [2,268 mi] of trackline) in the same region conducted in September and October 2011 during La Niña (anomalously cold) conditions encountered no loggerheads. In August 2018, a third aerial survey was conducted off of southern California and 2 loggerheads were spotted over more than 1,000 km (621.4 mi) of trackline (NMFS 2020b). Additionally, a line transect shipboard survey of the California Current Ecosystem was conducted from June to December 2018. During this survey, which occurred during mild El Niño conditions (i.e., from 0.5 to 0.9 degrees above normal), there were no sightings of loggerhead sea turtles (Moore 2021). It should be noted that the reliability of loggerhead sightings data from shipboard surveys is highly dependent on sea state. However, with the exception of the 2014/2015 marine heatwave, loggerhead sightings in the southern California Bight have been extremely rare in recent years. While loggerhead presence in the action area has likely been relatively rare during relatively normal offshore water temperatures in the past, increasing ocean temperatures associated with changing environmental conditions may, over time, allow foraging loggerheads to expand their range on a more regular basis (Eguchi et al. 2018).

Status

The North Pacific Ocean loggerhead DPS exhibits: low abundance (an estimated 8,733 nesting females); recently (i.e., less than one generation) increasing population growth (2.3% annually); and average demographic characteristics for the species (with the exception of low return rates for nesting females) (USFWS 2020). The DPS is characterized by limited genetic and environmental diversity (USFWS 2020). Though there is some diversity in nesting habitat, the majority of nesting occurs on small pocket beaches and beach armoring structures (designed to mitigate beach erosion) reduce available nesting habitat. All nesting beaches are at risk of inundation and rising sea level is likely to increase this risk (USFWS 2020). The DPS nests at multiple beaches throughout the Japanese Archipelago, and each life stage forages at multiple areas throughout the North Pacific Ocean. Though this redundancy would not protect the DPS from large-scale environmental changes, such as those likely to occur as a result of changing environmental conditions, it is likely to provide some insurance against local catastrophic events (USFWS 2020).

Based on the most recent 5-year review, the overall status of the North Pacific loggerhead DPS has not changed since it was listed as endangered in 2011 (USFWS 2020). The DPS continues to be endangered by intense (fisheries bycatch and changing environmental conditions) and numerous (habitat loss and modification, overutilization, and predation) threats acting on a small, subdivided population. Continued low abundance contributes to the North Pacific loggerhead DPS extinction risk because small populations are more likely than large ones to be extirpated as a result of stochastic events and threats (USFWS 2020).

Critical Habitat

No critical habitat has been designated for the North Pacific Ocean DPS loggerhead turtle.

Recovery Planning

Key recovery actions identified in the 1998 Recovery Plan (NMFS and USFWS 1998e) for U.S. Pacific Populations of the Loggerhead Turtle are:

- Reduce incidental capture of loggerheads by coastal and high seas commercial fishing operations;
- Establish bilateral agreements with Japan and Mexico to support their efforts to census and monitor loggerhead populations and to minimize impacts of coastal development and fisheries on loggerhead stocks;
- Identify stock home ranges using DNA analysis;
- Determine population size and status (in U.S. jurisdiction) through regular aerial or on water surveys; and
- Identify and protect primary foraging areas for the species.

4.2.14 Olive Ridley Turtle – Mexico's Pacific Coast Breeding Colonies and All Other Areas

The olive ridley sea turtle was listed under the ESA on July 28, 1978. The species was separated into two listing designations: endangered for breeding populations on the Pacific coast of Mexico, and threatened wherever found except where listed as endangered (i.e., in all other areas throughout its range).

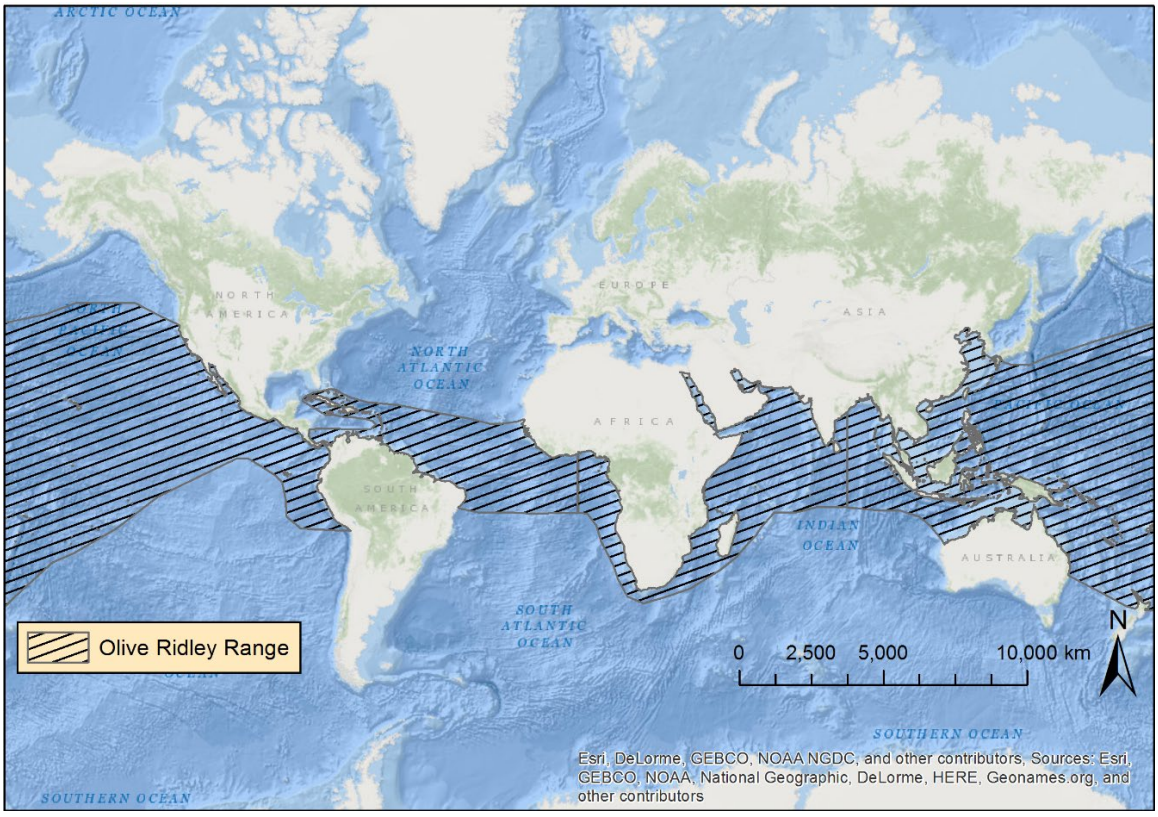


Figure 42. Map Identifying the Range of the Olive Ridley Sea Turtle

Life History

Olive ridley females mature at between 10 to 18 years of age. They lay an average of two clutches per season (three to six months in duration). The annual average clutch size is 100 to 110 eggs per nest. Olive ridleys commonly nest in successive years. Females nest in solitary or in arribadas, large aggregations coming ashore at the same time and location. As adults, olive ridleys forage on crustaceans, fish, mollusks, and tunicates, primarily in pelagic habitats.

Olive ridleys can nest throughout the year, but there tends to be a peak in nesting during the rainy season (Hart et al. 2014). Hatchlings emerge between 50 and 60 days after nesting (NMFS and USFWS 1998f).

Post-breeding behavior of olive ridleys in the eastern Pacific Ocean is unique in that they are nomadic, migrating across ocean basins. This contrasts with other sea turtle species, which typically migrate to a particular feeding ground after nesting. As adults, olive ridleys forage on crustaceans, fish, mollusks, and tunicates, primarily in pelagic habitats.

Vocalization and Hearing

See Section 4.2.10 (green sea turtle status of species) above for a general discussion of sea turtle vocalization and hearing.

Population Dynamics

Globally, olive ridley sea turtles can be found in tropical and subtropical waters in the Atlantic, Pacific and Indian Oceans. Major nesting arribada beaches are found in Nicaragua, Costa Rica, Panama, India and Suriname. There are two listed DPSs that occur in the action area, the endangered breeding colony populations on the Pacific coast of Mexico and the threatened populations from everywhere but the coast of Mexico endangered population. While the threatened olive ridley species is listed globally, there is genetic distinction between East and Indo-West Pacific olive ridleys from this DPS (NMFS and USFWS 2014).

The endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and USFWS 2014). The portion of the threatened population that nests in the East Pacific counts of adult olive ridley sea turtles on eastern Pacific nesting beaches similarly number at more than 1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Valverde et al. 2012; NMFS and USFWS 2014). The estimated minimum population size of olive ridley sea turtles (juveniles and adults) in the Western Pacific is 316,883 (NMFS 2023f).

The range of the endangered Pacific coast breeding population extends as far south as Peru and up to California. Olive ridley sea turtles from the Pacific coast breeding colonies nest on arribada beaches at Mismaloya, Ixtapilla and La Escobilla, Mexico. Solitary nesting takes place all along the Pacific coast of Mexico. There are six primary arribada nesting beaches in Mexico, the largest being La Escobilla, with about one million nesting females annually. Based on the current number of olive ridleys nesting in Mexico, three populations appear to be stable (Mismaloya,

Tlacoyunque, and Moro Ayuta), two increasing (Ixtapilla, La Escobilla) and one decreasing (Chacahua). Nesting trends in Mexico, where known, at non-arribada beaches are stable or increasing in recent years (NMFS and USFWS 2014).

The large arribada nesting populations in the eastern Pacific (excluding breeding populations in Mexico) have declined since the 1970s (NMFS and USFWS 2014). There are arribada nesting beaches in Nicaragua, Costa Rica and Panama. La Flor, Nicaragua had 521,440 effective nesting females in 2008 and 2009; Chacocente, Nicaragua had 27,947 nesting females over the same period (Gago et al. 2012). Two other arribada nesting beaches are in Nicaragua, Masachapa and Pochomil, but there are no abundance estimates available. Costa Rica hosts two major arribada nesting beaches; Ostional has between 3,564 and 476,550 turtles per arribada, and Nancite has between 256 and 41,149 turtles per arribada. Panama has one arribada nesting beach, with 8,768 turtles annually. Nesting at some arribada beaches continues to decline (e.g., Nancite in Costa Rica) and is stable or increasing at others (e.g., Ostional in Costa Rica).

Population growth rate and trend information for the threatened population of olive ridley sea turtles is difficult to discern, owing to its range over a large geographic area, and a lack of consistent monitoring data in all nesting areas, but some information is known (NMFS and USFWS 2014). Population trends at Nicaraguan arribada nesting beaches are unknown or stable (La Flor). Ostional, Costa Rica arribada nesting beach is increasing, while trends Nancite, Costa Rica, and Isla Cañas, Panama, nesting beaches are declining. For most solitary nesting beaches in the East Pacific Ocean, population trends are unknown, except for Hawaii Beach, Guatemala, which is decreasing.

Genetic studies have identified four main lineages for the olive ridley: east India, Indo-Western Pacific, Atlantic, and the eastern Pacific. In the eastern Pacific, rookeries on the Pacific Coasts of Costa Rica and Mexico were not genetically distinct, and fine-scale population structure was not found when solitary and arribada nesting beaches were examined. There was no population subdivision among olive ridleys along the east India coastline. Low levels of genetic diversity among Atlantic French New Guinea and eastern Pacific Baja California nesting sites are attributed to a population collapse caused by past overharvest (NMFS and USFWS 2014).

Sosa-Cornejo et al. (2021) evaluated nesting trends, hatching success, predation and poaching of olive ridley turtles on 2 beaches (Santuario Playa Ceuta and Playa Caimanero) in Northwestern Mexico based on long-term (30-40 year) datasets. They found a clear positive trend in the number of olive ridley sea turtle nests at both sites, which is consistent with trends found in other recent studies from the region. The predictive time-series model indicated that the numbers of nests will continue to increase through 2045, increasing three-fold at Santuario Playa Ceuta and six-fold at Playa Caimanero with respect to the last year (i.e., 2016) of monitoring (Sosa-Cornejo et al. 2021). Predation rates also appeared to increase over time at both nesting sites, while poaching rates only increased over time at Playa Caimanero. The rate of poaching of eggs at Santuario Playa Ceuta did not increase over time, likely because this beach is legally protected and far from urban settlements (Sosa-Cornejo et al. 2021). Average annual hatching success rates

over the time series ranged from 61-65%, but were highly variable from year to year at both nesting sites (Sosa-Cornejo et al. 2021).

Status

In the first half of the twentieth century, there was an estimated ten million olive ridleys nesting on the Pacific coast of Mexico (NMFS and USFWS 2014). Olive ridleys became targeted in fisheries in Mexico and Ecuador, which severely depleted the population; there was an estimated one million olive ridleys by 1969. Legal harvest of olive ridleys has been prohibited for decades, although illegal harvest still occurs. The population is subject to incidental capture in fisheries, exposure to pollutants, and changing environmental conditions. In spite of the severe population decline, the olive ridley breeding populations on the Pacific coast of Mexico appear to be resilient, evidenced by the increasing population trend (NMFS and USFWS 2014).

For threatened olive ridley turtles in all other areas, it is likely that solitary nesting locations once hosted large arribadas; since the 1960s, populations have experienced declines in abundance of 50 to 80% (NMFS and USFWS 2014). Many populations continue to decline. Olive ridley turtles continue to be harvested as eggs and adults, legally in some areas, and illegally in others. Incidental capture in fisheries is also a major threat. The olive ridley turtle is the most abundant sea turtle in the world; however, several populations are declining as a result of continued harvest and fisheries bycatch. The large population size of the range-wide population, however, allows some resilience to future perturbation (NMFS and USFWS 2014).

Critical Habitat

No critical habitat has been designated for the olive ridley sea turtle.

Recovery Planning

There has not been a Recovery Plan prepared specifically for the range-wide, threatened population of olive ridley turtles. The 1998 Recovery Plan (NMFS and USFWS 1998f) was prepared for olive ridleys found in the U.S. Pacific. Olive ridley turtles found in the Pacific could originate from the Pacific Coast of Mexico or from another nesting population. As such, the recovery goals in the 1998 Recovery Plan for the U.S. Pacific olive ridley turtle can apply to both listed populations. See the 1998 Recovery Plan for the U.S. Pacific olive ridley turtles for complete down listing/delisting criteria for their recovery goals. The following items were the recovery criteria identified to consider delisting:

- All regional stocks that use U.S. waters have been identified to source beaches based on reasonable geographic parameters;
- Foraging populations are statistically significantly increasing at several key foraging grounds within each stock region;
- All females estimated to nest annually at source beaches are either stable or increasing for over ten years;
- Management plan based on maintaining sustained populations for turtles is in effect; and

- International agreements in place to protect shared stocks.

4.2.15 Giant Manta Ray

The giant manta ray (previously *Manta birostris*) was listed as threatened under the ESA on January 22, 2018 (83 Fed. Reg. 2916). On November 22, 2023, the listing was revised to reflect the giant manta ray's updated taxonomy (*Mobula birostris*; 88 Fed. Reg. 81351). The species inhabits tropical, subtropical, and temperate bodies of water (Figure 43) and is commonly found offshore, in oceanic waters, and near productive coastlines (Marshall et al. 2009; Kashiwagi et al. 2011). They also occasionally occur within estuaries (e.g., lagoons and bays) and Intracoastal Waterways. In terms of range, within the Northern hemisphere, the species has been documented as far north as southern California and New Jersey on the United States west and east coasts, respectively, and Mutsu Bay, Aomori, Japan, the Sinai Peninsula and Arabian Sea, Egypt, and the Azores Islands (Gudger 1922; Kashiwagi et al. 2010; Moore 2012; CITES 2013). In the Southern Hemisphere, the species occurs as far south as Peru, Uruguay, South Africa, New Zealand and French Polynesia (Mourier 2012; CITES 2013).

Life History

Giant manta rays make seasonal long-distance migrations, aggregate in certain areas and remain resident, or aggregate seasonally (Dewar et al. 2008; Graham et al. 2012; Girondot et al. 2015; Stewart et al. 2016). The giant manta ray is a seasonal visitor along productive coastlines with regular up-welling, in oceanic island groups, and at offshore pinnacles and seamounts. The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal up-welling, seawater temperature, and possibly mating behavior. They have also been observed in estuarine waters inlets, with use of these waters as potential nursery grounds (Adams and Amesbury 1998; Milessi and Oddone 2003; Medeiros et al. 2015).



Figure 43. Extent of Occurrence (dark blue) and Area of Occupancy (light blue) for Giant Manta Rays based on Species Distribution (Lawson et al. 2017)

Giant manta rays are known to aggregate in various locations around the world in groups usually ranging from 100 to 1,000 (Notarbartolo-di-Sciara and Hillyer 1989; Graham et al. 2012; Venables 2013). These sites function as feeding sites, cleaning stations (areas where smaller fish and crustaceans eat parasites off giant manta rays), or sites where courtship interactions take place (Heinrichs et al. 2011; Graham et al. 2012; Venables 2013). The appearance of giant manta rays in these locations is generally predictable. For example, food availability due to high productivity events tends to play a significant role in feeding site aggregations (Notarbartolo-di-Sciara and Hillyer 1989; Heinrichs et al. 2011). Giant manta rays have also been shown to return to a preferred site of feeding or cleaning over extended periods of time (Dewar et al. 2008; Graham et al. 2012; Medeiros et al. 2015). In addition, giant and reef manta rays in Keauhou and Ho'ona Bays in Hawaii appear to exhibit learned behavior. These manta rays learned to associate artificially lighting with high plankton concertation (primary food source) and shifted foraging strategies to include sites that had artificially lighting at night (Clark 2010). While little is known about giant manta ray aggregation sites, the Flower Garden Banks National Marine Sanctuary and the surrounding region might represent the first documented nursery habitat for giant manta ray (Stewart et al. 2018). Stewart et al. (2018) found that the Flower Garden Banks National Marine Sanctuary provides nursery habitat for juvenile giant manta rays because small age classes have been observed consistently across years at both the population and individual level. The Flower Garden Banks National Marine Sanctuary may be an optimal nursery ground because of its location near the edge of the continental shelf and proximity to abundant pelagic food resources. In addition, small juveniles are frequently observed along a portion of Florida's east coast, indicating that this area may also function as a nursery ground for juvenile giant manta rays. Since directed visual surveys began in 2016, juvenile giant manta rays are regularly observed in the shallow waters (less than 5 m depth) from Jupiter Inlet to Boynton Beach Inlet (J Pate, Florida Manta Project, unpublished data). However, the extent of this purported nursery ground is unknown as the survey area is limited to a relatively narrow geographic area along Florida's southeast coast.

The giant manta ray appears to exhibit a high degree of plasticity in terms of its use of depths within its habitat. Tagging studies have shown that the giant manta rays conduct night descents from 200 to 450 m depths (Rubin et al. 2008; Stewart et al. 2016) and are capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpublished data 2011, cited in Marshall et al. (2011)). Stewart et al. (2016) found diving behavior may be influenced by season, and more specifically, shifts in prey location associated with the thermocline, with tagged giant manta rays (n=4) observed spending a greater proportion of time at the surface from April to June and in deeper waters from August to September. Overall, studies indicate that giant manta rays have a more complex depth profile of their foraging habitat than previously thought, and may actually be supplementing their diet with the observed opportunistic feeding in near-surface waters (Couturier et al. 2013; Burgess et al. 2016).

Giant manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderately sized fishes (Miller and Klimovich 2017). While it was previously assumed, based

on field observations, that giant manta rays feed predominantly during the day on surface zooplankton, results from recent studies (Couturier et al. 2013; Burgess et al. 2016) indicate that these feeding events are not an important source of the dietary intake. When feeding, giant manta rays hold their cephalic lobes in an "O" shape and open their mouth wide, which creates a funnel that pushes water and prey through their mouth and over their gill rakers. They use many different types of feeding strategies, such as barrel rolling (doing somersaults repeatedly) and creating feeding chains with other mantas to maximize prey intake.

The giant manta ray is viviparous (i.e., gives birth to live young). They are slow to mature and have very low fecundity and typically give birth to only one pup every two to three years. Gestation lasts approximately 10 to 14 months. Females are only able to produce between five and 15 pups in a lifetime (CITES 2013; Miller and Klimovich 2017). The giant manta ray has one of the lowest maximum population growth rates of all elasmobranchs (Dulvy et al. 2014; Miller and Klimovich 2017). The giant manta rays generation time (based on *M. alfredi* life history parameters) is estimated to be 25 years (Miller and Klimovich 2017).

Although giant manta rays have been reported to live at least 40 years, not much is known about their growth and development. Maturity is thought to occur between eight to ten years of age (Miller and Klimovich 2017). Males are estimated to mature at around 3.8 m disc width (slightly smaller than females) and females at 4.5 m disc width (Rambahiniarison et al. 2018).

Hearing

Giant manta rays are elasmobranchs, and although there is no known information on their sound production and hearing abilities, these abilities have been studied in other elasmobranchs species. Elasmobranchs, like all fish, have an inner ear capable of detecting sound and a lateral line capable of detecting water motion caused by sound (Hastings and Popper 2005; Popper and Schilt 2009). However, unlike most teleost fish, elasmobranchs do not have swim bladders, and thus are unable to detect sound pressure (Casper et al. 2012). The lack of a swim bladder also means elasmobranchs are not capable of producing many of the sounds produced by teleost fish that have swim bladders. In fact, elasmobranchs likely produce very few sounds, if any, and instead focus on listening to the sounds of their prey (Myrberg 2001).

Data for elasmobranchs fish suggest they can detect sound between 20 Hz to one kHz with the highest sensitivity to sounds at lower ranges (Myrberg 2001; Casper et al. 2003; Casper and Mann 2006; Casper and Mann 2009; Casper et al. 2012; Ladich and Fay 2013). Studies involving oceanic whitetip sharks show attraction to low frequency sounds, particularly those between 25 and 50 Hz, with less but still noticeable attraction at higher frequencies between 500 and 1,000 Hz (Myrberg et al. 1975b; Myrberg et al. 1975a; Myrberg et al. 1976; Myrberg et al. 1978; Myrberg 2001).

Population Dynamics

There are no current or historical estimates of global abundance of giant manta rays, with most estimates of subpopulations based on anecdotal observations. The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 2013) found that only ten

populations of giant manta rays had been actively studied, 25 other aggregations have been anecdotally identified, all other sightings are rare, and the total global population may be small. Subpopulation abundance estimates range between 42 and 1,500 individuals, but are anecdotal and subject to bias (Miller and Klimovich 2017). There is no available information on population trend for the species.

Clark (2010) suggests that giant manta rays may forage in less productive pelagic waters and conduct seasonal migrations following prey abundance. Satellite tracking studies using pop-up satellite archival tags registered movements of the giant manta ray from the Yucatan, Mexico, into the Gulf of America (448 km) (Marshall et al. 2016). Despite this large range, sightings are often sporadic. The timing of these sightings also varies by region (for example, the majority of sightings in Brazil occur during June and September, while in New Zealand sightings mostly occur between January and March) and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seawater temperature, and possibly mating behavior (De Boer et al. 2015; Armstrong et al. 2016). However, a study by Stewart et al. (2016) suggests that the species may not be as highly migratory as previously thought. Using pop-up satellite archival tags in combination with analyses of stable isotope and genetic data, the authors found evidence that giant manta rays may actually exist as well structured subpopulations off Mexico's coast that exhibit a high degree of residency (Stewart et al. 2016). Within its range, the giant manta ray inhabits tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and near productive coastlines (Marshall et al. 2009; Kashiwagi et al. 2011). As such, giant manta rays can be found in cooler water, as low as 19 degrees Celsius, although temperature preference appears to vary by region (Duffy and Abbott 2003; Marshall et al. 2009; Freedman and Roy 2012; Graham et al. 2012).

Status

Although manta rays have been reported to live for at least 40 years (Marshall and Bennett 2010; Kitchen-Wheeler 2013; Marshall et al. 2016) with low rates of natural mortality, the time needed to grow to maturity and the low reproductive rates mean that a female will be able to produce only five to fifteen pups in her lifetime. The best available data indicate that the giant manta ray has suffered population declines (up to 95%, in some areas), based on landings and market data, diver observations, and anecdotal sightings. The declines are largely believed to be caused by overfishing (Miller and Klimovich 2017). In areas where the species is not subject to fishing, populations may be stable. For example, Rohner et al. (2013) reported that giant manta ray sightings remained constant off the coast of Mozambique over a period of eight years. Given the migratory nature of this species, population declines in waters where the manta rays are protected have also been observed but attributed to overfishing of the species in adjacent areas within its large home range. With populations potentially ranging from around 100 to 1,500 individuals (see Table 4 in Miller and Klimovich (2017)), their life history traits and productivity estimates, particularly their low reproductive output and sensitivity to changes in adult survival rates, giant manta ray populations are inherently vulnerable to depletions, with low likelihood of recovery.

Critical Habitat

NMFS determined that the designation of critical habitat for giant manta ray was not determinable because data sufficient to perform the required analyses were lacking (84 FR 66652. 2019).

Recovery Planning

On December 19, 2019, NMFS published a recovery outline, which serves as an interim guidance to direct recovery efforts for giant manta ray (NMFS 2019f). The initial focus of the interim recovery program is two-fold: 1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and 2) to gather additional information through research and monitoring on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries (including at-vessel and post-release mortality), and other potential threats that may contribute to the species' decline.

Because the major threat currently contributing to the species' decline is overutilization in waters outside of U.S. jurisdiction, international coordination will be critical to ensuring recovery of the species. Therefore, to be effective, these actions would need to be undertaken throughout the species' range, both domestically and internationally.

4.2.16 Oceanic Whitetip Shark

On January 30, 2018, NMFS listed the oceanic whitetip shark (Carcharhinus longimanus) as a threatened species throughout its range (83 Fed. Reg. 4153). On May 14, 2024 we proposed a 4(d) Rule for Protective Regulations for the oceanic whitetip shark (89 Fed. Reg. 41917). The oceanic whitetip shark is globally distributed in epipelagic tropical and subtropical waters (Figure 44) between 30º North and 35º South latitudes (Rigby et al. 2019; Young and Carlson 2020).

Life History

The oceanic whitetip shark gives birth to live young (i.e., "viviparous"). Their reproductive cycle is thought to be biennial, giving birth on alternate years, after a lengthy 10 to 12-month gestation period. The number of pups in a litter ranges from one to 14 (mean = 6), and a positive correlation between female size and number of pups per litter has been observed, with larger sharks producing more offspring (Seki et al. 1998; Bonfil et al. 2008; IOTC 2014). Not a great deal is known about oceanic whitetip sharks' lifespan. Estimates range from 12 to 13 years (Seki et al. 1998; Lessa et al. 1999), to 17 years, and even up to 20 years old (Young et al. 2017). They are a slow-growing species, and growth rates are believed to be similar between the sexes (Seki et al. 1998; Lessa et al. 1999; Joung et al. 2016; Young et al. 2017). Adults can grow up to 3.4 m long and can weigh up to 230 kg. Age at maturity varies by ocean region, with six to seven years old recorded in the southwest Atlantic, and four to nine years old in the North Pacific, with the sexes having similar ages at maturity (Seki et al. 1998; Lessa et al. 1999; Joung et al. 2016).



Figure 44. Geographic Range of the Oceanic Whitetip Shark (Young and Carlson 2020)

Little is known about the movement or possible migration paths of the oceanic whitetip shark. Oceanic whitetip sharks are highly migratory, and satellite telemetry data of seven sharks tagged in French Polynesia revealed movement across net distances of 1,625 to 7,885 km (Burkhardt et al. 2025). Although the species is highly migratory and capable of making long distance movements, tagging data provides evidence that this species also exhibits a high degree of philopatry (i.e., site fidelity) in some locations (Shen et al. 2024). Juveniles seem to be concentrated in equatorial latitudes, while specimens in other maturational stages are more widespread (Tambourgi et al. 2013). Andrzejaczek et al. (2018) modeled the effects of sea surface temperature on oceanic whitetip shark movements and found they generally oscillated throughout the upper 200 m (656 ft) of the water column. However, in summer they reduced the amount of time spent in the upper 50 m (164 ft), and in winter oscillations decreased in amplitude and cycle length and sharks frequently occupied the upper 50 m.

Oceanic whitetip sharks are regarded as opportunistic feeders, eating teleosts (bony fish) and cephalopods. Large pelagic fish species commonly found in the stomachs of oceanic whitetips include, blackfin tuna, white marlin, and barracuda.

Hearing

See Section 4.2.15 (giant manta ray status of species) for a general discussion of hearing in elasmobranchs.

Population Dynamics

There is no range-wide abundance estimate available for oceanic whitetip sharks. However, the species was once one of the most abundant sharks in the ocean. Numerous lines of evidence, including the results of a recent stock assessment and several other abundance indices (e.g.,

trends in occurrence and composition in fisheries catch data, standardized catch-per-unit-effort data, and various biological indicators) indicate that the oceanic whitetip shark has experienced significant population declines throughout a majority of its global range (Young and Carlson 2020). Rigby et al. (2019a) used a Bayesian state-space tool for trend analysis of abundance indices (Just Another Red List Assessment or JARA) to determine a global abundance trend for the oceanic whitetip shark. The estimated area-weighted global population trend was a decline of 98-100%, with the highest probability of 80-99% reduction over three generation lengths (61.2 years based on IUCN criteria). However, it should be noted that there was no abundance data that spanned over three generations and the decline was based on the projected trend from the current observed data (NMFS 2023g). Evaluating bycatch data available since the 1990s, the species has declined by at least 75% throughout its range (Pacoureau et al. 2021) and at similarly high levels in all regions (NMFS 2024c).

Populations in the Eastern Pacific Ocean are thought to have declined between 80 and 90% since the late 1990s (Hall 2013). Tremblay-Boyer et al. (2019) analyzed trends in the Western and Central Pacific Ocean oceanic whitetip population across multiple models covering a wide range of uncertainty scenarios, including stock productivity and discard mortality. They concluded that current spawning stock biomass is predicted to be below 5% of the unfished spawning biomass and the population could go extinct in the long-term under current levels of fishing mortality. A 2019 stock assessment for oceanic whitetip shark showed that while the stock was still overfished and undergoing overfishing, there was also a small reduction in stock depletion, with increases in recruitment and a reduction in fishing mortality relative to reference points under certain catch scenarios (Tremblay-Boyer et al. 2019; Rice et al. 2021).

Rice et al. (2021) used the Stock Synthesis forecast module to make future projections for oceanic whitetip population biomass levels over a range of future catch scenarios. Based on this study, the population is projected to increase at a moderate pace over the projection period (through 2031) under the zero-catch scenario, as well as for many of the models under the projected status quo, 10% and 20% decrease catch scenarios. The average annual change from 2018-2028 also showed that the majority of the population trajectories are increasing (Rice et al. 2021).

NMFS (2025b) used data from Bigelow et al. (2022) to calculate a mean growth rate of 7.66% per year increase as a reasonable estimate of the current trend for oceanic whitetip sharks. Based on an estimate of 775,214 WCPO oceanic whitetip sharks in 2016, at a 7.66% rate of population growth based on the exponential growth function (Nt = Nt-1*er), NMFS (2025b) estimated a 2024 population abundance around 2.3 million oceanic whitetip sharks in the Pacific Ocean.

While there is limited research on the genetic diversity of oceanic whitetip sharks, that which exists indicates low genetic diversity. Compared to other pelagic sharks (e.g., silky sharks (*Carcharhinus falciformis*), oceanic whitetip sharks display relatively low mitochondrial DNA genetic diversity (Clarke et al. 2015; Camargo et al. 2016; Ruck 2016). As noted previously, the species appears to display a high degree of philopatry to certain sites, with females giving birth

on one side of a basin or the other, indicating little if any mixing with individuals of other regions (Howey-Jordan et al. 2013; Tolotti et al. 2015; Young et al. 2017). Thermal barriers (i.e., water temperatures less than 15° C) may prevent inter-ocean basin movements. Nuclear and mitochondrial DNA demonstrate genetic differentiation between samples collected in the Atlantic and Indo-Pacific Oceans (Ruck 2016; Ruck et al. 2024). There may also be additional substructure (e.g., mitochondrial differentiation between the eastern and western Atlantic Ocean (Ruck et al. 2024).

Oceanic whitetip sharks are distributed throughout open ocean waters, the outer continental shelf, and around oceanic islands, primarily from 10° North to 10° South, but up to 30° North and 35° South (Young et al. 2017). They can be found at the ocean surface and down to at least 152 m deep, but most frequently stay between depths of 25.5 and 50 m (Carlson and Gulak 2012; Young et al. 2017). They display a preference for water temperatures above 20° Celsius, but can be found in waters between 15° and 28° Celsius, and can briefly tolerate waters as cold as 7.75° Celsius during dives to the mesopelagic zone (Howey-Jordan et al. 2013; Howey et al. 2016).

Status

In addition to declines in oceanic whitetip catches throughout its range, there is also evidence of declining average size over time in some areas, and is a concern for the species' status given evidence that litter size is potentially correlated with maternal length. Such extensive declines in the species' global abundance and the ongoing threat of overutilization, the species' slow growth and relatively low productivity, makes them generally vulnerable to depletion and potentially slow to recover from overexploitation. The biology of the oceanic whitetip shark indicates that it is likely to be a species with low resilience to fishing and minimal capacity for compensation (Rice and Harley 2012).

The Recovery Status Review and IUCN Red List review (Rigby et al. 2019a) concluded that the species continues to decline, but there are some indications that these trends have slowed or even reversed. Numerous regulatory mechanisms exist to deter retention, modify gear, and ensure proper handling; and recent data suggest declining mortality rates in pelagic fisheries, especially in the Atlantic and Pacific Oceans (Worm et al. 2024). However, the most recent NMFS 5-year status review (NMFS 2025a) reaffirmed that the oceanic whitetip shark is likely to become an endangered species within the foreseeable future throughout its range because of habitat modification, overutilization, inadequate regulatory mechanisms, and manmade factors.

Critical Habitat

No critical habitat has been designated for the oceanic whitetip shark.

Recovery Goals

The three recovery objectives for the oceanic whitetip shark (NMFS 2024c) are to:

- Ensure the oceanic whitetip shark maintains resiliency and geographic representation, and is a functional component of the ecosystem, by increasing overall abundance to achieve viable populations in all ocean basins;

- Increase oceanic whitetip shark resiliency by managing or eliminating significant anthropogenic threats; and
- Ensure the continued viability of the oceanic whitetip shark through development and effective implementation of regulatory mechanisms for the long-term protection of the species.

In the absence of historical abundance data, NMFS developed four alternative demographic recovery criteria that can be used to determine when a stable and sustainable population size has been reached. Six threats-based recovery criteria that indicate the threats to the species are reduced, managed, or eliminated such that they are not contributing to the species being in danger of extinction within the foreseeable future were also specified in the plan (NMFS 2023d). For details on recovery and delisting criteria see the draft recovery plan (https://www.fisheries.noaa.gov/resource/document/draft-recovery-plan-oceanic-whitetip-shark).

4.2.17 Scalloped Hammerhead Shark – Eastern Pacific DPS

On July 3, 2014, NMFS listed four scalloped hammerhead shark DPSs: Eastern Pacific DPS and Eastern Atlantic DPS (entirely foreign) were listed as endangered; Central and Southwest Atlantic DPS and Indo-West Pacific DPS were listed as threatened (79 Fed. Reg. 38213). Only the Eastern Pacific DPS overlaps with the action area (Figure 45). Scalloped hammerheads are moderately large, coastal pelagic sharks found worldwide in coastal warm temperate and tropical seas in the Atlantic, Pacific and Indian Oceans between 46°N and 36°S (Miller et al. 2014).

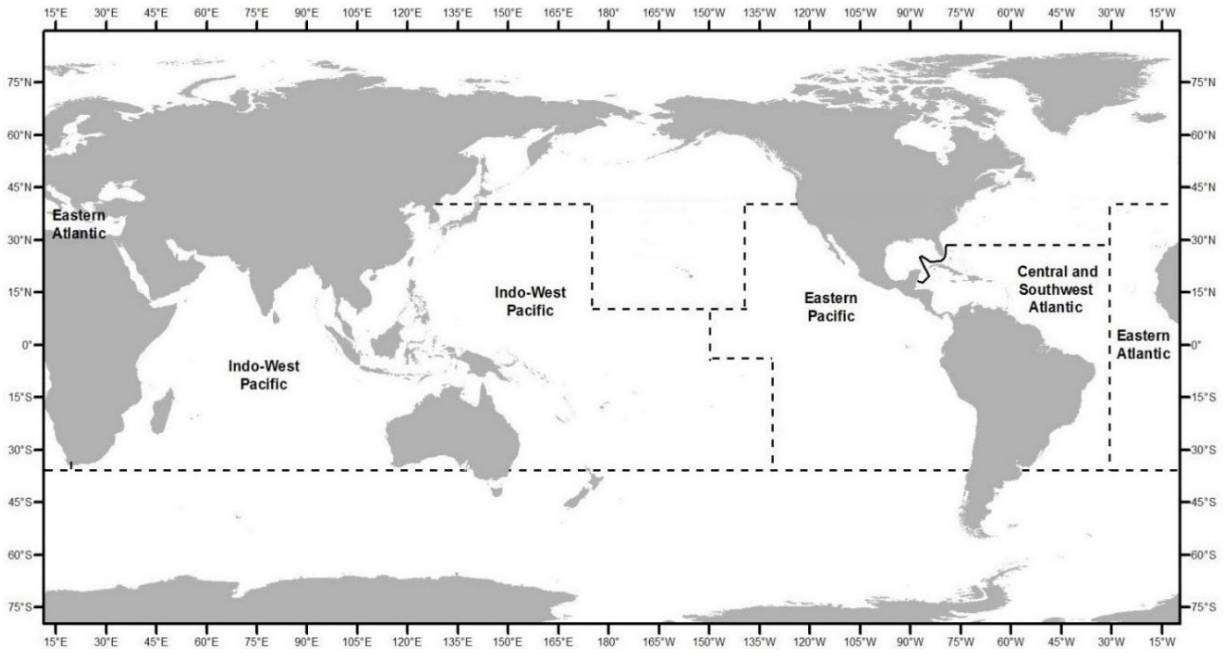


Figure 45. Scalloped Hammerhead Shark DPS Boundaries (Source: 78 Fed. Reg. 20717; April 5, 2013). Note: the Northwest Atlantic/Gulf of Mexico and Central Pacific DPSs are not listed under the ESA.

Life History

Rigby et al. (2019b) summarized the literature on age and growth from various scalloped hammerhead populations. Scalloped hammerheads reach a maximum size of 370–420 cm total length (TL). Males mature at smaller sizes (i.e., between 140–198 cm TL) compared to females (i.e., between 200–250 cm TL). On average, females mature at 13.2 years and live up to 35 years. They give birth annually with a litter size from 12-41 pups and generation length is 24.1 years (Rigby et al. 2019b). The age at maturity differs by region. Parturition, however, does not appear to vary by region and may be partially seasonal (Harry et al. 2011), with neonates present year round but with abundance peaking during the spring and summer months (Duncan and Holland 2006; Adams and Paperno 2007; Harry et al. 2011; Noriega et al. 2011). Females move inshore to birth, with litter sizes anywhere between one and 41 live pups. The scalloped hammerhead shark gives birth to live young (i.e., "viviparous"), with a gestation period of 9 to 12 months, which may be followed by a one-year resting period (Branstetter 1987; Stevens and Lyle 1989; NMFS 2020e) which may be followed by a one-year resting period (Liu and Chen 1999). Size at birth is estimated between one and two feet (0.3 to 0.6 m) TL.

Scalloped hammerheads are found over continental shelves and the shelves surrounding islands, as well as adjacent deep waters, but are seldom found in waters cooler than 22 degrees Celsius (Compagno 1984; Schulze-Haugen and Kohler 2003). They range from the intertidal and surface to depths of up to 450-512 m (Klimley 1993), with occasional dives to even deeper waters (Jorgensen et al. 2009). Scalloped hammerheads have also been documented entering enclosed bays and estuaries (Compagno 1984). Neonates and juveniles inhabit nearshore nursery habitats for up to one year or more as these areas provide valuable refuge from predation (Duncan and Holland 2006). Scalloped hammerheads are high trophic level, opportunistic predators whose diet includes crustaceans, fish and cephalopods. Estupiñán-Montaño et al. (2021) found that scalloped hammerheads in the Eastern Tropical Pacific show changes in trophic position versus foraging areas (inshore and offshore areas) according to sex, growth, and maturity stages, demonstrating a broad consumption of prey and a wide habitat use.

Hearing

See Section 4.2.15 (giant manta ray status of species) for a general discussion of hearing in elasmobranchs.

Population Dynamics

Systematic monitoring of population abundance does not exist for any of the scalloped hammerhead DPSs. However, scalloped hammerhead sharks have a life history that is susceptible to overharvesting, and data from shark deterrent programs, as well as diver and fishermen surveys, indicate all four DPSs have declined (NMFS 2020e). Historical estimates of effective population size (or the number of breeding individuals in the population) in the eastern Pacific range from 34,995 to 43,551 individuals (Nance et al. 2011). Using 15 microsatellite loci and mitochondrial DNA (mtDNA) from eastern Pacific tissue samples, Nance et al. (2011)

discovered that the current effective population size is significantly smaller (one to three orders of magnitude) than the historical effective population size, indicating that scalloped hammerheads in the eastern Pacific experienced a bottleneck and suffered significant declines.

While current abundance data for this DPS are sparse, local and regional population declines have been indicated from recent fishery dependent data (Arauz et al. 2004; White et al. 2015) (Peñaherrera-Palma et al. 2018). Evidence of heavy fishing pressure by industrial/commercial and artisanal fisheries, high at-vessel fishing mortality, limited regulatory mechanisms and poor enforcement indicate that the Eastern Pacific DPS is currently at or near a level of abundance and productivity that places its current and future persistence in question (Miller et al. 2014).

Based on information related to genetic variation among populations, behavior and physical factors, and differences in international regulatory mechanisms, the scalloped hammerhead Extinction Risk Analysis team identified six DPSs: Northwest Atlantic and Gulf of Mexico; Central and Southwest Atlantic; Eastern Atlantic; Indo-West Pacific; Central Pacific; and Eastern Pacific (Miller et al. 2014). Differentiation among populations is corroborated by more recent genetic data. Green et al. (2022) defined the biological stock structure of scalloped hammerheads in the Pacific into four regions: the Western Indian (Seychelles), the central Indo–Pacific (Papua New Guinea, Indonesia, Australia, Philippines, Taiwan and Fiji), Central Pacific (Hawaii) and the tropical East Pacific (Gulf of California). Mitochondrial DNA sequences from Harned et al. (2022) added to previously published data clustered scalloped hammerheads into three phylogenetic lineages: Atlantic, western Indian, and eastern Indian-Pacific regions.

Scalloped hammerhead sharks are highly mobile and partly migratory and are likely the most abundant of the hammerhead species (Maguire 2006); however the risk of local depletions is of concern. Low level of gene flow between locations suggests that scalloped hammerheads do not cross ocean basins (Green et al. 2022). Research indicates that individuals do not typically move greater than 200 km. Scalloped hammerhead sharks primarily occur over continental and insular shelves, ranging from surface waters to depths of 512 m with occasional dives to deeper water up to 1,000 m.

Satellite tracks from 23 pelagic shark species (n = 1,804), including scalloped hammerheads (n = 31) tagged from 2002-2017 in the Atlantic, Indian, and Pacific Ocean indicate the pelagic shark species occupy predictable habitat hotspots with a preference for fronts characterized by strong sea surface temperature and high productivity gradients (Queiroz et al. 2016; Queiroz et al. 2019). A study of the movements of scalloped hammerhead sharks at Cocos Island, Costa Rica and between oceanic islands in the eastern tropical Pacific demonstrates the importance of oceanic islands for this species, and suggests that such islands may form a network of hotspots in the Eastern Tropical Pacific (Nalesso et al. 2019). Zhang (2022) modeled seasonal distribution of scalloped hammerhead sharks based 11 environmental variables to predict their suitable habitats. Sea surface temperature was the most important predictor, followed by depth, salinity, primary productivity, distance to macrohabitats, and distance to shoreline. Seasonal suitable habitats were mainly found in tropical continental shelves and deep seas (Zhang 2022). Confirmed or potential

parturition and nursery areas include Rewa Delta, Fiji; Great Barrier Reef, Australia; Malpelo Island; the Colombian Pacific coast; Golfo Dulce; Wafer Bay, Isla del Coco, Costa Rica; El Tecuán Estuary off the Pacific coast of Jalisco, Mexico; and the Gulf of California (NMFS 2020e).

Status

Overutilization for commercial purposes remains a significant threat contributing to the extinction risk of the Eastern Pacific scalloped hammerhead shark. Scalloped hammerhead sharks have a life history (i.e., slow growth and relatively low productivity) that is susceptible to overharvesting. While new marine protected areas and shark sanctuaries have been established since the species was listed, protection from threats is not guaranteed as some of these areas do not include important foraging areas, migratory corridors, and nursery areas for the Eastern Pacific scalloped hammerhead shark (NMFS 2020e).

Critical Habitat

No critical habitat has been designated for the scalloped hammerhead shark.

Recovery Planning

NMFS has not prepared a recovery plan for the scalloped hammerhead shark.

4.2.18 Steelhead

Five DPSs of ESA-listed steelhead occur within the HCTT action area. We begin this section with a general discussion of steelhead life history traits, followed by a discussion of the population dynamics and status of each particular DPS.

General Life History Traits

Steelhead typically migrate to marine waters after spending two to three 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 (Busby et al. 1996). 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).

When ready to spawn female steelhead trout select 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.

4.2.19 Steelhead – Southern California DPS

On August 18, 1997, NMFS listed the Southern California DPS of steelhead (Figure 46) as endangered and reaffirmed the DPS's status as endangered on January 5, 2006. Critical habitat was designated for this species on September 2, 2005. This DPS includes naturally spawned anadromous *O. mykiss* (steelhead) originating below natural and manmade impassable barriers from the Santa Maria River to the U.S.-Mexico Border.

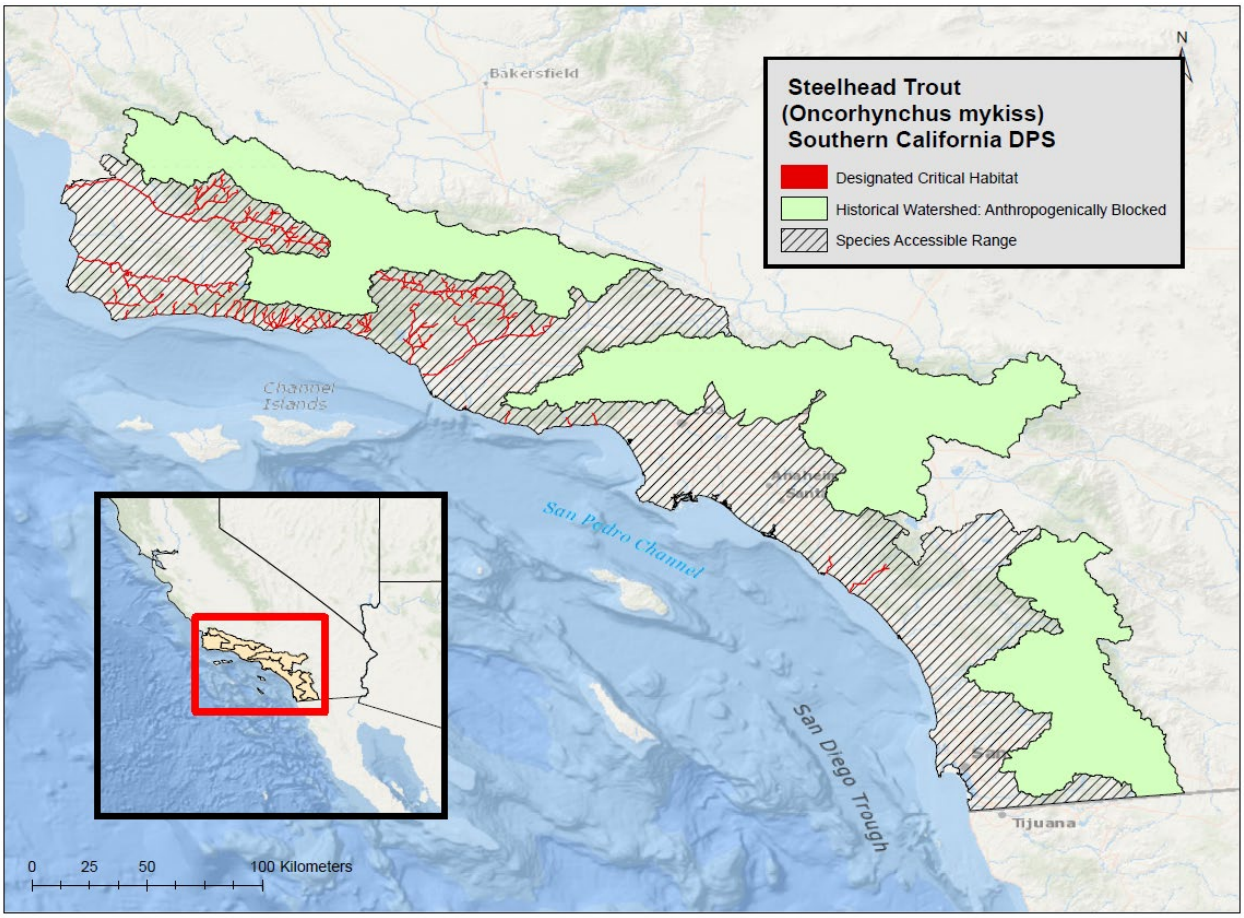


Figure 46. Steelhead Southern California DPS Inland Range and Designated Critical Habitat

Population Dynamics

The Southern California steelhead DPS is composed of mixed populations of native *O. mykiss*: anadromous fish (steelhead) and freshwater resident fish (rainbow trout). The ratio of anadromous and non-anadromous fish in a population or watershed can vary considerably, geographically and temporally (NMFS 2023b). Migration and life history patterns of Southern California steelhead populations are dependent on rainfall and streamflow (Moore 1980). Steelhead within this DPS can withstand higher temperatures compared to populations to the north. Before the construction of dams, location extirpation and reoccurrence of steelhead runs likely occurred naturally; however, the Southern California Steelhead DPS has no or very few drought refugia that currently are regularly connected to the ocean (NMFS 2023b).

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. Dagit et al. (2020) summarized observations of adult steelhead in the Southern California Steelhead DPS over the past several decades and found that their occurrence is consistently extremely rare. Only 177 adult steelhead were observed over the last 25 years, an average of seven per year for the entire geographic area inhabited by this DPS. Density of freshwater juvenile steelhead from 2000-2020 did not meet or exceed 0.30 fish/m^2 (a value provisionally identified as potentially appropriate new population density viability criterion) for any population within the DPS (NMFS 2023b). The recent drought and associate wildfires have been very hard on the Southern California Steelhead DPS: adult steelhead run counts have been in the single digits, with many streams having no observed adults over the past five to seven years (NMFS 2023b). There is no evidence that any population in the DPS has produced a run of adult steelhead greater than about 26 fish in the past 25 years. Except for one (Arroyo Sequit), all populations within the DPS for which data were recorded exhibited a negative trend (Boughton 2022), underscoring the extreme vulnerability of this DPS to a loss of genetic and phenotypic diversity and possible extinction (NMFS 2023b). Based on the most recent status review, it appears that periods with anadromous runs in the hundreds or thousands of fish can be ruled out under the existing conditions (NMFS 2023b).

New information on steelhead genetics and the monitoring methodology relevant to viability criteria suggests that the risk of permanently losing the anadromous phenotype over the long term may be very high, and is likely increasing due to the lack of unobstructed migration corridors between upstream drought refugia and the Pacific Ocean (Donohoe et al. 2021; NMFS 2023b). Recent research shows that the tendency to out-migrate to the ocean (versus maturing in freshwater) is associated with particular juvenile body sizes, gender, the presence of a particular inverted gene on chromosome Omy5, and interactions of these factors (Boughton 2022). The recent findings of the genetic architecture underlying anadromy provide a mechanism by which such rarity of the anadromous life history strategy can be sustained over time, but also suggest that ongoing adaptation of the anadromous phenotype is inhibited by its very low expression (NMFS 2023b). Consistently low abundances of adult steelhead in the Southern California Steelhead DPS over many decades indicate that ongoing evolutionary adaptation by AA genotype fish associated with anadromy is unlikely, unless freshwater conditions (including ingress and egress for adult and juvenile steelhead) improve or are restored. This implies a high and increasing potential risk of permanent loss of the anadromous life-history phenotype (NMFS 2023b).

Status

Southern California DPS steelhead face a number of threats, including: (1) fish passage impediments in the mainstems of the major rivers and their tributaries; (2) the alteration of the natural flow regime as a result of dams, diversions, and groundwater extraction; (3) the degradation of habitat as a result of the wide-spread presence of exotic aquatic species; (4)

prolonged drought and the related frequency and magnitude of wildfires in response to changing environmental conditions; and (5) the intensification of land uses at the urban/wildlands. (6) commercial and recreational fishing; and (7) changing environmental conditions overall (NMFS 2023b). 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 drought, and the lack of comprehensive monitoring, has also limited the ability to fully assess the status of individual populations and the DPS as a whole. The systemic anthropogenic threats identified at the time of the initial listing have remained essentially unchanged, though there has been significant progress in removing fish passage barriers in a number of the smaller and mid-sized watersheds. Threats to the Southern California Steelhead DPS posed by environmental variability resulting from projected changes in environmental conditions are likely to exacerbate the factors affecting the continued existence of the DPS.

Critical Habitat

On September 2, 2005, NMFS established critical habitat for the Southern California DPS steelhead in rivers and estuaries (70 Fed. Reg. 52487). This critical habitat is not present within the action area.

Recovery Planning

See the 2012 recovery plan for the southern California steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (NMFS 2012b). A viable population is defined as a population having a negligible risk (less than 5%) of extinction due to threats from demographic variation, natural environmental variation, and genetic diversity changes over a 100-year time frame. A viable DPS is comprised of a sufficient number of viable populations spatially dispersed, but proximate enough to maintain long-term (1,000-year) persistence and evolutionary potential (Mcelhany et al. 2000). The viability criteria are intended to describe characteristics of the species, within its natural environment, necessary for both individual populations and the DPS as a whole to be viable, i.e., persist over a specific period, regardless of other ongoing effects caused by human actions.

Recovery of the endangered Southern California Steelhead DPS will require recovery of a minimum number of viable populations within each of five population groups within their range. Recovery of these individual populations is necessary to conserve the natural diversity (genetic, phenotypic, and behavioral), spatial distribution, and abundance of the species, and thus the long-term viability of the DPS. Each population must exhibit a set of biological characteristics (e.g., minimum mean annual run size, persistence over variable oceanic conditions, spawner density, anadromous fraction, etc.) in order to be considered viable.

4.2.20 Steelhead – California Central Valley DPS

On March 19, 1998 NMFS listed the California Central Valley DPS of steelhead as threatened (63Fed. Reg. 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71Fed.

Reg. 834).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 47). The Central Valley DPS of steelhead also includes steelhead from two artificial propagation programs.

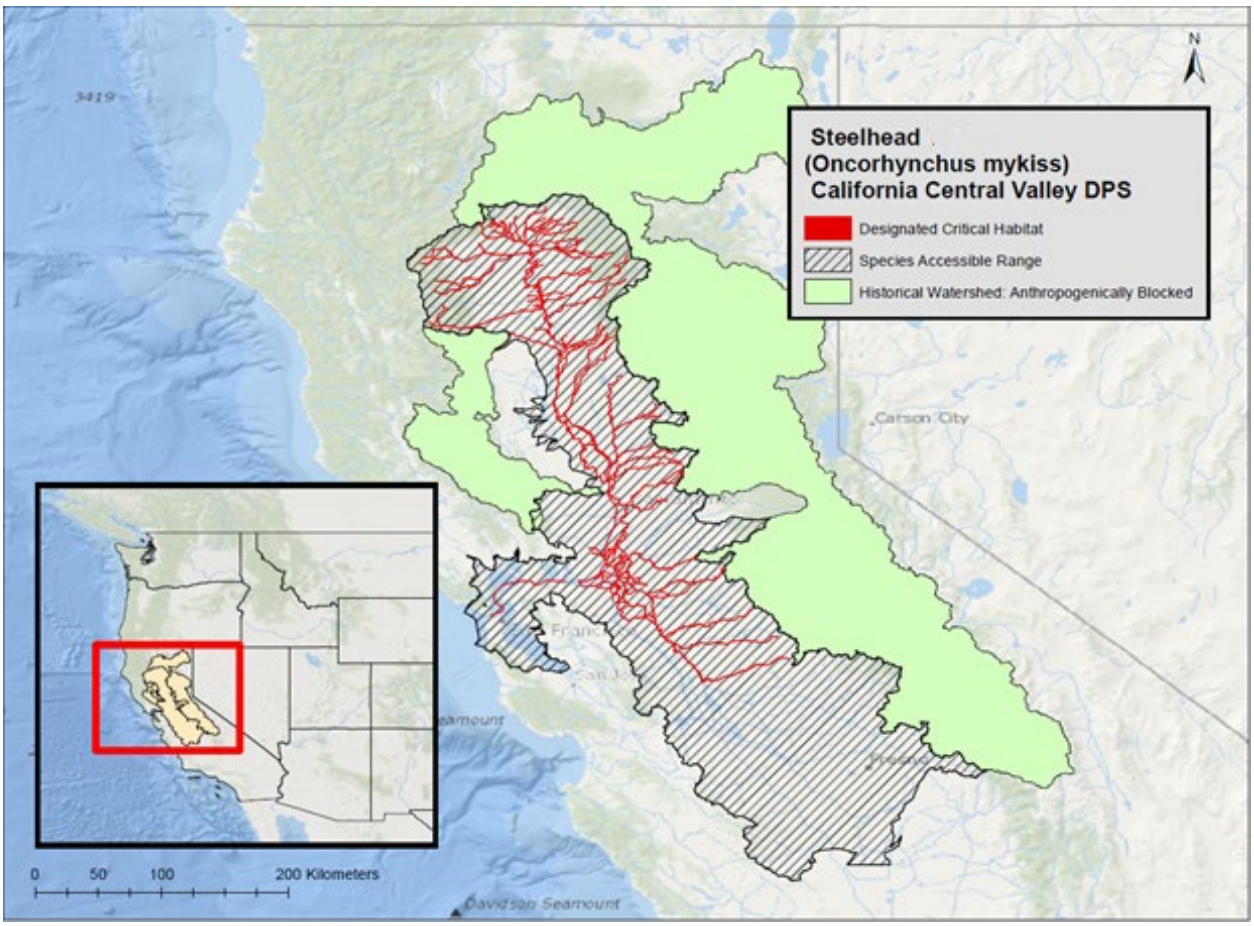


Figure 47. Geographic Inland Range and Designated Critical Habitat of California Central Valley Steelhead

Population Dynamics

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 60 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. California Central Valley steelhead lack long-term 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 2009). A more recent abundance estimate for the California Central Valley ESU of steelhead is between one to two thousand naturally spawned adult fish (NMFS 2016b).

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. 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).

Status

Many watersheds in the Central Valley are experiencing decreased abundance of California Central Valley steelhead. The majority (11 of 16) of populations for which there are data are at a high risk of extinction based on abundance and/or hatchery influence, with no population considered to be at a low risk of extinction (SWFSC 2023). Although steelhead viability has slightly improved for some populations, this modest improvement was driven by the increase in adult returns to hatcheries from their recent lows, while the state of natural-origin fish remains poor and largely unknown (SWFSC 2023). 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, 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 remain concerns for the DPS as a whole. Based upon the limited information available, SWFSC (2023) concluded that the overall viability of the Central Valley steelhead DPS was unchanged since the 2015 assessment and this population remains at a moderate risk of extinction.

Critical Habitat

NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70Fed. Reg. 52488). Designated critical habitat for the California Central Valley steelhead does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

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 2014b). 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

4.2.21 Steelhead – Central California Coast DPS

On August 18, 1997 NMFS listed the Central California Coast DPS of steelhead as threatened (62Fed. Reg. 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71Fed. Reg. 834). 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 48).

Population Dynamics

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). Information on the current abundance of adult steelhead in the CCC steelhead DPS remains relatively scarce. Population-level estimates of adult abundance are entirely lacking for the 25 independent populations in the North Coastal, Interior, Coastal San Francisco Bay, and Interior San Francisco Bay diversity strata identified as essential or supporting in the Federal recovery plan (SWFSC 2023). The interior Russian River winter-run steelhead has the largest spawning runs that may sustainable over the long-term, although hatchery management has eroded the population's genetic diversity (Bjorkstedt et al. 2005; NMFS 2008). A recently initiated program in the Russian River basin provides aggregate estimates of abundance for multiple independent and dependent populations within the basin. In recent years, basin-wide estimates of abundance of steelhead (combined natural and hatchery-origin) have ranged from about 800–2,000 adult fish (SWFSC 2023). However, since abundance estimates are not produced for individual populations within the basin, direct comparison with recovery targets is not yet possible. The Russian River monitoring program has provided quantitative evidence that hatchery origin steelhead constitute roughly 50% of all fish on natural spawning grounds and that these hatchery fish are being observed throughout the basin (SWFSC 2023).

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 2008). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good et al. 2005). Data from Scott Creek life cycle monitoring program, which provides the only consistent, long-term time series in the entire DPS,

indicates no significant trend in abundance, although without the strong 2018–2019 return, it is likely the trend would have been negative and significant (SWFSC 2023).

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 (Shapovalov and Taft 1954; Hayes et al. 2004). Most spawning takes place from January through April. 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).

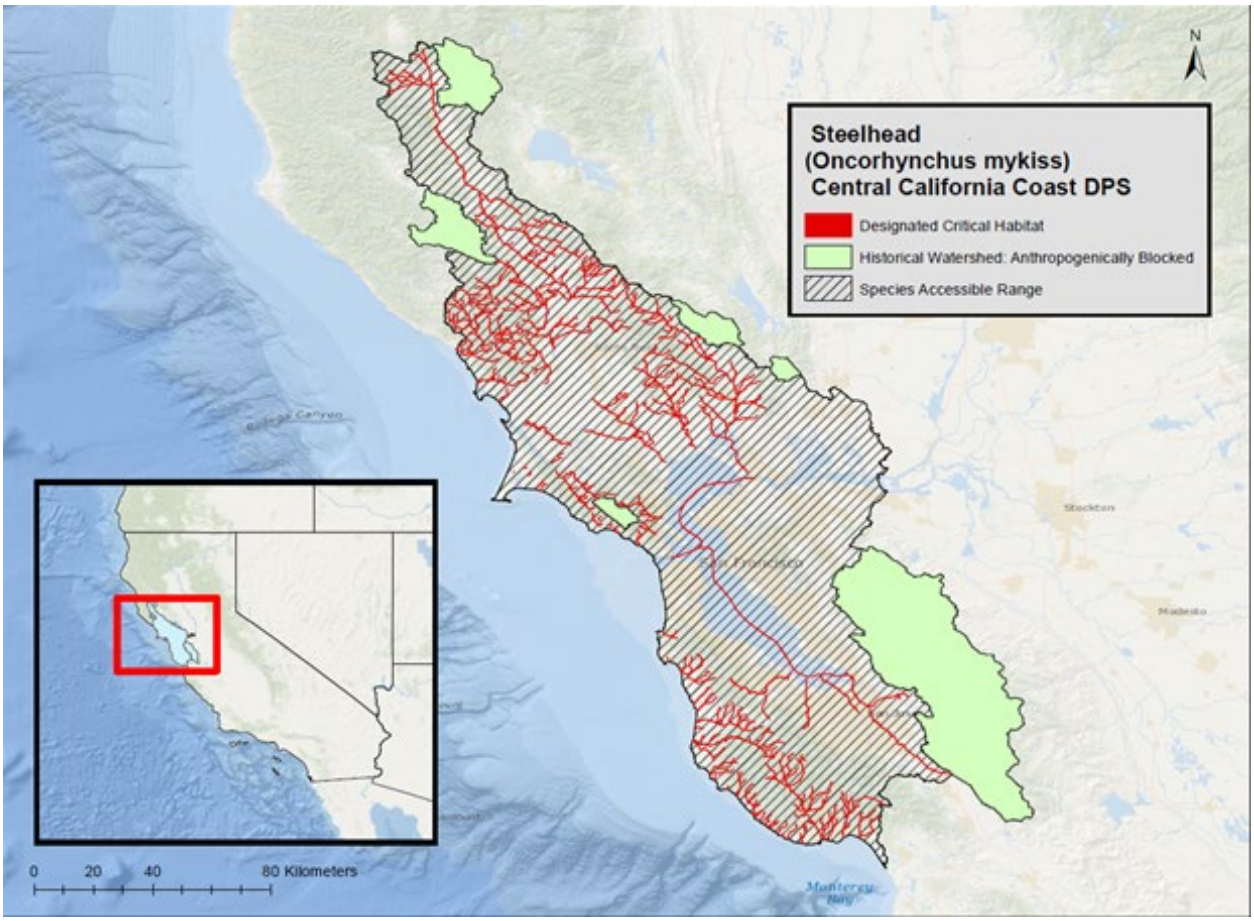


Figure 48. Geographic Inland Range and Designated Critical Habitat of Central California Coast Steelhead

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% 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 2008). While data availability for this DPS remains generally poor, based on new information used for the latest viability assessment, the overall extinction risk for the Central California Coast steelhead is moderate.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70Fed. Reg. 52630). Designated critical habitat for the Central California Coast steelhead does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

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.

4.2.22 Steelhead – Northern California DPS

On June 7, 2000 NMFS listed the Northern California DPS of steelhead as threatened (65Fed. Reg. 36074) and reaffirmed the DPS's status as threatened on January 5, 2006 (71Fed. Reg. 834). 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 49).

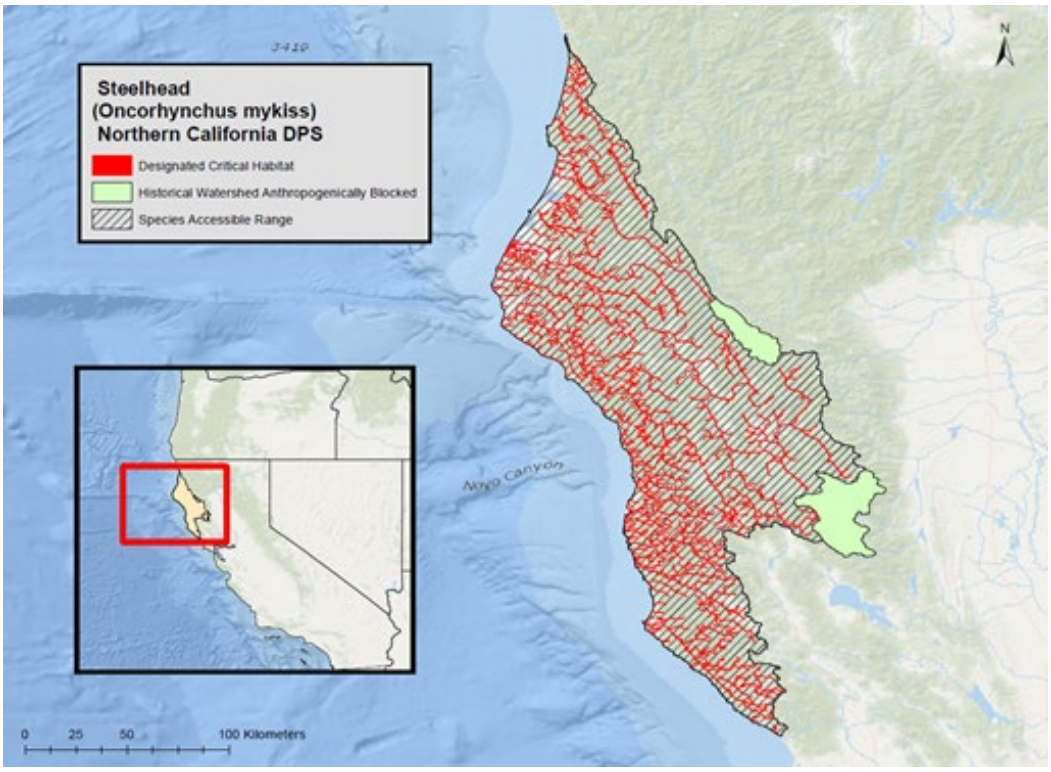


Figure 49. Geographic Inland Range and Designated Critical Habitat of Northern California DPS Steelhead

Population Dynamics

Data availability continues to be patchy across this DPS, with population-level estimates of abundance available for more than half the independent populations in the North Coastal Stratum, the majority of independent populations in the North-Central Coastal and Central Coastal strata, but lacking entirely in the Lower Interior and North Mountain Interior strata (SWFSC 2023). Estimated total abundance of adult Northern California DPS steelhead is around 7,000 fish (NMFS 2019b).

The Northern California DPS of steelhead includes both winter- and summer –run steelhead. 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). Juvenile Northern California DPS 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). Ocean distribution is not well known but coded wire tag recoveries indicate that most Northern California DPS steelhead migrate north and south along the continental shelf (Barnhart 1986).

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 3% to 15% of these goals (SWFSC 2023). An exception may be the Mad River population, where recent population abundances have averaged about 73% of the combined recovery targets for the lower and upper Mad River subpopulations (SWFSC 2023). The high fraction of returning fish that are of hatchery origin (currently

estimated to be 44%) is a concern. Trends for essential independent populations have been mixed, with most either stable or showing slight (non-significant) increases in abundance in recent years. Stratum-wide estimates for winter-run steelhead have trended slightly upwards over the last 10 years (SWFSC 2023).

Summer-run steelhead Northern California DPS steelhead populations remain a significant concern because of how few populations currently exist. The abundance of the Middle Fork Eel River population has remained stable for nearly five decades and is closer to its recovery target (~80%) than any other population in the DPS (SWFSC 2023). Populations in the Mad and Van Duzen rivers have averaged in the low hundreds of fish and are at 18% and 26% of their recovery targets, respectively (SWFSC 2023). With recent climatic trends indicating warming of air temperatures and high variability in stream discharge, summer-run steelhead populations are likely to experience increased risk in the near future due to both decreased thermal suitability of over-summering habitats for adults and reduced access to historical spawning areas in years of low precipitation (SWFSC 2023).

Status

Although data are limited for this steelhead DPS, the available data indicate that most populations remain well below viability targets (SWFSC 2023). Based on new information used for the latest viability assessment, the overall extinction risk for the Northern California steelhead DPS is moderate (SWFSC 2023).

Critical Habitat

NMFS designated critical habitat for Northern California DPS steelhead on September 2, 2005. Designated critical habitat for the Northern California DPS steelhead does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

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 2016f).

4.2.23 Steelhead – South-Central California Coast DPS

On August 18, 1997 NMFS listed the South-Central California Coast DPS of steelhead as threatened (62Fed. Reg. 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71Fed. Reg. 5248). 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 (Figure 50). 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.

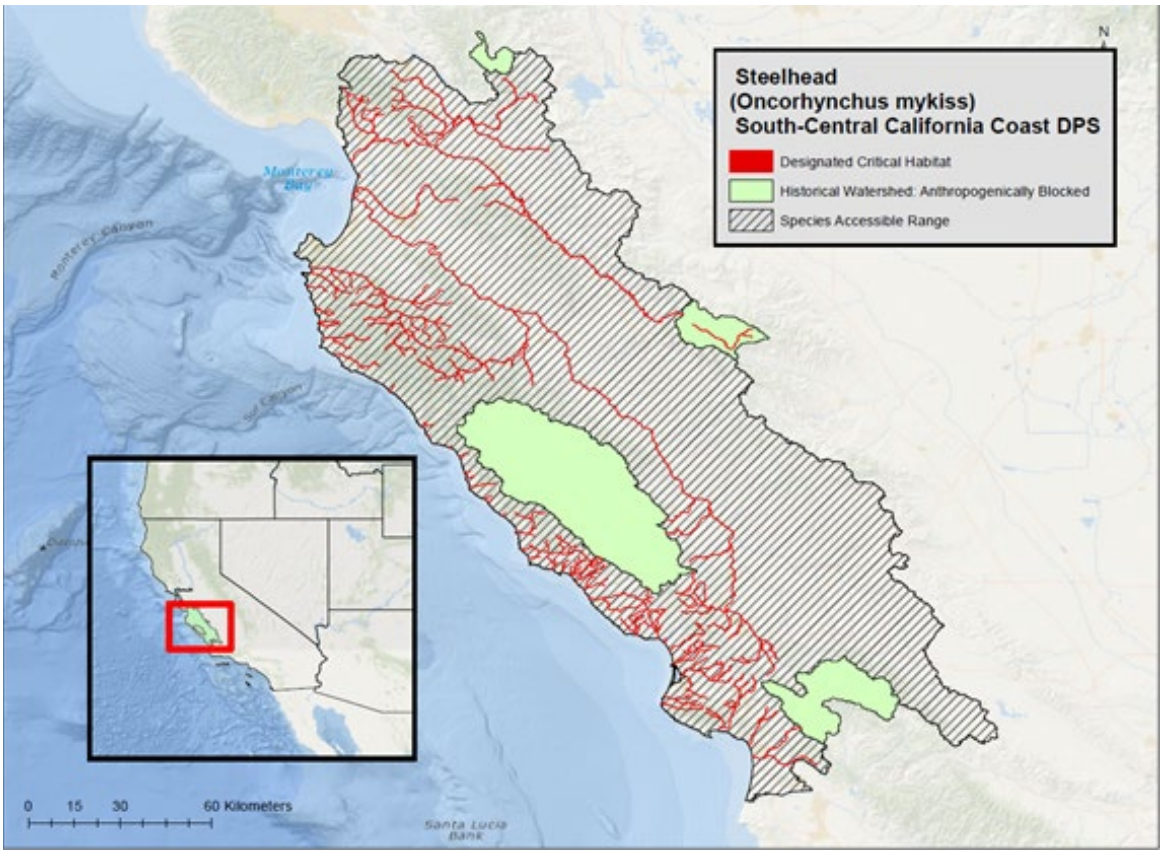


Figure 50. Geographic Inland Range and Designated Critical Habitat of South-Central California Coast Steelhead

Population Dynamics

Abundance of anadromous adults has been monitored for decades in the Carmel River at fish-passage facilities at two dams, San Clemente Dam and Los Padres Dam. The more complete counts at the lower dam (San Clemente) terminated in 2015 with the removal of the dam. Both series are partial counts due to steelhead spawning downstream of the dams. For the past 20 years there has been significant downward trend averaging -21% per year ($p = 0.0007$), and the mean run size of the most recent four years is only nine adults. Fish densities on the mainstem Carmel have been monitored for over 20 years at about 10 index sites distributed between the Los Padres Dam and the estuary. Prior to 2009 the mean density was usually above 0.3 fish/m^2 but from 2009 onward it only exceeded this threshold in two of eleven years, 2012 and 2019. These results indicate a statistically significant downward trend in fish density averaging -6% per year.

During the drought of 2012-2016, fish densities (juveniles and resident adults) in the mainstem Carmel River declined to very low levels, possibly zero in some years (SWFSC 2023). However, the wet uplands maintained even higher fish densities than the canyon section of the mainstem. This suggests that during droughts, the distribution of *O. mykiss* (rainbow trout) tends to retract into the relatively reliable habitats at high elevations, where surface flow is sustained and is also

less vulnerable to being lost into large alluvial groundwater basins (or extracted for out-of-stream uses). These findings suggest a mechanism by which population density becomes more stabilized at low abundance, by retracting into reliable drought refugia during periods of low-rainfall, and re-expanding into less reliable downstream habitats after the drought ends (SWFSC 2023). Thus the density-independence assumption of the original precautionary viability criterion may be more stringent than necessary for populations with adequate drought refugia (SWFSC 2023).

The abundance of adult steelhead has not been monitored in the Pajaro River (SWFSC 2023). Fish densities during the low-flow season were collected at 6 to 10 sites per year in lower Pajaro tributaries by Beck et al. (2019), using calibrated electrofishing. The average density dipped below 0.3 fish/m2 for five years during drought, but had recovered above this threshold as of 2019; the average density for the most recent four years was still below the threshold at 0.232 fish/m2. The combined run of the three populations in the Salinas River basin was monitored intermittently from 2011 through 2017 but no data have been reported since 2017 and the time series is too short to estimate a trend (SWFSC 2023). Run sizes in the Salinas have been extremely small, always less than 50 fish per year and zero in some years.

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 2013c). A substantial portion of the upper watersheds, which contain the majority of historical spawning and rearing habitats for steelhead, 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 2013c).

New genetic information provided insights into the mechanism by which runs of steelhead from this DPS decline to zero during adverse conditions for anadromy, and then be reconstituted by populations of rainbow trout surviving in drought refugia when conditions improve (SWFSC 2023). This is consistent with observations since listing of very low and/or erratic numbers of adult steelhead in the Salinas, Carmel, and Big Sur rivers, and is probably typical for steelhead in this region (SWFSC 2023). Unfortunately, the risk of permanently losing the anadromous phenotype over the longer term is still high and possibly increasing (SWFSC 2023). The size of steelhead runs is extremely low throughout the range of this DPS, and the mean fish densities in recent years are still below the provisional viability criterion of 0.3 fish/m^2 (SWFSC 2023).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005. Designated critical habitat for the South-Central California Coast steelhead does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

See the 2013 recovery plan for the South-Central California Coast steelhead DPS (NMFS 2013c) for complete down-listing/delisting criteria for recovery goals for the species.

4.2.24 Chinook Salmon

Three Chinook salmon Evolutionary Significant Units (ESUs) occur within the HCTT action area. We begin this section with a general discussion of Chinook salmon life history traits, followed by a discussion of the population dynamics and status of each particular ESU.

General Life History Traits

Chinook salmon are anadromous—they hatch in freshwater streams and rivers then migrate out to the saltwater environment of the ocean to feed and grow. They can grow as long as 4.9 ft and up to 129 pounds, but typical length and weight of mature fish are about 3 ft and 30 pounds. They spend a few years feeding in the ocean, then return to their natal streams or rivers to spawn, generally in summer or early fall. Chinook salmon sexually mature between the ages of two and seven but are typically three or four years old when they return to spawn. Chinook dig out gravel nests (redds) on stream bottoms where they lay their eggs. All Chinook salmon die after spawning.

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.

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.

4.2.25 Chinook Salmon – California Coastal ESU

On September 16, 1999, NMFS listed the California Coastal ESU of Chinook salmon as a "threatened" species (64 FR 50394. 1999). 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 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 51).

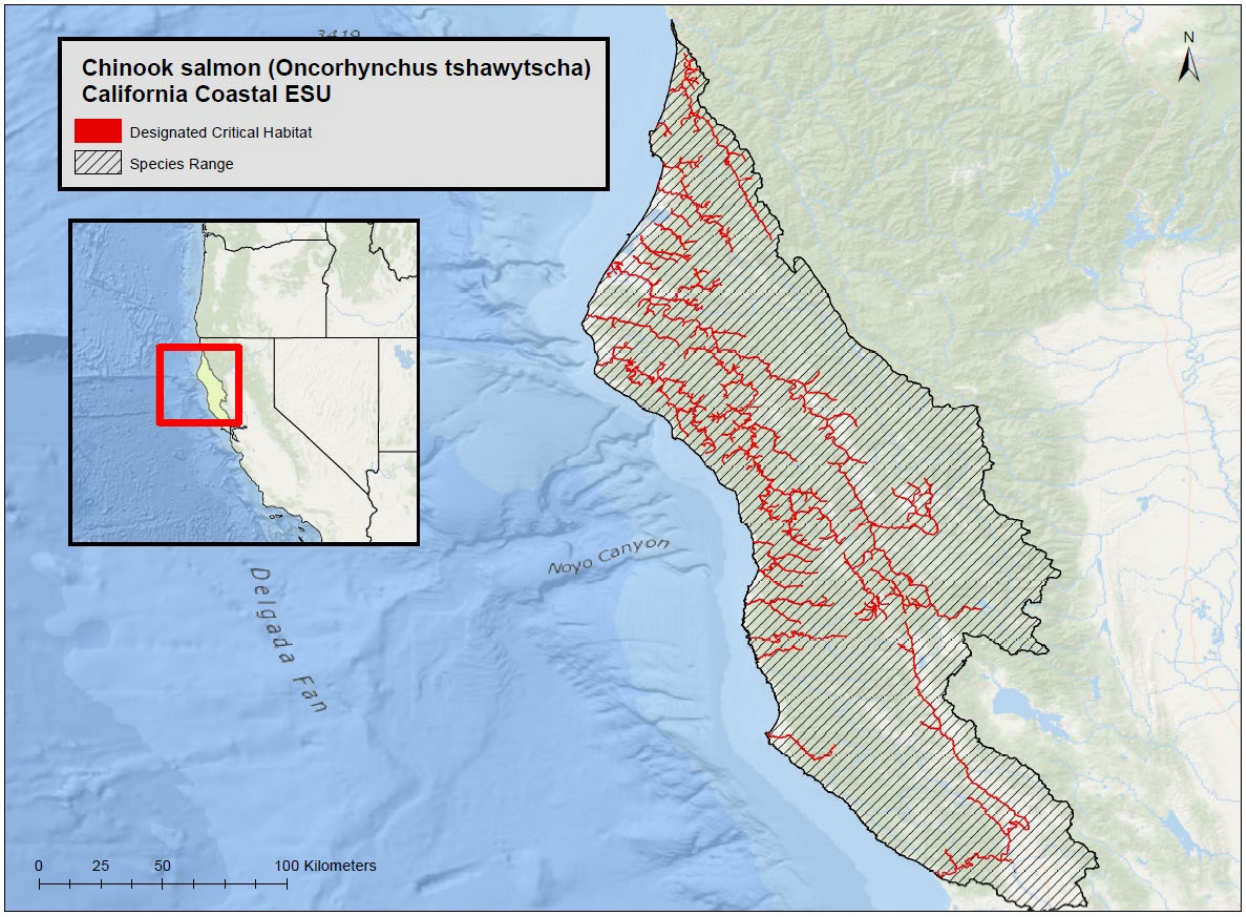


Figure 51. Geographic Inland Range and Designated Critical Habitat of California Coastal ESU Chinook Salmon

Population Dynamics

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.

Population-level estimates of abundance are currently available for three of seven independent populations of in the North Coastal stratum. Adult population estimates based on sonar counts for Redwood Creek from 2010 through 2019 averaged 2,896 (range 1,455–4,541) over 2011-2018) of sampling, showing a slightly positive, but not significant trend ($p = 0.31$) (SWFSC 2023). The population mean represents 85% of the recovery target of 3,400 spawners. Estimates

of Chinook salmon adult abundance based on sonar counts have also been made for the Mad River beginning in the 2014 season. Estimates have averaged just over 7,000 fish (range 2,169–12,667) from 2014 through 2018, and though the time series is too short for formal trend analysis, numbers have increased during this brief period (SWFSC 2023). The mean estimated abundance exceeds the recovery target of 3,000 for this population. Spawner surveys conducted in the Mattole River watershed since the 2013 spawning season resulted in redd estimates averaging 862 (range 331–2,202) per year (SWFSC 2023).

The North Mountain Interior stratum contains the upper Eel River Chinook salmon population, as well as the portion of the lower Eel River population that inhabits watersheds of the interior mountains of the Eel River basin, including the Van Duzen River and Larabee Creek basins (SWFSC 2023). A long-running time series of adult counts is available for the Van Arsdale Fish Station. From 1997 to 2019, an average of 680 Chinook salmon (range 26–3,471) have been counted, and there has been no significant trend in abundance (SWFSC 2023). A sonar-based program for estimating abundance of the Upper Eel River Chinook salmon population was initiated in 2019 and produced an estimate of 3,844 fish (SWFSC 2023).

While earlier assessments had reported the apparent extirpation of Chinook salmon populations in the North-Central Coastal Stratum, small numbers of Chinook salmon continue to return to these watersheds (SWFSC 2023).

Status

The California Coastal Chinook salmon ESU was historically comprised of 38 populations which included 32 fall-run populations and six spring-run populations across four Diversity Strata (NWFSC 2015). All six of the spring-run populations were classified as functionally independent, but are now considered extinct (NMFS 2016c). The loss of the spring-run life history type throughout the entire ESU is a significant diversity concern. 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. Recent increases in the Klamath River fall-run Chinook salmon age-4 ocean harvest rate suggests that the level of California Coastal Chinook salmon ocean fishery impacts has likely increased since the 2016 status review update (SWFSC 2023).

However, new survey data indicate that populations in the Mad and Eel River watersheds are doing better than believed in prior assessments, with the Mad River population currently at levels above recovery targets. Likewise, sonar-based abundance estimates for Redwood Creek suggest that this population is approaching its recovery target in favorable years. Trends in the longer time series are mixed, with the Freshwater Creek showing a significant decline in abundance and Van Arsdale showing no significant trend in counts over the long (23-year) or short (12-year) time series, despite having below-average counts over the last six years. Data from populations in the more southerly diversity strata indicate that most populations (all except the Russian River) have exhibited mixed trends in abundance but remain far from recovery targets (SWFSC 2023). In all Mendocino Coast populations (Ten Mile, Noyo, Big, Navarro, and Garcia rivers),

surveys have failed to detect Chinook salmon in 3–10 of the 11 or 12 years of monitoring, suggesting only sporadic occurrence in these watersheds (SWFSC 2023). Thus, concerns remain not only about the small population sizes, but the maintenance of connectivity across the ESU (SWFSC 2023). Only the Russian River population has consistently numbered in the low thousands of fish in most years, making it the largest population south of the Eel River (SWFSC 2023). In summary, the new information available indicates that recent trends across the ESU have been mixed and that overall extinction risk for the ESU is moderate (SWFSC 2023).

Critical Habitat

NMFS designated critical habitat for the California Coastal Chinook salmon on September 2, 2005 (70Fed. Reg. 52488). Designated critical habitat for the California Coastal Chinook salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

Recovery goals, objectives and criteria for the California Coastal chinook salmon are fully outlined in NMFS (2016f). 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.

4.2.26 Chinook Salmon – Central Valley Spring-Run ESU

On September 16, 1999, NMFS listed the Central Valley ESU of spring-run Chinook salmon as a "threatened" species (FR 64 50394). 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 52).

Population Dynamics

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 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. 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 2014b). The total estimated abundance of adult Central Valley spring-run Chinook salmon for the Sacramento

River watershed in 2019 was 26,553, approximately half of the population size in 2014 (N = 56,023).

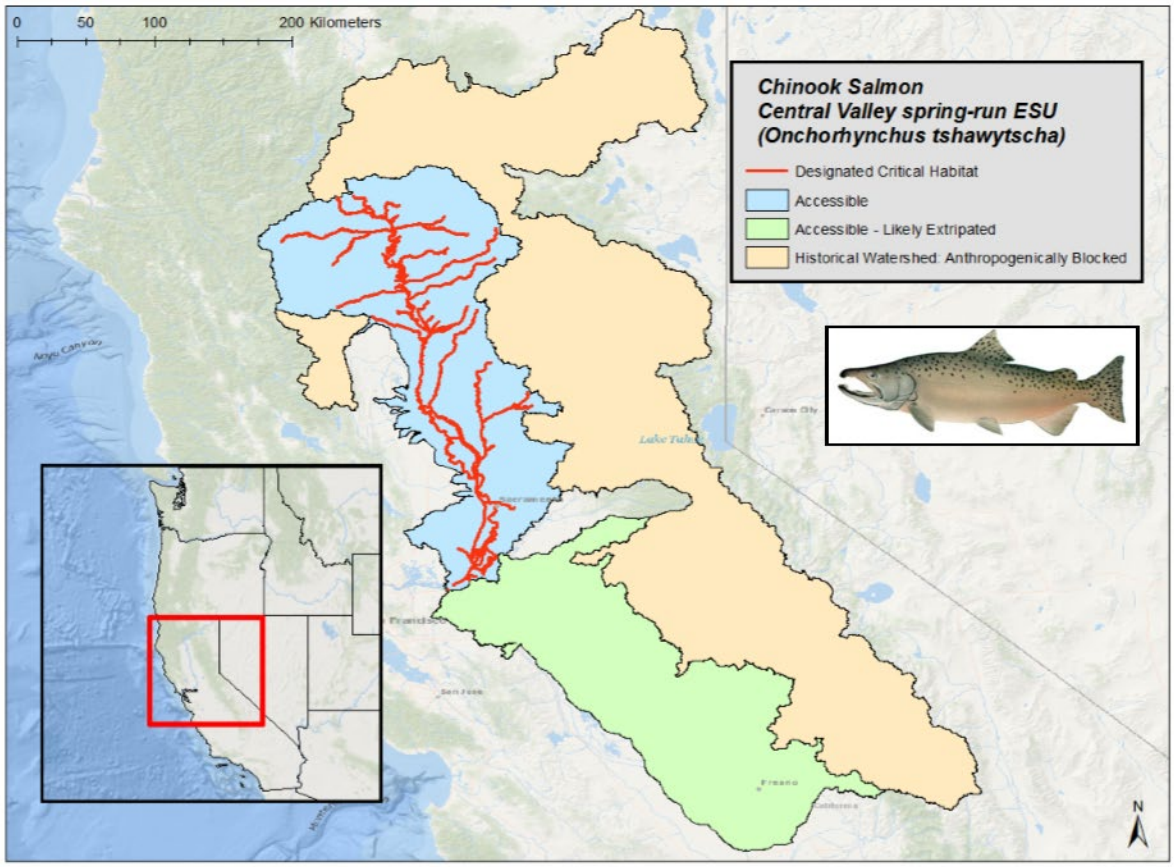


Figure 52. Geographic Inland Range and Designated Critical Habitat of Central Valley Spring-Run ESU Chinook Salmon

The Central Valley-wide abundance is driven largely by the annual variation in adults returning to Butte Creek. Prespawn mortality in 2021 was 92%, representing a biologically significant loss in spawning potential (SWFSC 2023). The rate of decline increased from 5% per year through 2019, to 19% over the most recent 10 years, indicating that the Butte Creek population is at a high risk of extinction (SWFSC 2023).

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 (64 FR 50394. 1999; Myers et al. 1998). 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 2014b).

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).

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 (Yoshiyama et al. 1998; Moyle 2002). 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).

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 changes in environmental conditions are likely to adversely affect spring-run Chinook salmon and their recovery (NMFS 2014b).

The viability of this ESU has declined since the 2015 assessment and the ESU is at greater risk of extinction (SWFSC 2023). The largest impacts are likely due to the freshwater drought conditions and unusually warm ocean conditions experienced by these cohorts, resulting in weakening viability metrics and greater risks of extinction to the majority of the populations (SWFSC 2023). The recent declines of many of the dependent populations, high pre-spawn mortality and poor juvenile survival during the 2012–2016 drought, unknown impacts due to

warm ocean conditions and reorganization of coastal marine food webs, are all causes for increased concern for the long-term viability of the ESU (SWFSC 2023). Extinction risks are of concern due to the low abundance of individuals, the magnitude of the abundance decline observed in recent years, and the ESU's pre-existing vulnerability. Due to the factors discussed above, the Central Valley spring-run Chinook salmon ESU remains at moderate to high risk of extinction (SWFSC 2023).

Critical Habitat

NMFS published a final rule designating critical habitat for Central Valley spring-run Chinook salmon on September 2, 2005. Designated critical habitat for the Central Valley spring-run Chinook salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

Recovery goals, objectives and criteria for the Central Valley spring-run Chinook salmon are fully outlined in the 2014 Recovery Plan (NMFS 2014b). 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.

4.2.27 Chinook Salmon – Sacramento River Winter-Run ESU

On January 4, 1994, NMFS listed the Sacramento River winter-run ESU of Chinook salmon as Endangered (59 FR 440. 1994). 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 53). 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.

Population Dynamics

The abundance of Sacramento River winter-run Chinook salmon has declined in the past few decades during recent periods of unfavorable ocean conditions (2005–2006) and droughts (2007–2009, 2012–2016) (SWFSC 2023). Temperature conditions during egg development and fry emergence were suboptimal for this ESU over the duration of rearing in 2014 and 2015, reaching lethal levels in both years due to reduced cold-water releases from Shasta Reservoir for this life stage (SWFSC 2023). The egg-to-fry survival estimate for brood year 2014 is 5%, which is significantly lower than the average of 26.4% for brood years 2002–2012 (SWFSC 2023). Sacramento River winter-run Chinook salmon declined in abundance since 2005 with a recorded

low of 795 in-river spawners in 2017. Escapement improved in 2018 and 2019 such that the mean population sizes for the years 2017-2019 (3,882 spawners) satisfy the low-risk criterion (i.e., N > 2500) (SWFSC 2023). However, the rising proportion of hatchery fish among returning adults threatens to increase the risk of extinction. The numbers of natural-origin spawners in 2017 and 2018 were low (153 and 461 individuals), indicating the significant contribution of hatchery-origin fish during years of high in-river mortality (SWFSC 2023). The decrease in hatchery reliance in the 2019 returns and the overall return of a larger number of natural-origin adults (run = 5,000 individuals) indicates there is potential for naturally spawning Sacramento River winter-run Chinook to rebuild during periods of favorable environmental conditions (SWFSC 2023).

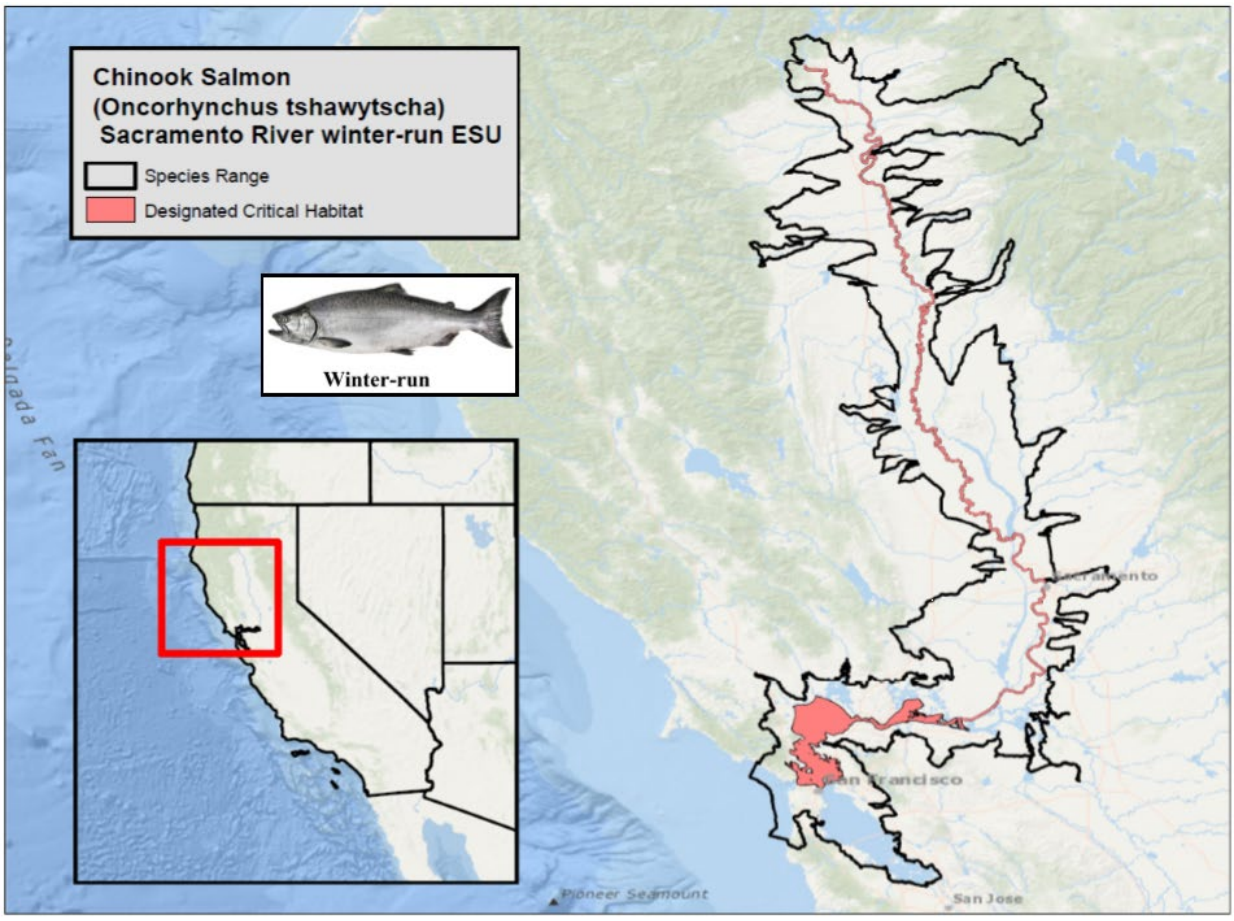


Figure 53. Geographic Inland Range and Designated Critical Habitat of the Sacramento River Winter-Run ESU of Chinook Salmon

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).

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 (USFWS 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 California Department of Fish and Wildlife annual escapement surveys (2000-2006). 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).

Status

Spatial structure and diversity are central tenants to the viability of populations and ESUs. The lack of population redundancy in the Sacramento River winter-run Chinook ESU is the primary factor contributing to its high extinction risk. The Sacramento River winter-run Chinook salmon ESU is composed of just one small population that experienced severe stress caused by California's 2011 to 2017 drought, one of California's worst droughts on record. 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). 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. Yet, reliance on production from the hatchery is increasing and remains a cause for serious concern to the long-term viability and genetic integrity of the population and the ESU (SWFSC 2023). In the summer of 2020, juvenile salmon were observed in Battle Creek indicating the first successful spawning of winter-run Chinook salmon in Battle Creek in over 100 years (SWFSC 2023). Until additional Sacramento River winter-run Chinook populations are established, the ESU will remain in the "high" biological extinction risk category (SWFSC 2023).

Critical Habitat

NMFS designated critical habitat for the Sacramento winter-run Chinook salmon on June 16, 1993 (58Fed. Reg. 33212). Designated critical habitat for the Sacramento winter-run Chinook salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

Recovery goals, objectives and criteria for the Sacramento River winter-run Chinook salmon are fully outlined in the 2014 Recovery Plan (NMFS 2014b). 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.

4.2.28 Coho Salmon

Three coho salmon Evolutionary Significant Units (ESUs) occur within the HCTT action area. We begin this section with a general discussion of coho salmon life history traits, followed by a discussion of the population dynamics and status of each particular ESU.

General Life History Traits

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. 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)

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 mm 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).

After one year in freshwater juvenile coho salmon undergo physiological transformation into "smolts" for outmigration to the ocean in the spring. 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). Coho salmon typically spend two 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 six months at sea (NMFS 2014a).

4.2.29 Coho Salmon - Central California Coast ESU

The Central California Coast coho salmon ESU was listed as threatened under the ESA on October 31, 1996 (64Fed. Reg. 56138). NMFS re-classified the ESU as endangered on June 28,

2005 (70 FR 37160. 2005). This 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. Coho salmon from three artificial propagation programs are also included in this ESU (Figure 54).

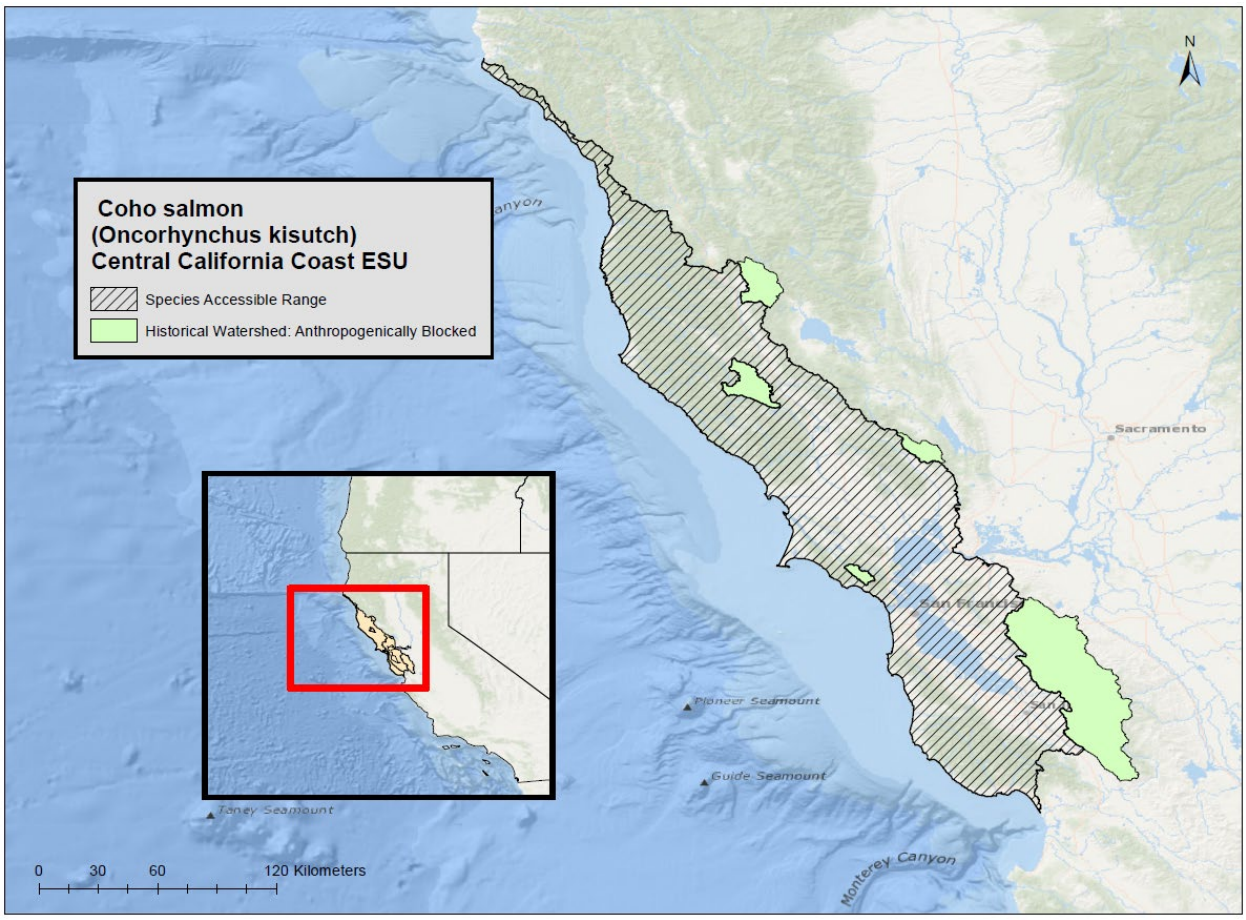


Figure 54. Geographic Inland Range of Coho Salmon Central California Coast ESU

Population Dynamics

Limited information exists on the abundance of coho salmon within the Central California Coast ESU. Abundance estimates for the Navarro Point Diversity Stratum averaged 428 fish (range 2–843) between the 2009 and 2018 spawning seasons, with no overall trend in abundance over this time period (SWFSC 2023). These estimates do not include the Gualala River watershed, which has not been monitored, although coho salmon are believed to be either extirpated or at very low numbers in this watershed (SWFSC 2023). The Navarro Point Diversity Stratum average abundance is roughly 5% of the downlisting spawner target and 3% of the delisting spawner target identified for the stratum in the Central California Coast ESU coho salmon recovery plan (NMFS 2012a).

No stratum-wide estimates of abundance are available for the Coastal stratum; however, given the population-level information available, it is clear this stratum is only at a small fraction of its recovery target of 15,300 (SWFSC 2023). The Russian River and Lagunitas Creek are the two largest watersheds within the Coastal Stratum. From 1998-2019 the redd count in Lagunitas Creek averaged 247 (range 26–634) (SWFSC 2023). Assuming an average spawner:redd ratio of 2:1, this average equates to approximately 20% of the recovery target of 2,600 for this population (SWFSC 2023). The population is considered at moderate risk based on the effective population size criterion. The long-term trend is slightly downward, though not significant. However, within the past ten years (2011–2019), the population appears to have increased from a low reached during the 2009 spawning season (SWFSC 2023). 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). Over the five years of surveys (2015-2019), an average of 128 redds have been estimated annually, which includes redds produced by both hatchery- and natural-origin fish. Assuming an average spawner:redd ratio of 2:1, adult numbers are less than 3% of the recovery target for this population (SWFSC 2023).

Stratum-level estimates of abundance are not available for the Santa Cruz Mountain stratum but it is clear that natural production of coho throughout the region is extremely low (SWFSC 2023).

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).

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 (SWFSC 2023). The available data for populations within the Central California Coast coho salmon ESU indicate that all independent and dependent populations remain far below recovery targets for abundance and, in some cases, are below high-risk thresholds. Recent data from the Lost-Coast-Navarro Point and Navarro Point-Gualala Point diversity strata suggest slight improvement in viability of independent populations from 2015-2019, with most populations having rebounded somewhat since low levels reached during California's multi-year drought between 2012 and 2015 (SWFSC 2023). Smolt-to-adult survival estimates along the Mendocino Coast indicate that marine survival of coho salmon was extremely low from brood years 2004 to 2008 (i.e., smolt outmigration years 2005–2009), but have since risen to levels more typically seen, even in years corresponding to the marine heat wave (SWFSC 2023). Overall, the available new information since the 2016 viability assessment indicates the extinction risk has not changed appreciably, with slight improvements in the two northern-most diversity strata, but little change in the Coastal Diversity Stratum and perhaps worsening conditions in the Santa Cruz Mountain Stratum (SWFSC 2023). The extinction risk for Central California Coast coho salmon as a whole thus remains high (SWFSC 2023).

Critical Habitat

Critical habitat for the Central California Coast ESU of coho salmon was designated on May 5, 1999 (64Fed. Reg. 24049). Designated critical habitat for the Central California Coast ESU of coho salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

See the 2012 Recovery Plan for complete down listing/delisting criteria for each of the following recovery goals (NMFS 2012a):

- Prevent extinction by protecting existing populations and their habitats;
- Maintain current distribution of coho salmon and restore their distribution to previously occupied areas essential to their recovery;
- Increase abundance of coho salmon to viable population levels, including the expression of all life history forms and strategies;
- Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta populations;
- Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally;
- Ensure all factors that led to the listing of the species have been ameliorated; and
- 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.

4.2.30 Coho Salmon - Oregon Coast ESU

The Oregon Coast ESU of coho salmon was listed as threatened under the ESA on August 10, 1998 (63Fed. Reg. 42587). The listing was revisited and confirmed as threatened on June 20, 2011 (76Fed. Reg. 35755). 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 55).

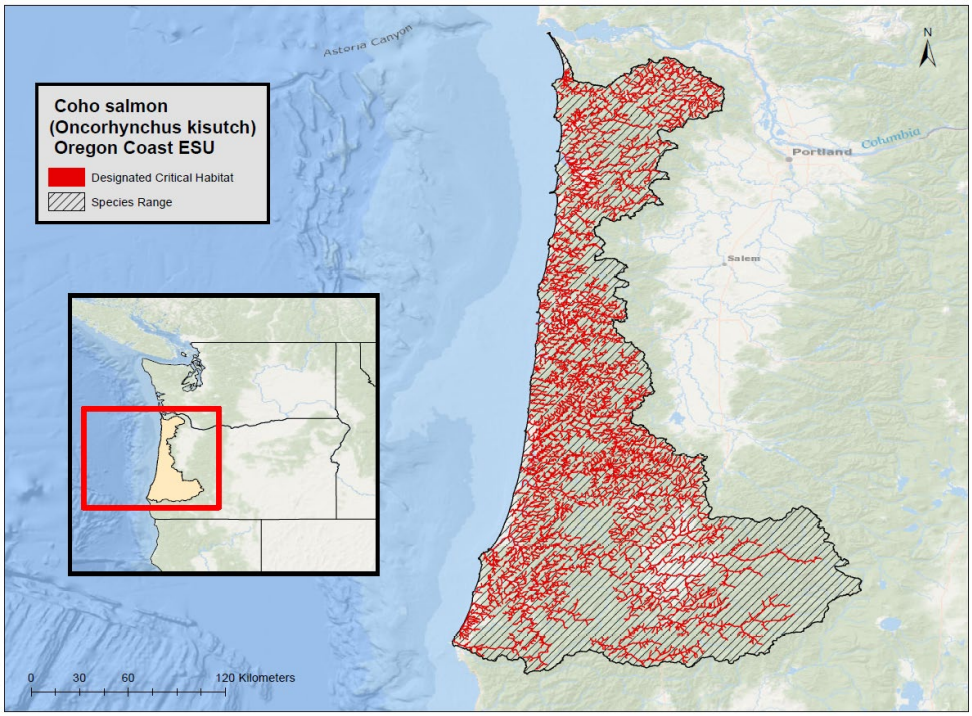


Figure 55. Geographic Inland Range and Designated Critical Habitat of Coho Salmon Oregon Coast ESU

Population Dynamics

Prior to 1940, recruitment of adults to the Oregon Coast coho salmon ESU is estimated to have averaged about 800,000 fish, ranging from 400,000–2,000,000 (Ford 2022). After 1940, typical recruitments dropped to about 300,000. Abundance and harvest started to decline in the 1980s until fisheries were closed in 1993 due to extreme low abundance and poor marine survival (Ford 2022). Oregon Coast coho salmon reached extremely low levels in the late 1990s, when 22,800–47,400 spawners were estimated across the entire ESU (Ford 2022).

While Oregon Coast coho salmon natural spawner abundances have shown an increasing trend since the early 1990's, all populations exhibited a substantial decrease in the geometric mean abundance between the 5-year period 2010–14 and the 5-year period 2015–19 (Ford 2022). Marine survival appears to be the principal driver of variation in abundance, and poor ocean conditions since 2014 have again resulted in lower spawner abundances, but not as low as those observed in the late 1990s (Ford 2022).

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 2016f). The large populations (abundances >6,000 spawners since 2015) include Nehalem, Tillamook Bay, Alsea, Siuslaw, Lower Umpqua, Coos, and Coquille (Ford 2022).

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.

Status

Based on the 2022 assessment (Ford 2022), Oregon Coast coho salmon ESU has a high certainty of persistence and a low to moderate certainty of sustainability. NMFS's analysis of the ESA section 4(a)(1) factors indicates that the collective risk to the persistence of this ESU has slightly improved since the 2015 5-year review. They note that this assessment was made in light of very poor ocean and freshwater survival conditions from marine heatwaves and droughts over the last five to seven years. Ford (2022) noted that, despite these adverse conditions, Oregon Coast coho salmon fared better than many other ESA-listed salmonid ESUs, and have shown a remarkable ability to avoid the extremely low abundances and marine survival rates observed in the late 1990s during the previous extended downturn.

Despite these somewhat optimistic results for Oregon Coast coho salmon ESU, due to ongoing threats and scientific uncertainty, the future of this population remains in question. A recent assessment of the vulnerability of ESA-listed salmonid species to changes in environmental conditions indicate that Oregon Coast coho salmon ESU have high overall vulnerability, high biological sensitivity and climate exposure, but only moderate adaptive capacity (Siegel and Crozier 2020). Because young coho spend a full year in freshwater before ocean entry, the juvenile freshwater stage is considered to be highly vulnerable. For all populations, reduced juvenile habitat complexity in the winter (and for some populations the summer period) from land management activities is the top key limiting factor for freshwater habitat. The ESU also scored high in sensitivity at the marine stage due to expected changes due to ocean acidification. Based on the latest assessment (Ford 2022), the overall risk of extinction for the Oregon Coast coho salmon ESU remains "moderate-to-low."

Critical Habitat

NMFS published a final rule designating critical habitat for Oregon Coast coho salmon on February 11, 2008 (70 FR 52488. 2005). Designated critical habitat for the Oregon Coast coho salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

See the 2016 Oregon Coast Coho Salmon Recovery Plan for detailed descriptions of the recovery goals and delisting criteria (NMFS 2016f). The plan states that the Oregon Coast coho salmon will be delisted when NMFS determines 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.

4.2.31 Coho Salmon - Southern Oregon and Northern California Coasts ESU

The Southern Oregon/Northern California Coast (SONCC) ESU of coho salmon was listed as threatened under the ESA on May 6, 1997 (62Fed. Reg. 24588), and the listing was revisited and confirmed as threatened on June 28, 20057160)(37160). This ESU includes naturally spawned coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California (Figure 56), as well as coho salmon from three artificial propagation programs.

Population Dynamics

The available data indicate that the independent populations within the SONCC coho salmon ESU with adequate data to determine a population-specific estimate of abundance remain below recovery targets (SWFSC 2023) and, in two cases (Shasta River and Mattole River), are below the high-risk thresholds in the recovery plan (NMFS 2014a). Of the seven time series available for this assessment, positive abundance trends were observed in the Elk and Scott rivers; the Elk River abundance trend was significantly different from zero, although the annual average abundance (166) and most recent 12-year average abundance (296) are well below the population recovery target of 2,400 (SWFSC 2023). The 12-year average of 670 fish in the Scott River is well below the recovery target of 6,500 adults and the population is categorized as at moderate-risk of extinction for the population size per generation criterion (SWFSC 2023). The remaining five populations had negative abundance trends, although only the Shasta River population trend was significantly different from zero (SWFSC 2023).

The Rogue Basin short-term (12-year) abundance trend is positive and the long-term (23-year) abundance trend is negative, but neither trend is significantly different from zero (SWFSC 2023). The Trinity River Basin short-term abundance trend and the long-term abundance trend are both negative, with the most recent 12-year time series having a significant negative trend with extremely high levels of hatchery-origin fish (SWFSC 2023). The Trinity River Basin 12-year average abundance estimate of 1,116 is 12% of the recovery target for the diversity stratum of

9,700 fish (SWFSC 2023). An estimated 97% of adults in this basin are hatchery-origin fish (SWFSC 2023).

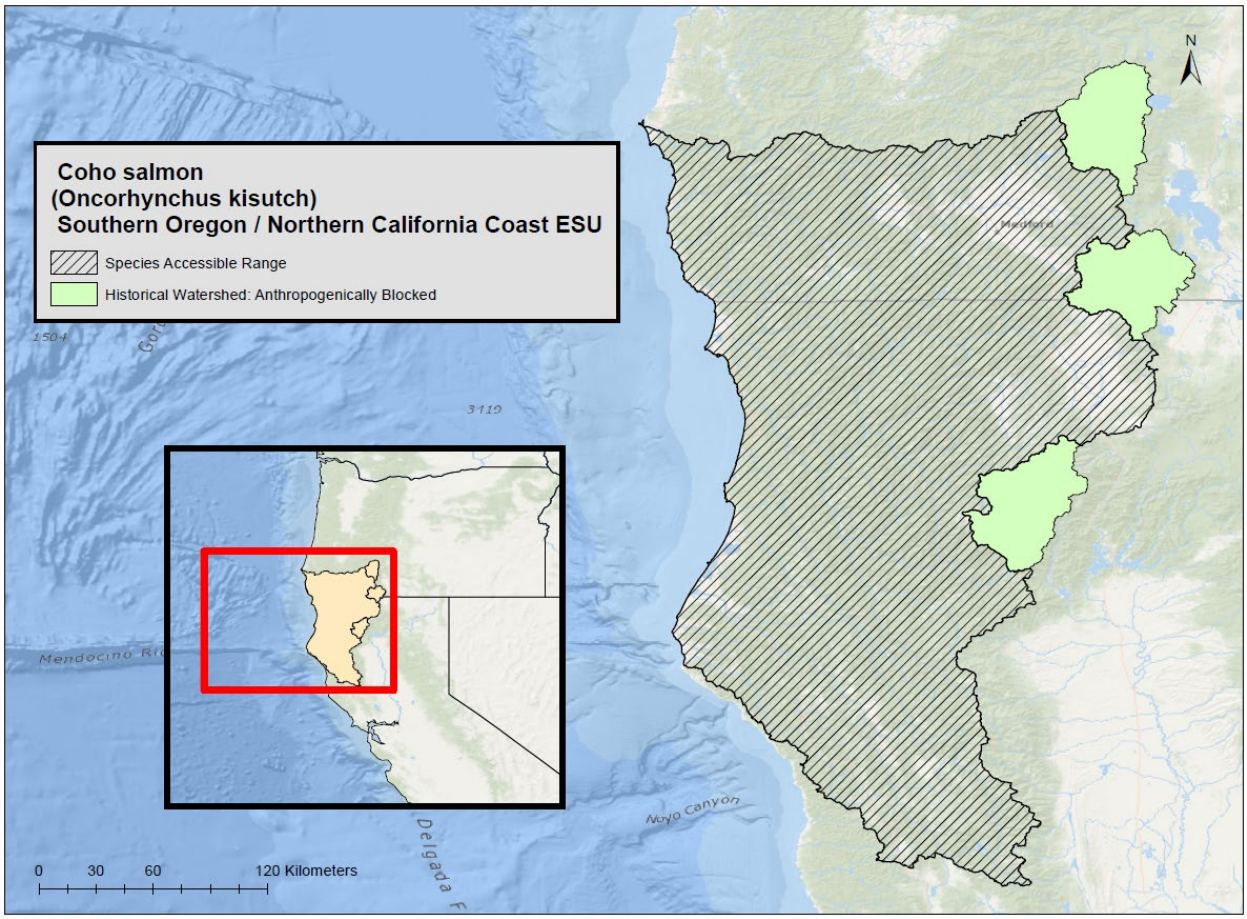


Figure 56. Geographic Inland Range of the Southern Oregon/Northern California ESU of Coho Salmon

Status

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 SONCC ESU of coho salmon. 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). The lack of increasing abundance trends across the ESU for most populations with adequate data should be considered in the context of the relatively poor ocean, freshwater, and climate conditions experienced over the past 10 to 15 years (SWFSC 2023). The SONCC coho salmon ESU extinction risk trend is declining (i.e., less viable) and this population remains at a moderate risk of extinction (SWFSC 2023).

Critical Habitat

NMFS designated critical habitat for the SONCC ESU of coho salmon on May 5, 1999 (64Fed. Reg. 24049). Designated critical habitat for the SONCC ESU of coho salmon does not overlap spatially with the HCTT action area and, therefore, will not be analyzed further in this opinion.

Recovery Goals

NMFS developed biological recovery objectives (NMFS 2014a) for SONCC coho salmon based information on ESU and population viability metrics. At the ESU level, SONCC coho salmon must demonstrate representation, redundancy, and connectivity (Williams et al. 2008). Representation relates to the genetic and life-history diversity of the ESU, which is needed to conserve its adaptive capacity. Redundancy addresses the need to have a sufficient number of populations so the ESU can withstand catastrophic events. Connectivity refers to the dispersal capacity of populations to maintain long-term demographic and genetic processes. Each SONCC coho salmon population must achieve sufficient abundance, growth rates, spatial structure, and diversity. The minimum needed conditions for each population vary depending on each population's role in recovery (i.e., core, non-core and dependent populations). For details on the recovery plan for SONCC coho salmon see NMFS (2014a).

4.2.32 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 57) (NMFS 2016d). The Southern DPS was first listed as threatened by NMFS on March 18, 2010 (75Fed. Reg. 13012).

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 between December and June (Gustafson 2016). 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 (Wydoski and Whitney 1979; Gustafson 2016). Adult eulachon are found in coastal and offshore marine habitats. Eulachon movements in the ocean are poorly known, although the amount of eulachon bycatch in the pink shrimp fishery seems to indicate that the distribution of these organisms overlap in the ocean. 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% 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.

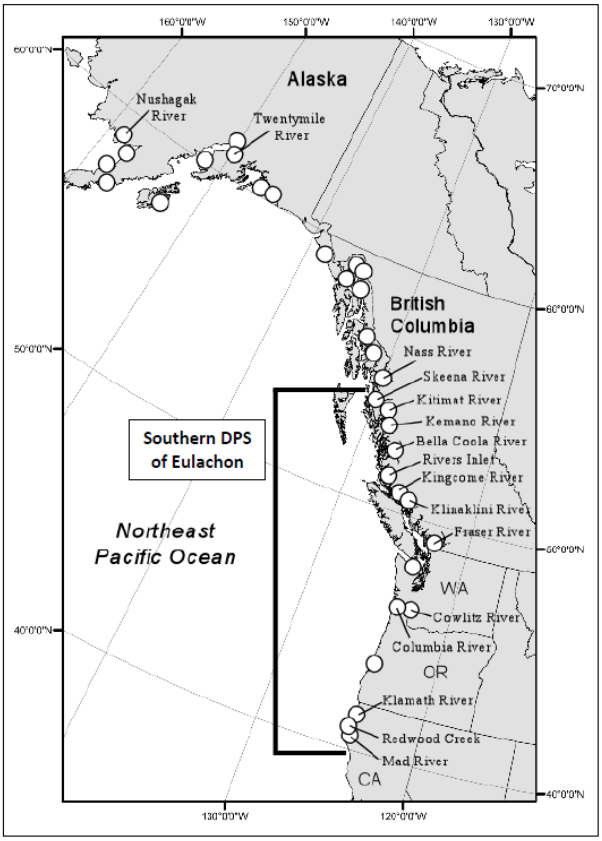


Figure 57. Map Identifying the Spawning Rivers of the Eulachon Southern DPS (NMFS 2016d)

Population Dynamics

In the early 1990s, there was an abrupt decline in the abundance of eulachon returning to the Columbia River with no evidence of returning to their former population levels since then (Drake et al. 2008). Eulachon abundance in monitored rivers has generally improved, especially in the 2013-2015 return years. However, recent poor ocean conditions and the likelihood that these conditions will persist into the near future suggest that population declines may be widespread in the upcoming return years. Therefore, it is too early to tell whether recent improvements in the status of southern DPS of eulachon will persist or whether a return to the severely depressed abundance years of the mid-late 1990s and late 2000s will recur (NMFS 2017d).

Southern DPS eulachon spawn in rivers south of the Nass River in British Columbia to the Mad River in California (Figure 57). In the portion of the species' range that lies south of the U.S.–Canada border, most eulachon production originates in the Columbia River Basin (Gustafson et al. 2010). 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 2020a). 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. The combined spawner estimate from the Columbia and Fraser rivers is 18,796,090 eulachon (NMFS 2020a).

Historically, the only other large river basins in the U.S. where large, consistent spawning runs of eulachon have been documented are the Klamath River in northern California and the Umpqua River in Oregon (NMFS 2017c). Eulachon may have historically occurred in the Sacramento River system and even farther south along the California and Baja California coast, but the lack of reported catches in recent decades suggests the species may have been extirpated from these areas (NMFS 2017c).

Adult and juvenile Southern DPS eulachon can be found in the Pacific Ocean, along the continental shelf, in waters from 50 to 200 m deep (Gustafson 2016). Information on the at-sea distribution of Southern DPS eulachon off the U.S. West Coast is lacking. Since at-sea directed harvest of eulachon in U.S. West Coast waters is prohibited, and the species is not an actively managed or monitored species by the Pacific Fishery Management Council, most information regarding offshore distribution comes from by fisheries bycatch data (NMFS 2017c). Bycatch of eulachon in the California ocean shrimp fishery fluctuated widely from 2011-2019, ranging from a high 2.2 million fish caught in 2015 to a low of 31 fish in 2017 (Gustafson et al. 2022). The tonnage of observed ocean shrimp and of fleetwide ocean shrimp landings was relatively stable during these years, indicating that yearly differences in eulachon distribution, or in the catchability of eulachon, likely contributed to the extreme fluctuations in eulachon bycatch in the California ocean shrimp fishery (Gustafson et al. 2022).

Southern DPS eulachon are genetically distinct from eulachon in the northern parts of its range (i.e., Alaska). Genetic analysis indicates that the Southern DPS exhibits a regional population structure, with separation among three main regions: from Northern California up to and including the Fraser River; north of the Fraser River to southeast Alaska; and within the Gulf of Alaska (Candy et al. 2015; Gustafson 2016; Sutherland et al. 2021).

Status

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. Warmer water temperatures associated with changing environmental conditions could alter the timing of spawning, and the availability of prey for larval and juvenile eulachon (NMFS 2016d). Further population decline is anticipated to continue as a result of changing environmental conditions and bycatch in commercial fisheries. However, the near-term outlook for eulachon productivity in the California Current Ecosystem is more positive, based on the presence of good ocean conditions (NMFS 2022a). The productivity potential as indicated by life history characteristics such as low age-at- maturity, small body size, planktonic larvae, and perhaps their high fecundity, confers eulachon with some resilience to

environmental perturbations, as they retain the ability to quickly respond to favorable ocean conditions (Bailey and Houde 1989; NMFS 2022a).

Critical Habitat

On December 19, 2011, NMFS established critical habitat the southern DPS of Pacific eulachon in rivers and estuaries in the states of California, Oregon, and Washington (76Fed. Reg. 65323). This critical habitat is not present within the action area.

Recovery Planning

The stated goals of the recovery plan for the southern DPS of eulachon are to: 1) Increase the abundance and productivity of eulachon; 2) Protect and enhance the genetic, life history, and spatial diversity of eulachon throughout its geographical range; and 3) Reduce existing threats to warrant delisting of the species. See the 2017 Recovery Plan for the Southern DPS eulachon, for complete down listing/delisting criteria for each of their respective recovery goals (NMFS 2017d).

5 ENVIRONMENTAL BASELINE

The *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 impacts to listed species or designated critical habitat from Federal agency activities or existing Federal agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR §402.02).

5.1 Large-Scale Changes in Environmental Conditions

Large-scale changes in environmental conditions have, and likely will continue to, adversely affect ESA-listed species and habitats within the HCTT action area. The global average surface temperature was 1.09°C higher in 2011–2020 than 1850–1900, with larger increases over land (1.59°C) than over the ocean (0.88°C). Global surface temperature has increased faster since 1970 than in any other 50-year period over at least the last 2000 years (Lee et al. 2023). As warming increases, the occurrence of marine heat waves, ocean acidification and deoxygenation are also likely to increase (Lee et al. 2023).

5.1.1 Acidification

The global atmospheric carbon dioxide concentration has now passed 400 parts per million, a level that last occurred about three million years ago, when both global average temperature and sea level were significantly higher than today. An increasing global atmospheric carbon dioxide

concentration also drives an increase in ocean acidity, which has increased by 26% since the beginning of the industrial era (Intergovernmental Panel on Climate Change 2014; IPCC 2014). Changes in ocean acidity change the way chemical reactions occur in the ocean, and this can affect the way life functions in the ocean. Many animals make shells or other hard structures that include calcium carbonate, which becomes more difficult in an acidic environment; requiring more of the animal's energy and resulting in weaker shells or structures (Doney et al. 2012). There is broad consensus that the further and the faster the Earth system is pushed by the increasing carbon dioxide concentrations, the greater the risk of changes and impacts, some of which are potentially large and irreversible (Wuebbles et al. 2017).

Significant impacts to marine mammals and sea turtles from ocean acidification will be indirectly tied to foraging opportunities resulting from ecosystem changes (Busch et al. 2013; Haigh et al. 2015; Chan et al. 2017). Nearshore waters off California have already shown a persistent drop in pH from the global ocean mean pH of 8.1 to as low as 7.43 (Chan et al. 2017). The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, and shrimp), which have in turn been linked to oceanographic features affected by changing environmental conditions (Learmonth et al. 2006). Ocean acidification may cause a shift in plankton community composition and biochemical composition that can impact the transfer of essential compounds to predators that eat plankton (Bermúdez et al. 2016). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1986; Payne et al. 1990; Clapham et al. 1999). Krill have been shown to suffer decreased larval development and survival under lower pH conditions (McLaskey et al. 2016). Krill also have lower metabolic rates after both short-term and long-term exposure to low pH (Cooper et al. 2016). Increased ocean acidification may also have serious impacts on fish development and behavior (Raven et al. 2005), including sensory functions (Bignami et al. 2013) and fish larvae behavior that could impact fish populations (Munday et al. 2009) and piscivorous ESA-listed species that rely on those populations for food.

5.1.2 Sea Level and Freshwater

Global average sea level has risen by about seven to eight inches since 1900, with almost half (about three inches) of that rise occurring since 1993, a rate of rise that is greater than during any preceding century in at least 2,800 years (Wuebbles et al. 2017). As global temperatures warm, seawater is expanding, but the biggest contribution to sea level rise is from freshwater added by melting ice sheets and glaciers. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years, and by one to four feet by 2100.

Some ESA-listed marine species rely on terrestrial habitats where they can haul out close to the water's edge for important life stages, such as nesting for sea turtles and pupping for pinnipeds. These habitats are threatened by sea level rise that is exacerbated by storm surges (Blondin et al. 2022). As these habitats shrink, they can result in crowding which creates stress during these sensitive life stages and opportunities for predators. Monk seals may experience more crowding

and competition for landing sites when islands shrink. This habitat loss can be particularly consequential for areas of low-lying remote islands without adequate alternates nearby.

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly impacting sea turtle nesting habitat (Wilkinson and Souter 2008). In some areas, increases in sea level alone may be sufficient to inundate turtle nests and reduce hatching success by creating hypoxic conditions within inundated eggs (Caut et al. 2009; Pike et al. 2015). Flatter beaches, preferred by smaller sea turtle species, would likely be inundated sooner than would steeper beaches preferred by larger species (Hawkes et al. 2014). Relatively small increases in sea level can result in the loss of a large proportion of nesting beaches in some locations. For example, a study in the northwestern Hawaiian Islands predicted that up to 40% of green turtle nesting beaches could be flooded with a 0.9 m sea level rise (Baker et al. 2006). The loss of nesting beaches would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form, or if the newly formed beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. Poloczanska et al. (2009) noted that extant marine turtle species have survived past climatic shifts, including glacial periods and warm events, and therefore may have the ability to adapt to ongoing changes in environmental conditions (e.g., by finding new nesting beaches). However, the authors also suggested since the current rate of warming is very rapid, expected changes may outpace sea turtles' ability to adapt.

Changing patterns of coastal erosion and sand accretion, combined with an anticipated increase in the number and severity of extreme weather events, may further exacerbate the effects of sea level rise on turtle nesting beaches (Wilkinson and Souter 2008). Changing environmental conditions are expected to affect the intensity of hurricanes through increasing sea surface temperatures, a key factor that influences hurricane formation and behavior (EPA 2010). Extreme weather events may directly harm sea turtles, causing "mass" strandings and mortality (Poloczanska et al. 2009). Studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to changing environmental conditions (Hawkes et al. 2009).

Freshwater from ice melt at high latitudes makes the ocean surface waters there less dense (less salty) which decreases the ability for the waters to sink and drive major global ocean currents. These currents impact the availability of nutrients that drive primary biological productivity, such as plankton blooms, which in turn impact major ecosystems that rely on that productivity, all the way up the food chain to large consumers like whales (Gennip et al. 2017). Increased sea surface temperature can alter patterns of stratification (water layers) and upwelling (vertical water movement), which influence the amount of nutrients supplied to surface waters from the deep sea leading to changes in plankton productivity and the fish populations that consume plankton (EPA 2010), and, therefore, declines in those species whose diets are dominated by fish.

5.1.3 Range Shift

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). Shifts in marine mammal range and distribution due to changing environmental conditions have been observed in some populations (Silber et al. 2017). Marine mammal species often exhibit strong dependence on or fidelity to particular habitat types, oceanographic features, and migration routes (Sequeira et al. 2018). Specialized diets, restricted ranges, or reliance on specific substrates or sites (e.g., for pupping) make many marine mammal populations particularly vulnerable to changing environmental conditions (Silber et al. 2017). Marine mammals with restricted distributions linked to water temperature may be exposed to range restriction (Learmonth et al. 2006; Issac 2009). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by changing environmental conditions, 47% would be negatively affected, and 21% would be put at risk of extinction. 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. He predicted up to a 35% 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 sea turtles were predicted to gain core habitat area, whereas loggerhead sea turtles and blue whales were predicted to experience losses in available core habitat. Such range shifts could affect marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Pike 2014; Birney et al. 2015). Species distributional shifts due to changing environmental conditions can alter bioenergetic demands and influence mortality events (Fedewa et al. 2020; Szuwalski et al. 2023).

Shifting ranges of important prey item for marine mammals have been observed across all ocean regions (Poloczanska et al. 2016). For ESA-listed species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009). Blue whales, as predators that specialize in eating krill, are likely to change their seasonal occurrence and distribution in response to changes in the distribution of krill (Payne et al. 1986; Payne et al. 1990; Clapham et al. 1999; Barlow et al. 2020; van Weelden et al. 2021). Abrahms et al. (2019) compared 10 years of blue whale movement data with the timing of the spring phytoplankton bloom resulting in increased prey availability in the California Current Ecosystem. Blue whales closely tracked the long-term average phenology of the spring bloom. This study showed that both long-term memory and resource tracking play key roles in the long-distance migrations of blue whales and possibly other marine megafauna. Results of this study suggest that such species may struggle to respond to rapid deviations from historical mean environmental conditions (Abrahms et al. 2019). If blue whales can 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. Findings from Millien et al. (2025) suggest that humpback whales integrate information they gather from their immediate environment to predict the oceanic conditions at distant locations and adjust the timing of their migration, maximizing their interaction with their preys. The ability of humpback whales to fully succeed in tracking their preys may be compromised by rapidly changing environmental conditions.

Changing environmental conditions can influence marine mammal reproductive success and fitness by altering prey availability. Breeding in many marine mammal species may be timed to coincide with maximum abundance of suitable prey, either for the lactating mother or the calf at weaning, so that any changes in the environmental conditions which determine prey abundance may cause a mismatch in synchrony between predator and prey, either in time or location (Learmonth et al. 2006). Migratory species that travel long distances between feeding and breeding areas may be particularly vulnerable to mismatching. Pecl and Jackson (2008) predicted changing environmental conditions 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, and reduced prey availability has been suggested to explain lower rates of conception in female sperm whales (Whitehead 1997).

Briscoe et al. (2025) integrated three decades of satellite tracking data from juvenile North Pacific loggerhead sea turtles with observations of sea surface temperature (SST) and chlorophyll-a concentrations to examine higher trophic level response to changing environmental conditions within the eastern bounds of the North Pacific Transition Zone (NPTZ), defined as the region between 30°N-45° N, 180°W-140°W. Between 1997–2024, the NPTZ warmed by 1.6°C and experienced an approximately 19% decline in mean surface chlorophyll-a concentration, a proxy for reduced productivity (Briscoe et al. 2025). Over the same period, the average latitude of loggerhead sea turtle foraging habitat in the NPTZ shifted northwards by 450–600 km. In most years both the southern and northern range limits for the loggerhead turtle have shifted northward in tandem, indicating a habitat range shift rather than a range expansion (Briscoe et al. 2025).

5.1.4 Turtles - Nesting

Sea turtles have temperature-dependent sex determination and altered sex ratios have been observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009a). Many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patrício et al. 2017; Jensen et al. 2018). Within the action area for this opinion, female biased green sea turtle sex ratios have been reported at foraging locations in San Diego Bay, California (Allen et al. 2017). An analysis of hawksbill stranding data from Hawaii over three decades identified nearly five times the number of females compared to males (Brunson et al. 2022). The study excluded data from nesting females and if the vulnerabilities to becoming stranded is assumed to be similar between the sexes when at sea, then the data suggest the potential for a large female bias

in the overall population of hawksbills in Hawaiian waters. Genetic analyses and behavioral data suggest that populations with temperature-dependent sex determination may be unable to evolve rapidly enough to counteract the negative fitness consequences of rapid global temperature change (Hays 2008 as cited in Newson et al. 2009). A fundamental shift in population demographics may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2011; Azanza-Ricardo et al. 2017). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al. 2009).

5.1.5 Salmon and Steelhead

Studies examining the effects of long-term changes in environmental conditions to salmon and steelhead populations have identified a number of common mechanisms by which climate variation is likely to influence sustainability of steelhead populations (NMFS 2016g). Effects of changing environmental conditions on salmonids tend to be negative across multiple life-stages (Wade et al. 2013; Wainwright and Weitkamp 2013). Considering the action area for this opinion, we focus here on the effects of changing environmental conditions on salmonids in the marine environment. Northward range shifts are a response to changing environmental conditions expected in many marine fish species, including salmon (Cheung et al. 2015). Salmon and steelhead marine migration patterns could be affected by contraction of thermally suitable habitat.

Numerous researchers have reported that salmon and steelhead marine survival is highly variable over time and often correlated with large-scale climate indices (Petrosky and Schaller 2010; Stachura et al. 2013; Litzow et al. 2014; Sydeman et al. 2014). Many fish communities, including key salmon and steelhead prey and predators, experience changes in abundance and distribution during warm ocean periods (Pearcy 2002; Cheung et al. 2009). However, food chain dynamics in the open ocean are flexible and difficult to predict into the future, and in the case of steelhead poorly understood (Grimes 2007). To what extent a future warmer ocean will mimic historic conditions of warm-ocean, low-survival periods is not known. Current indications are that a warmer Pacific Ocean is generally less productive at mid latitudes, and hence likely to be less favorable for salmon and steelhead (NMFS 2016g). The full implications of ocean acidification on salmon are not known at this time (Council 2010). Olfaction and predator-avoidance behavior are negatively affected in some fish species, including pink salmon (Leduc et al. 2013; Ou et al. 2015). Pink salmon also showed reductions in growth and metabolic capacity under elevated carbon dioxide conditions (Ou et al. 2015). Some high-quality salmon prey (e.g., krill) might be negatively affected by ocean acidification, but there are several possible pathways

by which higher trophic levels might compensate for changes at a lower trophic level and impacts could conceivably be positive (Busch et al. 2013).

5.1.6 Sharks and Rays

Because habitat for many shark and ray species is comprised of open ocean environments occurring over broad geographic ranges, large-scale changes in environmental conditions that affect ocean temperatures, currents, and potentially food chain dynamics, may impact these species. Chin et al. (2010) conducted an integrated risk assessment to assess the vulnerability of several shark and ray species on the Great Barrier Reef to the effects of changing environmental conditions. Scalloped hammerheads were ranked as having a low overall vulnerability to changing environmental conditions, with low vulnerability to each of the assessed environmental factors (i.e., water and air temperature, ocean acidification, freshwater input, ocean circulation, sea level rise, severe weather, light, and ultraviolet radiation). Hazen et al. (2012) used spatial models of top predator species distributions with global climate models to project changes in North Pacific biodiversity hotspots and to identify species groups most at risk from changing environmental conditions. Sharks, as a group, showed the greatest risk of pelagic habitat loss, although none of the sharks modeled in this study are ESA-listed.

5.1.7 Invasives, Disease, Vulnerable Populations

The potential for invasive species to spread increases under the influence of climatic change. As water temperatures warm, native species ranges can shift poleward, opening ecological niches that could be occupied by more invasive and exotic species, introduced via ships ballast water or other sources (Ruiz et al. 1999; Philippart et al. 2011). Invasive species that are better adapted to warmer water temperatures can outcompete native species that are physiologically geared towards lower water temperatures (Lockwood and Somero 2011). Shifts in environmental conditions and host ranges can result in the spread of novel pathogens to new areas (Simmonds and Eliott. 2009). For example, research suggests increases in sea surface temperature can increases harmful algal blooms (Simmonds and Eliott. 2009). Moore et al. (2011) estimated that the impacts of a dinoflagellate establishment would likely intensify with changing environmental conditions, resulting in roughly 13 more days of potential bloom conditions per year by the end of the 21st century. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immune system functions and reproductive parameters in wildlife to the detriment of population health and viability.

5.1.8 Summary

Changing environmental conditions can 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 (MacLeod et al. 2005; Robinson et al. 2005; Kintisch 2006; Learmonth et al. 2006; McMahon and Hays 2006; Evans and Bjørge 2013; IPCC 2014; Lonhart et al. 2019; Sanford et al. 2019; Santos et al. 2024). Increasing

atmospheric temperatures have already contributed to documented changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Mantua et al. 1997; Karl 2009). Changing environmental conditions will likely have the most pronounced effects on vulnerable species whose populations are already in tenuous positions (Williams et al. 2008). As such, we expect the risk of extinction for many ESA-listed species to increase in the future due to changing environmental conditions.

Environmental changes are occurring within the HCTT action area and are expected to continue into the future. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of changing environmental conditions. Many of the ESA-listed species and habitats considered in this opinion have likely already been impacted by the threat of changing environmental conditions through the pathways described above.

5.2 Oceanic Temperature Regimes

The California Current Large Marine Ecosystem 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). Oceanographic conditions in the Pacific Ocean can change due to periodic shifts in patterns of atmospheric pressure systems caused by the Southern oscillation, which leads to El Niño and La Niña events, and the Pacific decadal oscillation. These climatic events can alter habitat conditions and prey distribution for ESA-listed species in the action area (Beamish 1993; Mantua et al. 1997; Hare and Mantua 2001; Benson and Trites 2002; Stabeno et al. 2004; Mundy 2005; Mundy and Cooney 2005).

During 2014–2016, a record-breaking marine heat wave occurred in the North Pacific, (including large portions of the action area) that resulted in an unprecedented multi-year warming event. 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 0-300 m. This warm water region was accompanied by a strong El Niño event from 2015-2016. The heatwave pushed some prey species, like anchovies, closer to shore (Santora et al. 2020). This subsequently shifted whale distribution and increased whale foraging in nearshore waters where there is more overlap with commercial and recreational fixed-gear fisheries, including that for Dungeness crab. At the same time, there was a change in fishing effort in central California due to a harmful algal bloom at the start of the Dungeness crab fishery season (Santora et al. 2020; Saez et al. 2021). The delay at the start of the Dungeness fishing season in the fall increased fishing effort in the spring, which coincided with the arrival of humpback whales (Saez et al. 2021). 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.

Feist et al. (2021) found that a delayed Dungeness crab fishery opening in California due to a harmful algal bloom in spring of 2016 led to about 5–7 times average levels of fishing activity, which was consistent with a high rate of large whale entanglement that year. These results support the hypotheses that habitat compression caused by a marine heatwave increased the overlap of whales with fishing activity, despite minimal changes in the fisheries themselves (Feist et al. 2021). Ingman et al. (2021) found that humpback and blue whales migrations towards feeding areas around the Farallon Islands has been occurring earlier in recent years and that the timing was significantly influenced by a mix of local oceanography, regional, and basin-scale climate variables. Earlier arrival time without concomitant earlier departure time results in longer periods when blue and humpback whales are at risk of entanglement in pot and trap fishery gear off central California (Ingman et al. 2021). More information on large whale entanglements in fishing gear can be found below in Section 5.10.2 Marine Mammal Fishery Interactions.

Based on a study of 1,819 cetacean strandings across 26 species in the Pacific Northwest from 2000-2019, Warlick et al. (2022) found that strandings are often correlated with certain environmental variables, including sea surface temperature, chlorophyll concentration, and the Pacific Decadal Oscillation, depending on the species. For humpback whale strandings, which have increased in recent years, the best model predicts peak strandings in summer and lower strandings in months after higher chlorophyll concentrations and lower temperatures (Warlick et al. 2022).

Gálvez et al. (2020) examined the effect of the 2013-2015 warm water anomaly in the northeast Pacific Ocean on weight gain in neonate Guadalupe fur seals off the coast of the Baja California Peninsula, Mexico. Lowest birth weights and slowest weight gains were observed in 2014 while low weights and lowest survival rate were observed in 2015, all of which were likely due to the warm water anomaly (Gálvez et al. 2020). The authors conclude that the evident sensitivity of Guadalupe fur seal neonates to regional warming conditions highlights their vulnerability under changing environmental conditions, which could impede this species' recovery (Gálvez et al. 2020). Alterations in maternal care and prey availability due to the Northeast Pacific Heatwave are directly linked to higher rates of emaciation as a leading cause of Guadalupe fur seal pup mortality (Gálvez et al. 2023). Data collected from scat and lanugo samples from 2013-2016 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).

Studies suggest that periods of what was once considered "anomalous" warm water conditions are occurring more frequently. The current rate of increase in oceanic and atmospheric heat content has increased the frequency and duration of marine heatwaves, or "prolonged discrete anomalously warm water events" (Hobday et al. 2016); a trend that is expected to continue (Oliver et al. 2018) in conjunction with changes in the strength, direction, and variability of

major ocean currents (Hoegh-Guldberg and Bruno 2010). On a global scale, the occurrence probabilities of the duration, intensity, and cumulative intensity of most documented, large, and impactful marine heatwaves have increased more than 20-fold as a result of changing environmental conditions (Laufkötter et al. 2020).

The Pacific Decadal Oscillation is the leading mode of variability in the North Pacific, capable of altering sea surface temperature, surface winds, and sea level pressure over longer periods than either El Niño or La Niña/Southern Oscillation events (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 (Hare et al. 1999; Childers et al. 2005). 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% 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 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.

ZoBell et al. (2025) studied acoustic recordings over 15 years from six sites within the Southern California Bight. They found that biological sound levels associated with fin and blue whale calls exhibit distinct seasonal and interannual variations, and that whale vocalization patterns fluctuate in response to major climatic events, such as the marine heatwave, and shifts in ocean-atmosphere climate variability, such as the Pacific Decadal Oscillation. Baleen whale sound levels before, during, and after a marine heatwave were assessed; sound levels decreased in southern sites and increased in northern sites adjacent to the California Current, underscoring the potential for range shifts and habitat compression during warm years for these species (ZoBell et al. 2025).

5.3 Sound

The ESA-listed species that occur in the action area are regularly exposed to multiple sources of anthropogenic sounds. Anthropogenic sound is generated by commercial and recreational vessels, aircraft, sonar, ocean research activities, dredging, construction, offshore mineral exploration, military testing and training activities, and other human activities. These activities occur within the action area to varying degrees throughout the year. Sources of anthropogenic noise are becoming both more pervasive and more powerful, increasing both oceanic background

sound levels and shorter term peak intensity levels (Hildebrand 2004), which can impact ESA-listed species.

Sounds are often considered to fall into one of two general types, impulsive and non-impulsive, which differ in the potential to cause physical effects to animals (See Southall et al. (2007) for in-depth discussion). Impulsive sound sources produce brief, broadband (wide range of frequencies) signals that are atonal transients and occur as isolated events or repeated in some succession. They are characterized by a relatively rapid rise from ambient pressure to a maximal pressure value followed by a rapid decay period, and generally have an increased capacity to induce physical injury. Non-impulsive sounds can be tonal, narrowband, or broadband, brief or prolonged, and may be either continuous or non-continuous. Some can be transient signals of short duration but without the essential properties of pulses (e.g., rapid rise time). The duration of non-impulsive sounds, as received at a distance, can be greatly extended in a highly reverberant environment.

Anthropogenic sound within the marine environment is recognized as a potential stressor that can harm marine animals and significantly interfere with their normal activities (NRC 2005). The species considered in this opinion may be impacted by anthropogenic sound in various ways. Damage to marine mammal hearing and mass stranding events due to high-intensity sound exposure have been documented (Hildebrand 2004). Anthropogenic sounds may also produce a behavioral response including, but not limited to, changes in habitat preference to avoid areas of higher sound levels, changes in diving behavior, or (for cetaceans) changes in vocalization (MMC 2007). Many researchers have described behavioral responses of marine mammals to the sounds produced by boats and vessels, as well as other sound sources such as rotary-wing and fixed-wing aircraft, and dredging and construction. Most observations have been limited to short-term behavioral responses, which include temporary cessation of feeding, resting, or social interactions. Habitat abandonment can lead to more long-term effects, which may have implications at the population level. Interference, or masking, occurs when a sound is a similar frequency and similar to or louder than the sound an animal is trying to hear (Francis 2013). Masking can interfere with an individual's ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environmental cues (Richardson et al. 1995c). Masking can reduce the range of communication, particularly long-range communication, such as that for blue and fin whales. Recent scientific evidence suggests that marine mammals, including blue and fin whales, compensate for masking by changing the frequency, source level, redundancy, or timing of their signals, but the long-term implications of these adjustments are currently unknown (Parks 2003; Mcdonald et al. 2006; Parks et al. 2009).

There are limited studies on the hearing abilities of sea turtles, their use of sound, and their vulnerability to sound exposure. Some evidence suggests that sea turtles are able to detect (Ridgway et al. 1969a; Bartol et al. 1999; Bartol and Ketten 2006; Martin et al. 2012) and behaviorally respond to acoustic stimuli (O'Hara and Wilcox 1990; Moein et al. 1995; McCauley et al. 2000; DeRuiter and Doukara 2012). Sea turtles may use sound for navigation, locating prey, avoiding predators, and general environmental awareness (Dow Piniak et al. 2012).

Fishes may use sound for various purposes, including: communication, mating behavior, the detection of prey and predators, orientation and migration, and habitat selection. Similar to marine mammals and sea turtles, if an underwater sound is sufficiently intense, a fish could suffer some level of injury and at lower levels may result in temporary hearing impairment, physiological stress, changes in behavior or the masking of biologically important sounds. Although there are many information gaps about how a range of fish species respond to different sound sources under various conditions, anything that interferes with the ability of a fish to detect and respond to biologically relevant sounds could decrease their fitness and therefore, survival.

Despite the potential impacts on individual ESA-listed marine mammals and sea turtles, information is not currently available to determine the potential population level effects of cumulative anthropogenic sound sources in the marine environment (MMC 2007). For example, we currently lack empirical data on how sound impacts growth, survival, reproduction, and vital rates, nor do we understand the relative influence of such effects on the population being considered. As a result, the consequences of anthropogenic sound on ESA-listed marine mammals and sea turtles at the population or species scale remain uncertain.

This section is divided into subsections addressing the potential stressors from the following major anthropogenic sound sources: vessels and commercial shipping; seismic surveys; military activities; active sonar; pile driving, construction and offshore industry operations. A more detailed discussion of the effects on these sound sources on ESA-listed species can be found in the effects analysis Sections 6.1.1, 6.2.1, and 6.3.1 below.

5.3.1 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 m for fast-moving (greater than 20 knots) supertankers to 140 dB re: μPa^2-s at 1 m for smaller vessels (NRC 2003). Sound emitted from large vessels is the principal source of low frequency noise in the ocean and marine mammals are known to react or to be affected by that noise (Richardson et al. 1995c; Foote et al. 2004; Hildebrand 2005; Hatch and Wright 2007; Holt et al. 2008; Melcon et al. 2012; Anderwald et al. 2013; Kerosky et al. 2013; Erbe et al. 2014; Guerra et al. 2014; May-Collado and Quinones-Lebron 2014; Williams et al. 2014). Although large vessels emit predominantly low frequency sound, studies report broadband sound from large cargo vessels above two kHz, 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 (NRC 2003; Hildebrand 2009b; Mckenna et al. 2012). Shipping constitutes a major source of low-frequency

(five to 500 Hz) sound in the ocean (Hildebrand 2004), particularly in the Northern Hemisphere where the majority of vessel traffic occurs. Commercial shipping's contribution to ambient noise in the ocean increased by as much as 12 dB between approximately the 1960s and 2005 (McDonald et al. 2008; Hildebrand 2009a). Frisk (2012) reported that between 1950 and 2007 ocean noise in the 25 to 50 Hz frequency range has increased 3.3 dB/decade. 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 location data records, spanning the years 2014-2018, show that the vast majority of vessel traffic within the Hawaii and California portions of the action area are from non-military vessels; just under 96% of traffic by the number of vessels transits and 97% of traffic by the estimated amount of time spent in the training and testing areas (Starcovic and Mintz 2021). The three most common types of commercial vessels in the action area are cargo, bulk carrier, and tanker ships (Figure 58).

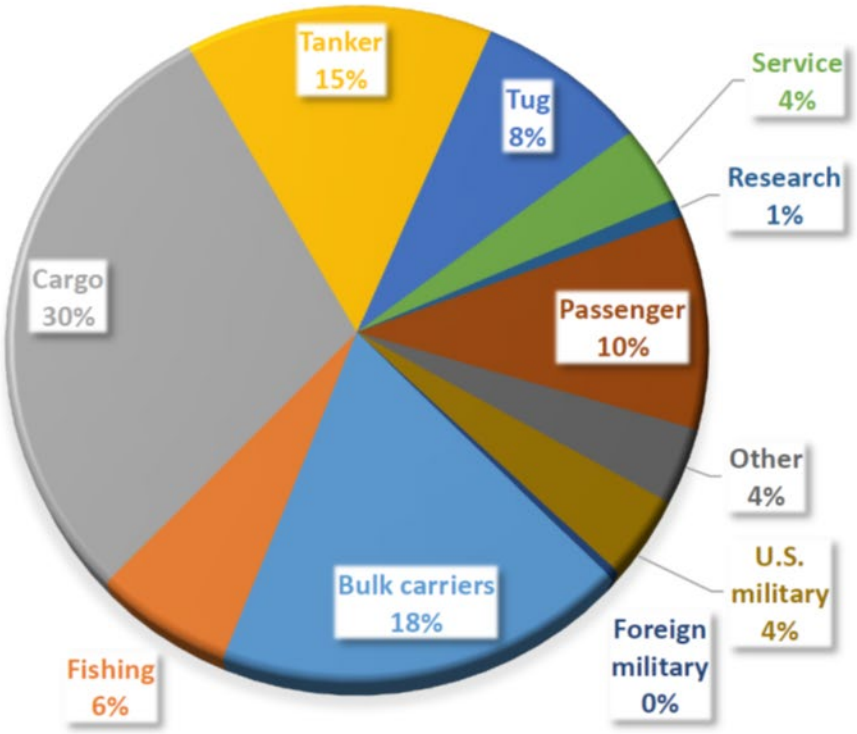


Figure 58. Percent of Vessel Traffic Intensity (i.e., Number of Vessel Transits or Tracks) in HCTT Action Area by Vessel Type, 2014-2018 (Starcovic and Mintz 2021).

The relative geographic distribution of nonmilitary vessel traffic in HCTT is shown in Figure 59. The heaviest vessel traffic is along the coast of California and near the main Hawaiian Islands. Marine species dependent upon coastal and estuarine nearshore environments around the Main

Hawaiian Islands or off California are more likely to experience the cumulative effects of vessel sound. The ports of Los Angeles/Long Beach are one of the biggest and busiest shipping complexes in the nation and the vessel Traffic Separation Scheme's lanes run through the PMSR, causing commercial vessel noise to be the main source of underwater anthropogenic noise in the area (Rice et al. 2018; Wiggins et al. 2018). Redfern et al. (2017) found that shipping channels leading to and from the ports of Los Angeles and Long Beach may have degraded the habitat for blue, fin, and humpback whales due to the loss of communication space where important habitat for these species overlaps with elevated noise from commercial vessel traffic. Blue whales off California altered call levels and rates in association with changes in local vessel traffic (McKenna 2011). In Northern California, the Port of Oakland is among the largest and most active shipping ports in the state.[14] Studies have shown that areas with high levels of shipping traffic, daily average sound levels in the 63 and 125 Hz one-third octave bands were found to be near or higher than 100 dB re: 1 µPa (Haver et al. 2021). Daily average sound levels were between approximately 10–20 dB higher relative to areas with lower levels of shipping activity (Haver et al. 2021). Shipping lanes running adjacent to the coast also run adjacent to or through portions of National Marine Sanctuaries and some of the designated BIAs for cetaceans (Calambokidis et al. 2015; Moore et al. 2018).

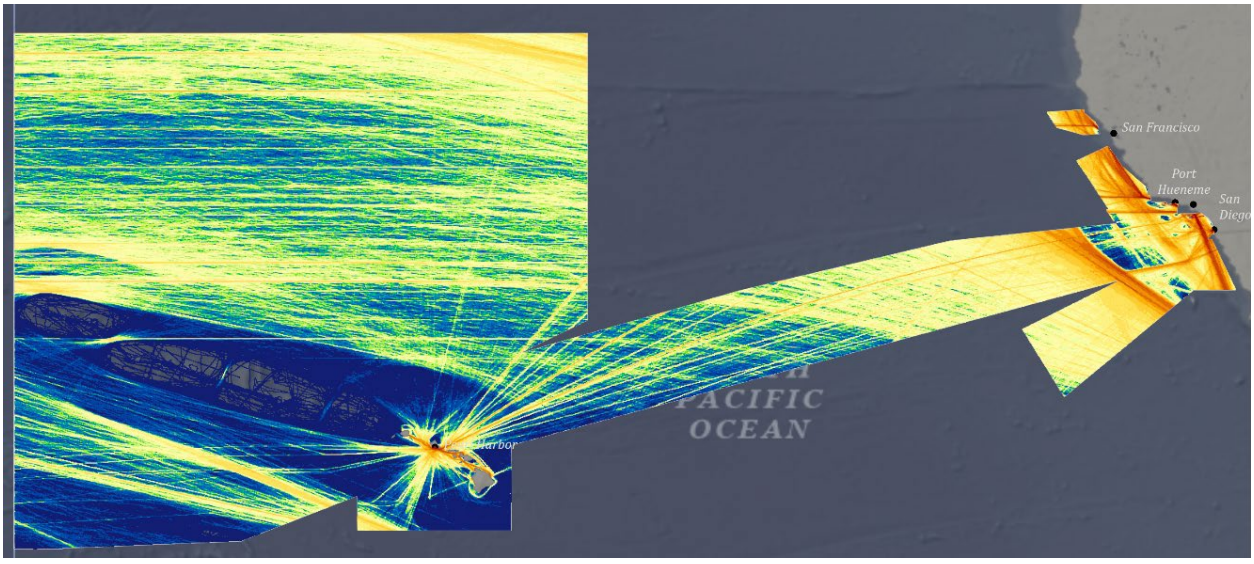


Figure 59. Relative Distribution of Nonmilitary Vessel Traffic Intensity (i.e., Number of Vessel Transits or Tracks) in HCTT, 2016-2023 (Navy 2025k)

5.3.2 Acoustic Geophysical Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor and return signals that are interpreted for geologic data.

[14] https://pacmar.com/article/study-california-ports-a-major-economic-generator-for-u-s/ Accessed 2/10/25.

Numerous seismic surveys have been conducted off the coast of California and within the HRC over the past several decades; see Figure 60 for example of seismic survey track lines. Seismic survey activity is often associated with oil and gas development in other regions, like the Gulf of America that has over 2,000 active leases, but there are only 30 active leases in the Pacific OCS Region[15] and seismic surveys are primarily for scientific research. No new oil and gas leasing has occurred in the Pacific Region since 1984 and very little geophysical data is collected in the Pacific OCS region, and almost none since 2000.[16] Geological research is conducted off California to identify possible seafloor or shallow-depth geologic hazards and to locate potential archaeological resources and benthic habitats that should be avoided.

There are two major categories of acoustic geophysical 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 towed behind a vessel along with receiver arrays consist of hydrophones used to detect underwater sound waves encased in plastic tubing called streamers. When an airgun array fires, an acoustic energy pulse is emitted and reflected or refracted back from the seafloor. These reflected/refracted acoustic signals create pressure fluctuations, which are detected and recorded by the streamers. Seismic airguns generate intense low-frequency sound pressure waves capable of penetrating the seafloor and are fired repetitively at intervals of 10 to 20 seconds for extended periods (NRC 2003). Most of the energy from airguns is directed vertically downward, but significant sound emission also extends horizontally. Peak SPLs from airguns usually reach 235 to 240 decibels at dominant frequencies of five to 300 Hz (NRC 2003). 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 more often use other sound sources: sub-bottom profilers (at 2.5-7 kHz), echosounders (single-beam at 12-240 kHz; multibeam at 50-400 kHz), boomers (at 300-3,000 Hz), sparkers (at 50-4,000 Hz), compressed high intensity radar pulse sub-bottom profiler (at 4-24 kHz), pingers (at 2 kHz), and side-scan sonars (16-1,500 kHz). These sound sources are powered mechanically or electromagnetically.

[15] https://www.boem.gov/oil-gas-energy/leasing/lease-stats-2-1-25 Accessed 2-18-25.
[16] https://www.boem.gov/oil-gas-energy/resource-evaluation/red-bluebook2023 Accessed 2-18-25.

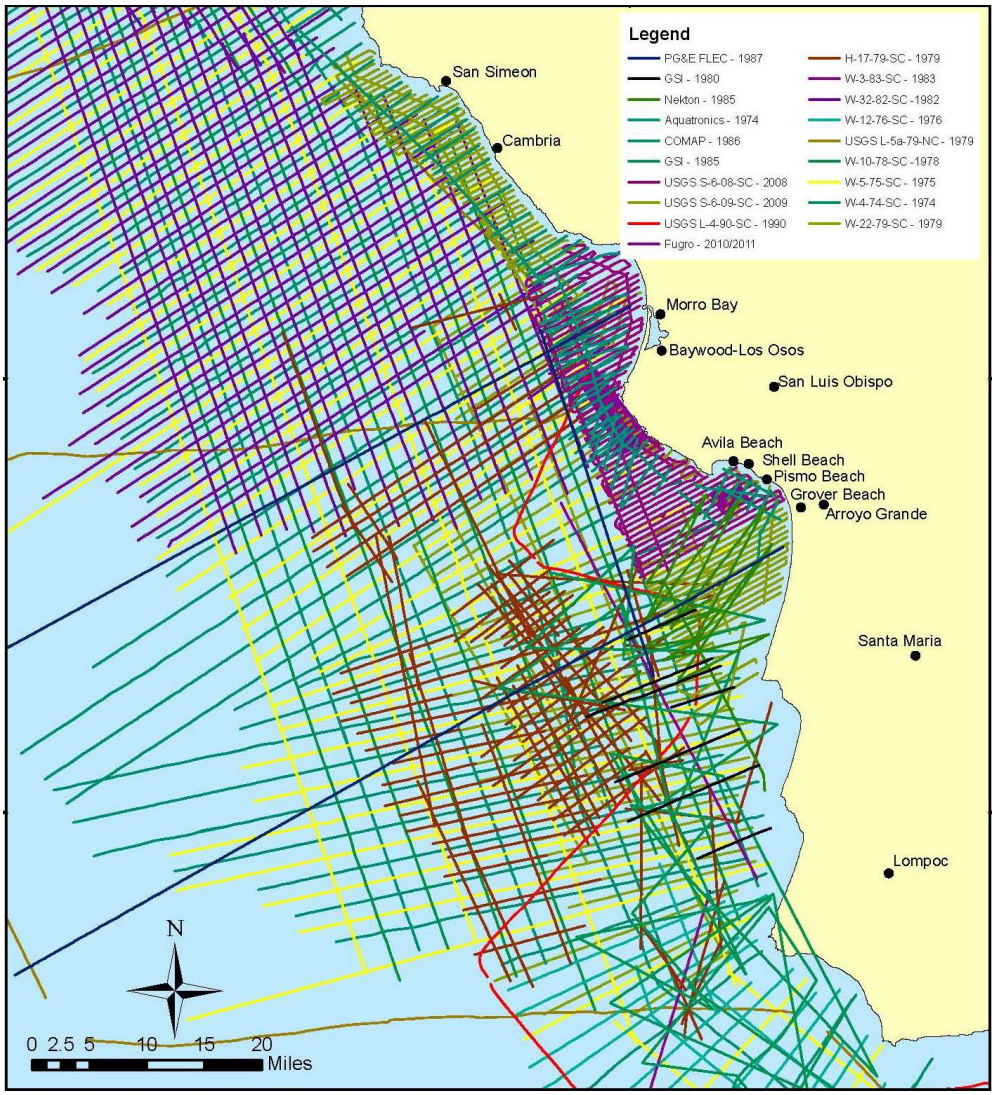


Figure 60. Seismic Survey Track Lines within the California Portion of the Action Area (NMFS 2015a)

Effects analysis of high-resolution geophysical surveys (HRG) often determine the activities may affect but not likely to adversely affect (NLAA) NMFS listed species. Exposure to strong impulsive sounds from airgun arrays can result in auditory damage, such as temporary or permanent threshold shifts (TTS or PTS) in hearing sensitivity by decreasing the range of sound an animal can detect within its normal hearing ranges (reviewed in Finneran 2015). 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 survey sound fields. Whales may exhibit avoidance, alter foraging behavior and alter their vocalizations. These responses are expected to be temporary with behavior returning 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. 2007; Zimmer and Tyack 2007; Tal et al. 2015), 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 surveys. Avoidance behavior and physiological responses from airgun exposure may affect the natural behaviors of sea turtles (McCauley et al. 2000). Measurements of a seismic vessel operating 3-dimensional airgun arrays in 100 to 120 m water depth corresponded to behavioral changes at around 2 km and avoidance around one kilometer.

MMPA authorizations and ESA consultations specify the conditions under which researchers can operate seismic sound sources, such as airguns, including mitigation measures to minimize adverse effects to protected species. One such mitigation measure is the delay or suspension of seismic activities whenever marine mammals or sea turtles are observed within their designated clearance or shutdown zones. The zones will vary in size based on species hearing sensitivity and sound source characteristics.

The National Science Foundation funds marine geophysical research[17] and awards support seismic surveys around the world using airguns. Funding cycles focus on particular regions due to the logistical challenge of global research vessel transit and shore side support. NSF supported seismic surveys have not occurred in the US Pacific since 2022 which was focused off Oregon and Washington. They have not been conducted near Hawaii since 2018.

5.3.3 Military Training and Testing Activities

The U.S. Pacific Fleet is the world's largest naval fleet with an estimated Area of Responsibility of 100 million nm^2 including the entire action area. Additionally, the U.S. Marine Corps Forces Pacific Area of Responsibility includes the entire action area. The Navy has conducted training and testing activities and other military readiness activities in the HRC and SOCAL RC 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 assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development,

[17] https://www.nsf.gov/funding/opportunities/mgg-marine-geology-geophysics Accessed 2-18-25.

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.

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 AINJ to marine mammals and sea turtles. 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 HSTT action area. Conservation measures include employing visual observers and implementing mitigation zones when training and testing using active sonar or explosives. USCG's District 14 provides services to a broad expanse of the Pacific including the action area. USCG operations are ongoing and are expected to continue into the future and are similar in nature and scope to those described above for the Navy and Marine Corps.

There are missile launches that occur in the action area, such as at the PMRF in Hawaii and the PMSR in California, that can create noise in air that could result in disturbance to ESA-listed pinnipeds hauled out of the water, e.g., Hawaiian monk seal and Guadalupe fur seal. Similarly, there are rocket launches that occur at the Vandenberg Space Force Base, California, that support various Federal programs but also include launches by commercial space operations.

The Air Force has conducted training and testing activities in the action area in the past, and these activities are ongoing and are expected to continue into the future. Air Force activities generally involve the firing or dropping of munitions (e.g., bombs, missiles, rockets, and gunnery rounds) from aircraft towards targets located on the surface, though Air Force training exercises may also involve boats. These activities impact ESA-listed species through physical disturbance, boat strikes, debris, ingestion, and effects from sound and pressure produced by detonations. Air Force training and testing activities constitute a Federal action and take of ESA-listed species resulting from these Air Force activities have previously undergone separate section 7 consultations.

The effects of military activities within the action area on ESA-listed species are described in more detail below (See Section 5.12 for Ongoing Military Training and Testing Activities in the Action Area).

5.3.4 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 kHz and less, mid-frequency for one to 10 kHz; high-frequency for 10 to 100 kHz; and very high-frequency for greater than 100 kHz (Hildebrand 2004). Low frequency systems are designed for long-range detection (Popper et al. 2014b). The effective source level of an low-frequency active array, when viewed in the horizontal direction, can be 235 dB re: 1µPa at 1 m or higher (Hildebrand 2004). Signal transmissions are emitted in patterned sequences that may last for days or weeks.

Military sonar systems encompass all three frequency divisions and includes sources with wider beam widths and higher source levels compared with commercial sonar systems. Activities are episodic and can last from hours, days to weeks, and over a month (NRC 2003). An example of a low-frequency active sonar system is the U.S. Navy Surveillance Underwater Towed Array Sensor System (SURTASS), discussed in more detail below (See Section 5.12 Ongoing U.S. Navy Training and Testing Activities in the Action Area). 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.

High-frequency sonars are used in the PMSR portion of the action area by the Coast Guard and active mid-frequency sonar has been used by the Navy in waters off Southern California, overlapping a portion of PMSR. For HSTT Phase III, the Navy proposed to use nearly 50,000 hours of hull-mounted mid-frequency active sonar throughout the HSTT action area over the seven year period from late 2018 through late 2025, although the actual number of hours that will be used is typically much lower than proposed, based on reports of previous missions (NMFS 2018b).

Compared with military sonar systems, commercial sonar systems use higher frequency signals, have lower source levels, narrower beam patterns that are downward directed, shorter pulse lengths, and are typically operated for minutes to days (NRC 2003). Usage is widespread across locations and sectors including recreation, fishing, shipping, and research. Examples of common commercial sonar systems include depth finders and fish finding sonar (15 to 200 kHz) (Širović et al. 2020), both of which focus sound in a downward beam. Depth sounders are operated primarily in nearshore and shallow environments, however, fish finders are operated in both deep and shallow areas. Sources such as depth finders, multi-beam echosounders, and side-scan sonar can be used for sea floor mapping, similar to HRG surveys, and are also utilized for military applications.

5.3.5 Pile Driving and Construction Sound

Industrial activities and construction both in the ocean and along the shoreline can contribute to underwater noise. Pile driving is commonly used for the construction of foundations for a large number of structures including bridges, buildings, retaining walls, harbor facilities, offshore wind turbines, and offshore structures for the oil and gas industry. Pile driving during construction activities is of particular concern because it generates noise with a very high source level. During pile installation, noise is produced when the energy from construction equipment is transferred to the pile and released as pressure waves into the surrounding water and sediments. The impulsive sounds generated by impact pile driving are characterized by a relatively rapid rise time to a maximal pressure value followed by a decay period that may include a period of diminishing, oscillating maximal and minimal pressures (Illingworth and Rodkin 2001; Illingworth and Rodkin Inc. 2007; Reyff 2012). The amount of noise produced by pile driving depends on a variety of factors, including the type and size of the impact hammer, size of the pile, the properties of the sea floor, and the depth of the water. The predominant energy in pile impact impulses is at frequencies below approximately 2000 Hz, with most occurring below 1000 Hz (Laughlin 2006; Reyff 2008; Reyff 2012). Pressure levels from 190-220 dB re: 1 µPa were reported for piles of different sizes in a number of studies (NMFS 2006). Impact pile driving occurs over small spatial and temporal scales and produces high-intensity, low-frequency, impulsive sounds with high peak pressures that can be detected by mammals, sea turtles and other marine species (Dow Piniak et al. 2012). Injury to sea turtles and marine mammals is caused by pressure wave damage to hair cells, ear canals, or ear drums as these structures compress and expand with passage of the wave. Vibratory pile driving produces a continuous sound with peak pressures lower than those observed in impulses generated by impact pile driving (Popper et al. 2014b).

5.3.6 Offshore Industry Operations

The operation of offshore oil and gas extraction platforms produces nearly continuous noise primarily from 20 to 1,000 Hz (Greene and Richardson 1988) and includes ancillary noise from support vessels and machinery. Drill ships are generally the loudest with most broadband energy between 10 Hz and 10 kHz (Richardson et al. 1995c). Drilling machinery noise is transmitted through the ship hull and from the dynamic positioning thrusters during operations. The active leases in the US Pacific region are near the coast of Southern California, an area that is already subject to commercial vessel traffic noise (see Section 5.3.1).

Offshore wind leases have been awarded off the coast of California. Installation and operation of wind turbines, including power production related infrastructure, is expected to add more sound to the marine environment. The deep waters off California will require floating wind turbine platforms. It could be a few years before offshore wind power developers are ready to build out their lease areas on the Pacific Outer Continental Shelf, therefore, details to assess impacts from the floating structures and installation techniques are not available yet.

5.3.7 Acoustic Deterrents

Noise-generating acoustic deterrents are used within fisheries to minimize interactions with marine mammals that prey on the catch or stock, damage fishing gear or become fatally entangled in nets as bycatch. While such deterrents are intended to reduce fishery interactions with marine mammals, they can also pose a potential threat in terms of hearing damage and behavioral effects like displacement or disturbance which may result in reduced foraging effort and success (Simonis et al. 2020). Seal bombs are one type of marine mammal acoustic deterrent primarily used in fisheries along the North American west coast. Several California fisheries use seal bombs as deterrents, including purse seine and set gillnet fisheries (Baumann-Pickering et al. 2013; Bland 2017; Wiggins et al. 2018; Wiggins et al. 2019).

Krumpel et al. (2021) evaluated long-term passive acoustic monitoring recordings to study the use of seal bombs in Southern California fisheries from 2005-2016. This study revealed high numbers of these small-charge underwater explosions at many nearshore sites (up to 2,800/day), the vast majority of which occurred at night. Underwater explosions related to commercial fishing activities represent a relevant anthropogenic noise source off Southern California because they were detected on more than one-third of all recording days and were also detected in the vicinity, or inside of, the Channel Islands and Monterey Bay National Marine Sanctuaries. Received sound exposure levels of up to 189 dB re: 1 mPa2-s indicate the potential for negative effects on marine life, especially in combination with the persistence of recurring explosions during periods of peak occurrence (Krumpel et al. 2021). The maximum SELs (unweighted, noncumulative) of 164–189 dB re: 1 mPa2-s per site, recorded in this study, partly exceed AINJ and TTS thresholds for cetaceans. Seal bomb use declined over the years of the study, which could be an influence of reduced availability of market squid due warm water events off California (Krumpel et al. 2021).

5.4 Dredging

Nearshore and offshore coastal areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Hydraulic dredging can directly harm large marine animals (e.g., sea turtles) by lethally entraining them through the dredge drag-arms and impeller pumps. Large animals that are entrained in hydraulic dredges rarely survive the encounter. Hopper dredges, in particular, are capable of moving relatively quickly compared to turtles and fish which can be overtaken and entrained by the suction draghead of the advancing dredge.

Dredging can also indirectly affect marine species through habitat modification, changes in prey availability, and water quality degradation, including changes in dissolved oxygen and salinity gradients (Jenkins et al. 1993; Secor and Niklitschek 2001; Campbell and Goodman 2004). Dredging operations also emit sounds at levels that could potentially disturb individuals of many marine taxa. Depending on the type of dredge, peak SPLs from 100 to 140 dB re: 1 µPa were reported in one study (Clarke et al. 2003). As with pile driving, most of the sound energy associated with dredging is in the low-frequency range, less than 1000 Hz (Clarke et al. 2003).

Based on a literature review of the impacts of dredging activities on marine mammals, Todd et al. (2014) found that dredging is unlikely to cause physiological damage to marine mammal auditory systems, but is more likely to lead to masking and behavioral disturbances, and baleen whales could be more at risk than other taxa.

Dredging projects typically occur in harbors, ports and nearshore coastal areas. Dredging operations can impact sea turtles directly and some ESA-listed fishes, especially if they rely on estuarine environments, like salmonids or sturgeon, could potentially be impacted directly or their habitat.

5.5 Pollution

Several different types of anthropogenic pollution resulting from past, present and ongoing human activities adversely affect ESA-listed species and habitats within the action area. For this opinion, we focus on three primary categories of marine and estuarine pollutants: contaminants and pesticides; nutrient loading and algal blooms; and marine debris. This section provides a general discussion of the three major pollutant categories above, including the stressor pathways and anticipated effects on ESA-protected resources, with an emphasis on geographic areas, habitats or species within the action area that are particularly susceptible to these threats.

5.5.1 Contaminants and Pesticides

Coastal habitats are often in close proximity to major sources of pollutants and contaminants, including pathogens, which make their way into the marine environment from land-based industrial, domestic and agricultural sources. Sources include wastewater treatment plants, septic systems, industrial facilities, agriculture, animal feeding operations, and improper refuse disposal. Agricultural discharges, as well as discharges from large urban centers, contribute contaminants like coliform bacteria and are major sources of nutrient enrichment (see Section 5.5.3 Nutrient Loading and Algal Blooms) to coastal watersheds. Urban runoff is the largest source of contaminants near the coast of California and when storm events occur, runoff plumes can become large oceanographic features that extend for many miles (Center for Ocean Solutions 2009). Urbanization has resulted in increased contaminants being deposited in Contaminants can be carried long distances from terrestrial or nearshore sources and ultimately accumulate in offshore pelagic environments (USCOP 2004). Global oceanic circulation patterns result in a considerable amount of pollutants that are scattered throughout the open ocean and accumulating in gyres and other places due to circulation patterns (Crain et al. 2009).

Chemical contaminants, particularly those that are persistent in the environment, are a particular concern for marine animals that often occupy high trophic positions. Persistent organic pollutants, which include legacy pesticides (e.g., dichlorodiphenyltrichloroethane [DDT], chlordane), legacy industrial-use chemicals (e.g., PCBs), and emerging contaminants of concern (e.g., polybrominated diphenyl ethers and perfluorinated compounds), accumulate in fatty tissues of marine organisms and are magnified through the food web leading high exposure levels in upper trophic predators (National Academies of Sciences and Medicine 2016). Ocean contamination resulting from chemical pollutants is a concern for cetacean conservation and has

been the subject of numerous studies (Krahn et al. 2007; Fair et al. 2010; Moon et al. 2010; Ocean Alliance 2010; Desforges et al. 2016). High concentrations of PCBs and DDT have been reported in tissues of marine mammals in most parts of the world, particularly in coastal regions adjacent to heavy coastal development and/or industry. These legacy persistent organic pollutants have been linked to a number of adverse health effects including endocrine disruption, reproductive impairment or developmental effects, and immune dysfunction or disease susceptibility (National Academies of Sciences and Medicine 2016). Polybrominated diphenyl ethers commonly used as flame retardants, are another class of persistent organic pollutants that have spread globally in the environment and have also been reported in a broad array of marine mammal species (National Academies of Sciences and Medicine 2016).

Savery et al. (2014) documented detectable lead concentration in 93% of 337 blubber biopsies from sperm whales sampled throughout the world. Ylitalo et al. (2008) analyzed blubber and blood samples for organochlorines from 158 Hawaiian monk seals at four of their six primary breeding colonies in the Northwestern Hawaiian Islands. They found that the health and fitness of Hawaiian monk seals from three of the four subpopulations may be at risk from elevated contaminant levels. Lopez et al. (2012) examined concentrations of a large suite of persistent organic pollutants in blubber and serum of juvenile and adult monk seals from the Main Hawaiian Islands. Adult females had the lowest blubber levels of most persistent organic pollutants, whereas adult males had the highest levels. Contaminant levels from the Main Hawaiian Islands were at similar or lower levels than those from remote Northwestern Hawaiian Island populations. In an analysis of cetacean blubber samples obtained from animals stranded in Hawaii between 1997 and 2011, higher levels of persistent organic pollutants were found in killer whale and false killer whale, as opposed to baleen whales, which had lower levels (Bachman et al. 2014). Kratofil et al. (2020) measured high PCB concentrations in 29 of 41 individuals sampled from the Main Hawaiian Islands Insular population at levels beyond those that would cause adverse health impacts (Kannan et al. 2000).

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, (Godley et al. 1999; Fujihara et al. 2003; Storelli et al. 2008; Barbieri 2009; García-Fernández et al. 2009). Cadmium has been found in leatherbacks at the highest concentration compared to any other marine vertebrate (Gordon et al. 1998). Newly emerged hatchlings have higher concentrations than are present when eggs are laid, suggesting that metals may be accumulated during incubation from surrounding sands (Sahoo et al. 1996). Arsenic has been found to be very high in green turtle eggs (Van de Merwe et al. 2009). Sea turtle tissues have been found to contain organochlorines, including chlorobiphenyl, chlordane, lindane, endrin, endosulfan, dieldrin, perfluorooctane sulfonate, perfluorooctanoic acid, DDT, and PCB (Gardner et al. 2003; Keller et al. 2005; Alava et al. 2006; Storelli et al. 2007; Oros et al. 2009). PCB concentrations are reportedly equivalent to those in some marine mammals (Davenport et al. 1990; Oros et al. 2009). Levels of PCBs found in green sea turtle eggs have exceeded recommended levels for human consumption (Van de Merwe et al. 2009).

Barraza et al. (2020) assessed persistent organic pollutants (POPs) in green turtle aggregations in the Seal Beach National Wildlife Refuge and San Diego Bay. Turtles from San Diego Bay had significantly ($p < 0.001$) higher total PCBs than Seal Beach National Wildlife Refuge turtles. Significant variation was found in POP levels through time and space, even among adjacent foraging locations.

Several studies have reported correlations between organochlorine concentration level and indicators of sea turtle health or fitness. Organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2006; Oros et al. 2009). Accumulation of these contaminants can also lead to deficiencies in endocrine, developmental and reproductive health (Storelli et al. 2007). Balazs (1991) suggested that environmental contaminants are a possible factor contributing to the development of the viral disease fibropapillomatosis in sea turtles by reducing immune function. Day et al. (2007) investigated mercury toxicity in loggerhead sea turtles by examining trends between blood mercury concentrations and various health parameters. They concluded that subtle negative impacts of mercury on sea turtle immune function are possible at concentrations observed in the wild. Keller et al. (2004) investigated the possible health effects of organochlorine contaminants, such as PCBs and pesticides on loggerhead sea turtles. Although concentrations were relatively low compared with other species, they found significant correlations between organochlorine contaminants levels and health indicators for a wide variety of biologic functions, including immunity and homeostasis of proteins, carbohydrates, and ions.

Toxic perfluorinated alkyl substances (PFASs) were detected in North Pacific sea turtles sampled from Hawaii, Palmyra Atoll, and the Northern Marianas Islands (Wood et al. 2021). PFAS levels in sea turtles were correlated with human population and land use. A risk assessment of these contaminants showed concern for immunosuppression in Kailua green turtles and toxicity for hawksbill development as maternal offloading of PFASs was detected in unhatched hawksbill eggs from Hawaii (Wood et al. 2021).

The chemical components of pesticides used on land flow as runoff into the marine environment and can bioaccumulate in the bodies of marine mammals, which can then be transferred to their young through mother's milk (Fair et al. 2010). There is growing evidence that the presence of chemical contaminants in their tissues puts marine mammals at greater risk for adverse health effects and potential impact on their reproductive success (Krahn et al. 2007; Fair et al. 2010; Godard-Codding et al. 2011).

Exposure to chemical pollutants may act in an additive or synergistic manner with other stressors resulting in significant population level consequences (Desforges et al. 2016). Despite the vast evidence indicating that marine animals are exposed to anthropogenic, as well as natural, chemicals capable of producing significant toxic effects, only a few studies have actually examined the impacts on population survival or reproductive rates. Such observational assessments are inherently challenging due to the difficulty in controlling for confounding or interacting variables, as well as the sublethal but chronic nature of chemical contaminant effects,

and the difficulty of observing mortality or reproductive endpoints, particularly in long-lived species such as cetaceans and sea turtles (National Academies of Sciences and Medicine 2016).

Many pollutants in the environment, such as brevotoxins, heavy metals, and PCBs, have the ability to bioaccumulate in fish species. A number of studies have shown that because of the higher trophic level position and longevity of some sharks and rays, these pollutants tend to biomagnify in liver, gill, and muscle tissues (Young et al. 2017). The large size and vast lipid stores in the elasmobranch liver provide the capacity for a substantial sequestration of lipophilic contaminants. Overall, sharks and rays are likely exposed to a number of pollutants and contaminants in their habitat that have the potential to cause negative physiological impacts to these species, although the effects of these pollutants and potential risk to the viability of the species remain unknown. Gelsleichter et al. (2020) reported toxic, non-essential metal mercury concentrations in oceanic whitetip shark were among the highest ever reported among four other pelagic shark species and correlated significantly with shark length. The authors concluded that mercury poses health risks to oceanic whitetip sharks and can include neurobehavioral effects and reduction of reproductive fitness, with the latter impacting the ability of the population to recover.

PAHs are another group of organic compounds that can result in adverse effects on marine species. Anthropogenic sources of PAHs include crude oil, fumes, vehicle exhaust, coal, organic solvents, and wildfires. Exposure may be continual, associated with run-off from impervious cover in developed coastal regions, or natural seeps that produce low-level but steady exposure. Acute events such as oil spills (see subsection that follows) may produce pulses of more significant exposure. Depending on the route of exposure (inhalation/aspiration, ingestion, direct dermal contact), PAHs can produce a broad range of health effects including lung disease, disruption of the hypothalamic-pituitary-adrenal axis, and altered immune response (National Academies of Sciences and Medicine 2016). Although PAHs are more rapidly metabolized and do not accumulate, as is the case with persistent organic pollutants, the toxic effects (lung disease, hypothalamic-pituitary-adrenal axis damage) may be long-lasting and initiate chronic disease conditions.

5.5.2 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 61 shows the locations of reported marine oil spill incidents on the coast of California from 2004 to 2020. PAHs were mentioned in the previous section as a component of oil (crude and refined) and motor exhaust, are 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). Acute exposure 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). 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 can impact sea turtles, for example, leatherback sea turtles are known 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 (Vargo et al. 1986; NOAA 2010).

In 2017 NMFS completed an ESA section 7 consultation with the Bureau of Ocean Energy Management (BOEM) and the Bureau of Safety and Environmental Enforcement (BSEE) on the effects to ESA-listed species from the proposed continuation of offshore oil and gas development and production activities in the Southern California Planning Area of the Pacific Outer Continental Shelf. That consultation found that the oil and gas activities on which the agencies consulted were not likely to adversely affect ESA-listed species or designated critical habitat. In October 2021, an offshore pipeline connected to an oil processing platform on the Pacific Outer Continental Shelf off Huntington Beach leaked, spilling oil about 94,600 liters of oil into the ocean. The consultation was reinitiated in order to reassess and update the oil spill risk analysis and to consider newly designated critical habitat for humpback whales (Central America and Mexico DPSs) in portions of the action area. The oil spill risk analysis determined that up to 26 East Pacific DPS green sea turtles could experience some level of adverse health effects, up to half those individuals (13) could die, and up to 30,344 black abalone could experience some level of adverse health effects, including some mortalities. ESA-listed cetaceans were not expected to be adversely affected under the spill risk scenarios.

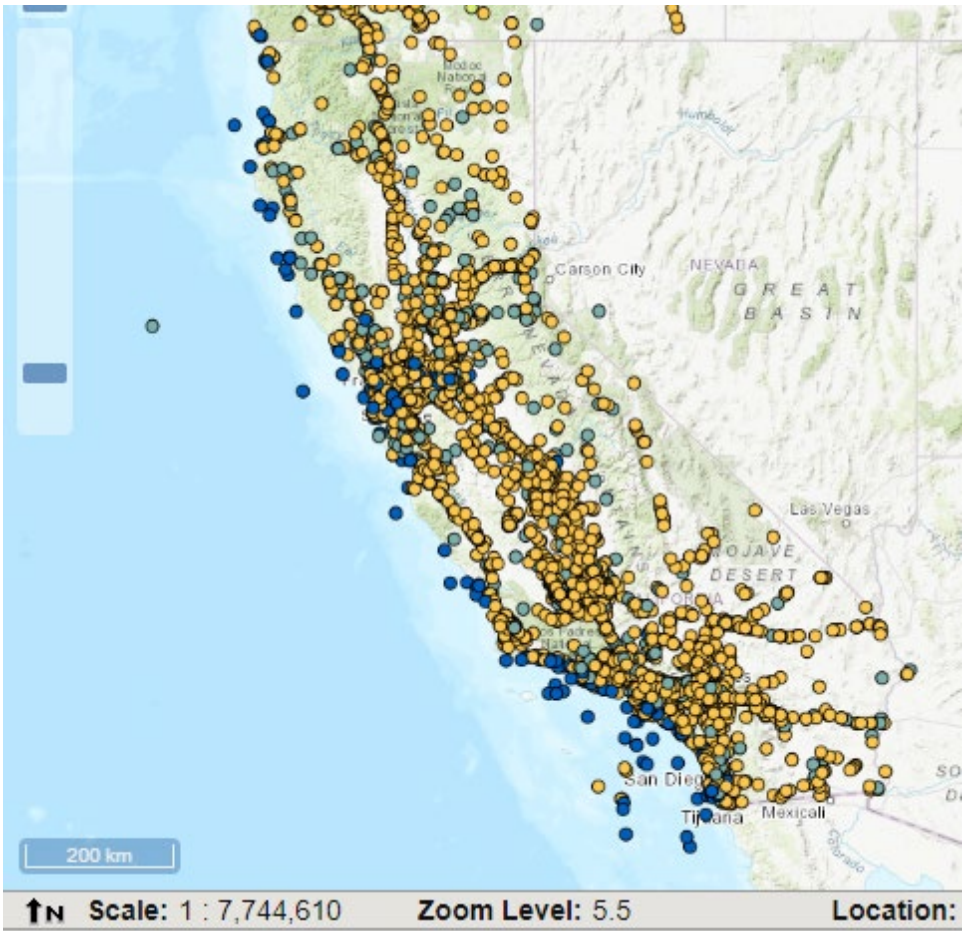


Figure 61. Oil Spill Incidents on the Coast of California from 2004 to 2020 (NOAA Office of Response and Restoration 2021). Marine oil spills locations are indicated by the dark blue circles.

5.5.3 Nutrient Loading and Algal Blooms

Industrial and municipal activities can result in the discharge of large quantities of nutrients into coastal waters. Excessive nutrient enrichment results in eutrophication, a condition associated with degraded water quality, algal blooms, oxygen depletion, loss of seagrass and coral reef habitat, and in some instances the formation of hypoxic "dead zones" (USCOP 2004). Hypoxia (low dissolved oxygen concentration) occurs when waters become overloaded with nutrients such as nitrogen and phosphorus, which can enter the marine environment from runoff (e.g., after a rain storm) of agricultural areas or from dense urban areas, sewage treatment plant effluent, bilge water, atmospheric deposition, and other sources. An overabundance of nutrients can stimulate algal blooms resulting in a rapid expansion of microscopic algae (phytoplankton). When excess nutrients are consumed, the algae population dies off and the remains are consumed by bacteria. Bacterial consumption decreases the dissolved oxygen level in the water which may result in mortality of fish and crustaceans, reduced benthic and demersal organism abundance, reduced biomass and species richness, and abandonment of habitat to areas that are sufficiently oxygenated (Craig et al. 2001; Rabalais et al. 2002). Higher trophic level species

(e.g. turtles and marine mammals) may be impacted by the reduction of available prey as a result of hypoxic conditions.

Marine algal toxins are produced by unicellular algae that are often present at low concentrations but that may proliferate to form dense concentrations under certain environmental conditions (National Academies of Sciences and Medicine 2016). When high cell concentrations form, the toxins that they produce can harm marine life, and this is referred to as a harmful algal bloom (HAB). Marine mammals can be exposed to HAB toxins directly by inhalation or indirectly through food web transfer, and these toxins can cause severe neurotoxic effects (Van Dolah 2005). Mortality and morbidity related to HAB toxins have been increasingly reported over the past several decades, and biotoxicosis has been a primary contributor to large scale die-offs across marine mammal taxa (Van Dolah 2005; Simeone et al. 2015). A recent survey of the peer reviewed literature on marine mammal diseases and marine mammal mass mortality events suggests an increase in the frequency of marine mammal die-offs resulting from exposure to harmful algal blooms over the past 40 years (Gulland and Hall 2007).

Harmful algal blooms along the Pacific coast of North America are dominated by two organisms: *Alexandrium catenella* which produces saxitoxin, the causative agent of paralytic shellfish poisoning, and several *Pseudo-nitzschia* species whose toxic strains produce domoic acid, the causative agent for amnesic shellfish poisoning (alternately called domoic acid poisoning) (Anderson et al. 2008). Okadaic acid, produced primarily by *Dinophysis* species, the causative agent of diarrhetic shellfish poisoning, are less common but potentially emerging threats (Danil et al. 2021). Prior to 2000, toxic blooms were considered rare and unusual in southern California (Lange et al. 1994). In 2006, Busse et al. (2006) reported the presence of domoic acid in San Diego during elevated abundances of toxic Pseudo-nitzschia and concurrently in fish and mussels. This study provides evidence for the transfer of domoic acid from a local algal source in San Diego to higher trophic levels. Domoic acid concentrations off southern California during 2003, 2006, 2007, 2011, and 2017 were comparable to some of the highest values that have been recorded in the literature (Smith et al. 2018). Observations during these years linked domoic acid in the diets of marine mammals to mass mortality events among these animals (Smith et al. 2018). Danil et al. (2021) detected domoic acid in the feces and urine of 13 species of stranded or bycaught southern California cetaceans over an 18 year period (2001–2018), including fin whales (four out of five positive) and humpback whales (one out of four positive). Domoic acid concentrations known to cause acute toxicity in pinnipeds (1,000 ng/g in feces) (Goldstein et al. 2008) were recorded in one of the fin whales sampled (Danil et al. 2021).

Red tides have been reported off the coast of southern California for over a century (McGowan et al. 2017). Red tides occur when blooms of marine phytoplankton reach such high concentrations that the sea surface becomes noticeably discolored. In La Jolla, California, blooms are often caused by bioluminescent dinoflagellates (e.g., *Lingulodinium polyedrum*) (McGowan et al. 2017). Red tides and other algal blooms in southern California can be caused by toxic algal species, resulting in fish and shellfish mortality (Lewitus et al. 2012). Regardless

of toxicity, the sheer concentrations of organisms can lead to oxygen depletion and fish kills when blooms persist over extended periods.

A comprehensive study in Alaska that sampled over 900 marine mammals across 13 species, including several mysticetes, odontocetes, and pinnipeds, found detectable concentrations of domoic acid in all 13 species and saxitoxin in 10 of the 13 species (Lefebvre et al. 2016). Algal toxins may have contributed to the stranding and mortality of 30 whales (fin whales and humpback whale) found around the islands in the western Gulf of Alaska and the southern shoreline of the Alaska Peninsula starting in May 2015 (Rosen 2015; National Oceanic and Atmospheric Administration 2016; Summers 2016; Savage et al. 2017). Many whales that may occur in the action area may also feed in Alaskan waters.

5.5.4 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% 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 is fishing gear also represents a major, on-going threat to many marine species. Globally, 6.4 million tons of fishing gear is lost in the oceans every year (Wilcox et al. 2015). Drifting "ghost gear" in the ocean 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. 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 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 (Lutcavage et al. 1997; Laist et al. 1999). Santos et al. (2015) found that a surprisingly small amount of plastic debris was sufficient to block the digestive tract and cause death. They reported that 10.7% of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4% had ingested enough plastic to have killed them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death. Gulko and Eckert (2003)

estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives.

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% of sea turtles globally have ingested plastic debris and oceanic life stage turtles are at the highest risk of debris ingestion. Based on their model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler et al. 2016). This study also found the North Pacific gyre, which encompasses much of the HCTT action area, to be a regional hotspot for sea turtle debris ingestion. The North Pacific Ocean is among the most polluted ocean regions globally, with roughly half of all fish specimens and more than 75% of all sea turtle specimens examined in this region having consumed plastic (Savoca et al. 2022). 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 (NRC 1990; Lutcavage et al. 1997; Laist et al. 1999). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Kiessling 2003; Dagorn et al. 2013).

The Northwestern Hawaiian Islands, are within the North Pacific gyre. Those islands and reefs are accumulating marine debris that presents potentially lethal entanglement hazards and ingestion threats to numerous marine animals within the action area. From 1996 through 2014, nearly 837 metric tons (1.8 million lbs.) of marine debris, primarily derelict fishing gear, have been removed from the shallow reefs and shorelines of the Northwestern Hawaiian Islands (PISC 2016).

The HCTT transit corridor, which connects the Hawaii and California portions of the action area, overlaps with a major ocean plastic accumulation zone that is referred to as the Great Pacific Garbage Patch. Lebreton et al. (2018) estimated that at least 79,000 tons of ocean plastic are floating inside an area of 1.6 million km^2. At least 46% of this plastic was comprised of fishing nets, and microplastics accounted for 8% of the total mass but 94% of the estimated 1.8 trillion pieces floating in the area (Lebreton et al. 2018). Historical data from surface net tows (1970-2015) indicate that ocean plastic pollution within the Great Pacific Garbage Patch is increasing exponentially (Lebreton et al. 2018).

Abandoned, lost or discarded fishing gear includes all fishing gear in the environment no longer actively managed, whether it is lost accidentally or intentionally. This source of marine debris is estimated to make up 80 % of marine plastic pollution in Hawaii (Høiberg et al. 2025). In the Main Hawaiian Islands, the proportion of green turtles stranding with fishing line has increased steadily since the late 1990s, emerging as the most common stressor and primary threat as of

2016 (Høiberg et al. 2025). The relatively high prevalence of hook-and-line injuries in Hawaiian green sea turtles establishes this as the leading stranding threat of concern (Høiberg et al. 2025). Hazards associated with derelict or active fishing lines and hooks include: tight constrictions around flippers which can lead to tissue death and flipper amputation, or restricting the turtles' ability to reach the water's surface to breathe, leading to mortality; line constricting the neck and mouth, impeding their ability to feed; ingestion of hooks and line, causing internal blockages within the digestive system and buoyancy issues that can impede movement, predator avoidance, and feeding; external injuries from hooks that may cause bleeding and infection and impede movement (Høiberg et al. 2025).

Hawaiian monk seals become entangled in fishing and other marine debris at rates higher than reported for other pinnipeds (Henderson 2001). A total of 347 cases of monk seals entangled in fishing gear or other debris have been observed from 1982 to 2014 (Carretta et al. 2017). Nine documented deaths resulted from entanglement in marine debris (Carretta et al. 2017). From 2017-2021 there was one documented Hawaiian monk seal mortality resulting from marine debris and the average number of Hawaiian monk seal non-serious (or mitigated serious) injuries from marine debris was about 12 incidents per year(Carretta et al. 2025).

Many cases of marine debris ingestion and entanglement have been reported around the world for marine mammals (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). de Stephanis et al. (2013) also describe a case of mortality of a sperm whale related to the ingestion of large amounts of marine debris in the Mediterranean Sea.

An emerging threat to baleen whales, is the presence of microplastics (plastic fragments smaller than 5 mm) in the marine environment (Germanov et al. 2018). Microplastics are now widespread in the oceans and sediments, and have been found to interact with persistent organic pollutants and contaminate marine life when ingested (do Sul and Costa 2014; Mosca Angelucci and Tomei 2022; Ramanayaka et al. 2022). The marine food web might be affected by microplastic biomagnification, and baleen whales are at particular risk from microplastic ingestion as a result of their filter-feeding activity, particularly as microplastics are also absorbed and ingested by their prey (Fossi et al. 2012; Burkhardt-Holm and N'Guyen 2019; Kahane-Rapport et al. 2022). Kahane-Rapport et al. (2022) found that baleen whales (blue, fin, and humpbacks) in the California Current Ecosystem predominantly feed at depths that coincide with the highest measured microplastic concentrations in the pelagic ecosystem (i.e., 50–250 m), and they predicted that nearly all (99%) microplastic ingestion occurred via trophic transfer. Fish-

feeding whales such as humpbacks are likely exposed to lower levels of microplastic ingestion, estimated 200,000 pieces per day, as compared to krill-feeding whales such as blue whales that may ingest 10 million pieces per day (Kahane-Rapport et al. 2022).

5.6 Whaling

Whales within the action area were historically targeted, at first just through localized subsistence hunting, then later through broad scale commercial whaling. From 1864 through 1985, at least 2,400,000 baleen whales (excluding minke whales) and sperm whales were killed worldwide (Gambell 1999). From 1900 to 1965 nearly 30,000 humpback whales were taken in the Pacific Ocean, with an unknown number of additional animals taken prior to 1900 (Perry et al. 1999). Sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). In addition, 9,500 blue whales and 25,800 sperm whales were reported killed by commercial whalers in the North Pacific between 1910-1965 (Ohsumi and Wada 1972) (Barlow et al. 1997). Many of the whaling numbers reported in the twentieth century likely represent minimum estimates, as illegal or underreported catches are not included.

Prior to current prohibitions on whaling, most large whale species were significantly depleted to the extent it was necessary to list them as endangered under the Endangered Species Preservation Act of 1966. Since the end of large-scale commercial whaling, the primary threat to these species has been eliminated, although many whale species have not yet fully recovered from those historic declines. Although commercial whaling no longer targets the large, endangered whales in the proposed action area, historical whaling may have altered the age structure and social cohesion of these species in ways that continue to influence them.

In 1982, the International Whaling Commission issued a moratorium on commercial whaling, which went into effect in 1986. There is currently no legal commercial whaling by International Whaling Commission Member Nations party to the moratorium; however, whales are still killed commercially by countries that filed objections to the moratorium. Presently three types of whaling take place: (1) aboriginal subsistence whaling to support the needs of indigenous people; (2) special permit whaling; and (3) whaling conducted either under objection or reservation to the International Whaling Commission moratorium. Japan, Iceland and Norway have maintained whaling either through special permit or under objection/reservation to the moratorium.[18] Some of the whales killed in these fisheries are likely part of the same population of whales occurring within the action area for this consultation.

5.7 Directed Harvest of Sea Turtles

Sea turtles have been harvested throughout history as both a protein source (for meat or eggs) and as raw material in the manufacture of ornaments and artifacts. An additional threat unique to hawksbill turtles is the tortoiseshell trade. Tortoiseshell is made from hawksbill scutes and is

[18] https://iwc.int/management-and-conservation/whaling Accessed 2/21/25.

used to produce products such as sunglasses, bracelets, and ornamental boxes that are often sold illegally on the black market (Shattuck 2011).

For centuries, the harvest of sea turtles and turtle eggs was primarily limited to small-scale, artisanal and subsistence fisheries. In many parts of the world, the customs and traditions associated with the harvest, consumption and artistic use of sea turtle products have been passed from generation to generation and have developed cultural meaning and significance over time (Campbell 2003). Historically, green turtles have played a large role in Polynesian and Micronesian cultures. In addition to being used as a food source, native peoples all over the Pacific utilized all parts of the turtle making tools and jewelry out of the bones, and containers and utensils out of the carapace. A primary threat to the West Pacific leatherback population is the legal and illegal harvest of turtles and their eggs.

Although small-scale turtle fisheries still exist today, by the mid-20th century directed turtle harvest was dominated by large-scale commercial operations with access to global markets (Stringell et al. 2013). The Hawaiian green turtle was in a steep decline as of the 1970s because of direct harvest of both turtles and eggs by humans. By the late 1960s, the global capture of sea turtles had peaked at an estimated 17,000 tons (FAO 2011). Based on Japanese commercial import data, between 1970 and 1986 an estimated two million turtles (mostly hawksbills, greens, and olive ridleys) were harvested to satisfy the demand for turtle products in Japan alone (Milliken and Tokunaga 1987). To maximize efficiency, commercial harvesting effort was often concentrated at mass nesting sites or arribadas with high densities of breeding adult turtles.

Increased conservation awareness at the international scale has led to greater protection of marine turtles in recent decades. The Convention on International Trade in Endangered Species (CITES), which went into effect in 1975, helped to reduce demand and promote regional cooperation in increasing turtle populations. All six ESA-listed sea turtles are listed in CITES Appendix I, which provides the greatest level of protection, including a prohibition on commercial trade. Marine turtle species have also been listed on the International Union for Conservation of Nature Red List of Threatened Species since 1982 (IUCN 2017). In 1981, Ecuador, one of the two largest turtle harvesting nations at the time, banned the export of sea turtle products. In 1990, following international pressures, Mexico, the other major turtle exporter, closed commercial fisheries and instituted a moratorium on the take of turtles and eggs (Senko et al. 2014).

Leatherback turtles are protected by regulatory mechanisms in all four nations where this population nests, but laws are largely ignored and not enforced (NMFS and USFWS 2020a). In Indonesia, the direct harvest of turtles and eggs likely persists, although this threat has been minimized at Jamursba-Medi, Wermon, and Buru Island beaches due to the presence of monitoring programs and associated educational outreach activities (NMFS and USFWS 2020a).

Humber et al. (2014) documented the change in the legal take of sea turtles over the past three decades. Just considering the 46 countries that still allow sea turtle directed take (including the four with current moratoria), turtle harvest has decreased by more than 60% over the past three

decades. The average number of turtles killed in these fisheries annually has declined steadily over time: 116,420 in the 1980s; 68,844 turtles in the 1990s; and 45,387 in the 2000s (Humber et al. 2014). While legal directed take of sea turtles has declined significantly, illegal harvest may still represent a significant source of sea turtle mortality, one that is more difficult to estimate. The scale of global illegal take is likely to be severely underreported due to the inherent difficulty in collecting data on such activity (Humber et al. 2014).

5.8 Human-Caused Mortality of Pinnipeds

The average annual observed human-caused mortality and serious injury of Guadalupe fur seals from the most recent available non-fishery sources data, 2013-2017, is 2.6 animals annually (13 animals / 5 years) (Carretta 2021b). Marine debris was recorded as the source of injury or mortality in twelve of these stranding cases. Other sources included entanglement in fishing nets, blunt force trauma and shootings. The actual number of Guadalupe fur seals killed or injured is likely greater since not all injured or killed animals strand (e.g., shark predation) and not all strandings are reported.

Currently, human activities in the Northwest Hawaiian Islands are limited and human disturbance is relatively rare, but human-seal interactions have become an important issue in the Main Hawaiian Islands. The number of (non-fishery) human-caused mortalities of Main Hawaiian Island monk seals during the 5-year period of 2017-2021 was six cases, which included intentional blunt force traumas and gunshots (Carretta et al. 2025). The number of human-caused mortalities of monk seals that are recorded should be considered a minimum estimate since only about half (46%) of carcasses of monk seals which died in the Main Hawaiian Islands are detected (Harting et al. 2021).

5.9 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.” 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 2023a). A primary cause for the UME was not identified but ecological factors, including El Niño (see Oceanic Temperature, Section 5.2), the marine heatwave and Pacific Coast Domoic Acid Bloom (see Nutrient Loading and Algal Blooms, Section 5.5.3), were likely contributors (NMFS 2023a).

An UME was declared for Guadalupe fur seals beginning in January 2015 (NMFS 2021a). 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 off those states (Figure 62 and Figure 63). Starting in 2015, strandings in California were 8 times higher than the historical average, and continued to remain well above average through 2020 (NMFS 2021a). Guadalupe fur seal strandings generally peak from April through June each year. Stranded individuals were mostly weaned pups and juveniles, ages 1-2 years old. Most stranded

individuals showed signs of malnutrition and had secondary bacterial and parasitic infections. Strandings decreased in 2021 and the UME was closed in September of that year.

From 2018 to 2023, an eastern North Pacific gray whale (not an ESA-listed population) UME occurred along the western coast of North America (NMFS 2024d). It occurred from December 2018 through November 2023, with peak strandings occurring between December 2018, and December 2020. NOAA Fisheries and partners investigated the event. The UME involved 690 gray whale strandings, including 347 in the United States, 316 in Mexico, and 27 in Canada (Figure 64). Strandings occurred from Alaska to Mexico along the west coast of North America, including in the whale's wintering, migratory, and feeding areas. The investigative team concluded that the preliminary cause of the UME was localized ecosystem changes in the whale's Subarctic and Arctic feeding areas that led to changes in food, malnutrition, decreased birth rates, and increased mortality all documented during the UME.

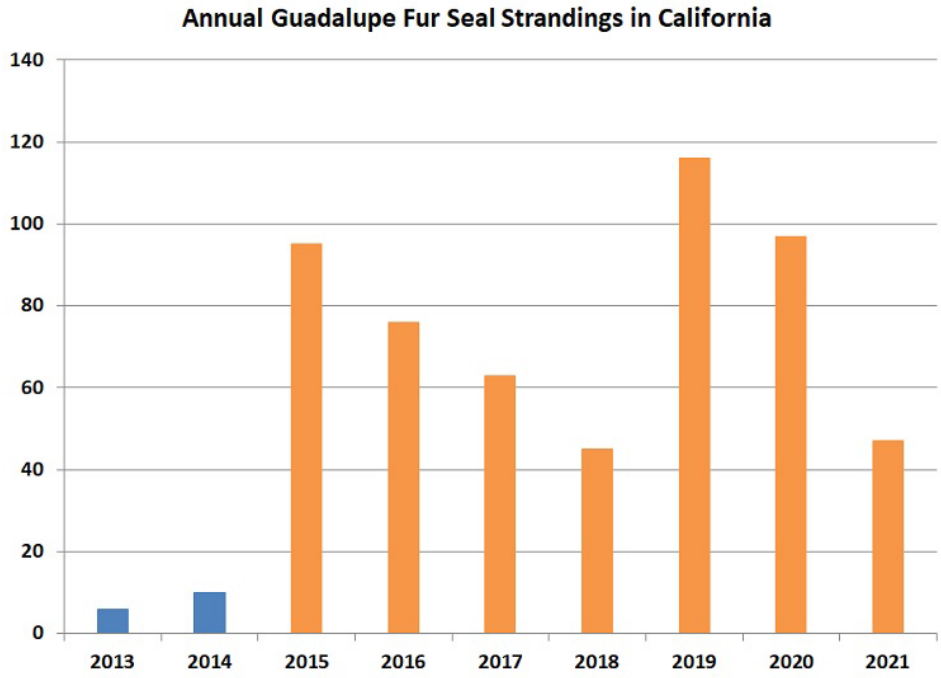


Figure 62. Guadalupe Fur Seal Annual Strandings in California, 2013 to 2021. Orange bars indicate unusual mortality event years (NMFS 2021a).

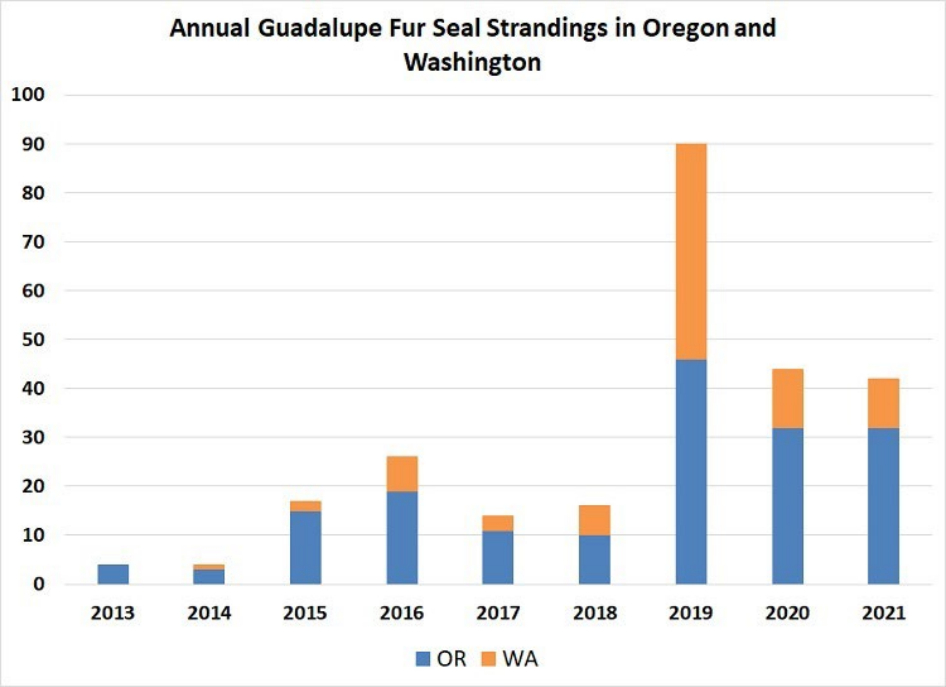


Figure 63. Guadalupe Fur Seal Annual Strandings in Oregon and Washington, 2013 to Present. Blue – Oregon; Orange – Washington (NMFS 2021a).

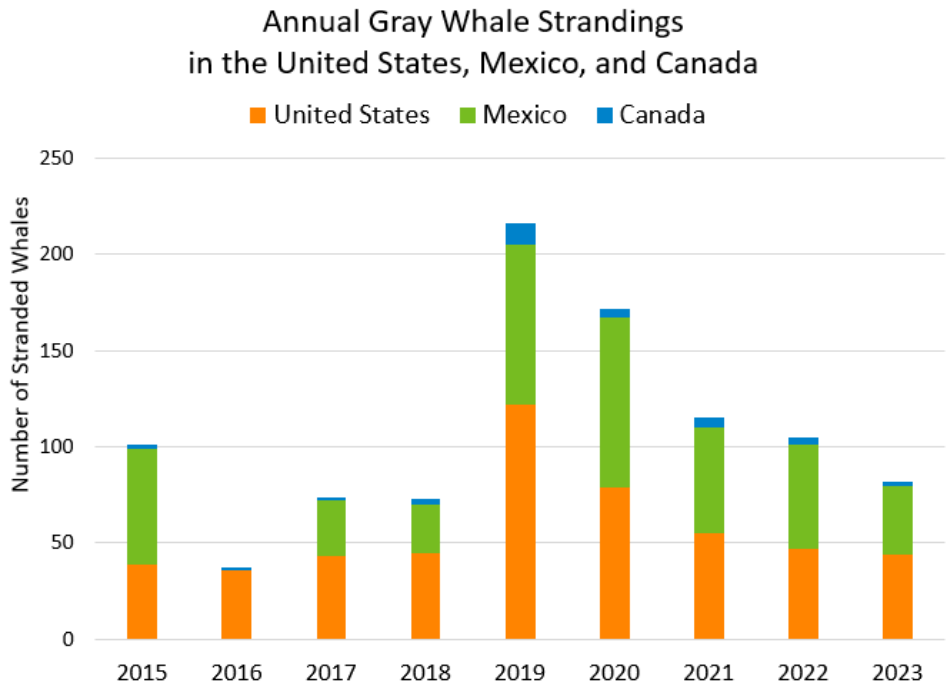


Figure 64. Annual Gray Whale Strandings in the United States, Mexico, and Canada, 2015 to 2023

5.10 Fisheries Catch and Bycatch

In this section, we summarize the best available information on fisheries bycatch, accidental capture or other unintended interactions, of ESA-listed species in the action area.

5.10.1 Sea Turtles

Sea turtle bycatch occurs in both large-scale commercial fishing operations as well as small-scale and artisanal fisheries throughout the action area. Sea turtle bycatch rates (i.e., individuals captured per unit of fishing effort) and mortality rates (i.e., individuals killed per number captured) can vary widely both within and across particular fisheries due to a combination of factors. These include gear types and gear configurations, fishing methods (e.g., depth fished, soak times), fishing locations, fishing seasons, time fished (i.e., day versus night), and turtle handling and release techniques used (Wallace et al. 2010; Lewison et al. 2013). Entanglement in fishing gear and/or plastics can result in severe ulcerative dermatitis, and amputation of flippers (Orós et al. 2005). If mortality is not directly observed during gear retrieval, it may occur after the turtle is released due to physiological stress and injury suffered during capture. Recent studies indicate that underwater entrapment in fishing gear (i.e., trawls and gillnets) followed by rapid decompression when gear is brought to the surface may cause gas bubble formation within the blood stream (i.e., embolism) and tissues leading to organ injury, impairment, and even post-release mortality in some bycaught turtles (Garcia-Parraga et al. 2014; Fahlman et al. 2017).

Lewison et al. (2014) used the bycatch data from 1990-2008 to identify global hotspots of turtle bycatch intensity. High-intensity sea turtle bycatch was most prevalent in three regions: the eastern Pacific Ocean, southwest Atlantic Ocean, and Mediterranean Sea. Spotila (2000) reported a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s of 1,500 animals. He estimated that this represented about a 23% mortality rate (or 33% if most mortality was focused on the East Pacific population). Lewison et al. (2004) estimated between 2,600 and 6,000 loggerhead turtles were captured and killed in Pacific Ocean longline fisheries in 2000. Average annual longline bycatch mortality estimates for the years 2013 to 2020, by sea turtle species, based on annual Western & Central Pacific Fisheries Commission reports[19] for participating countries operating in the Western and Central Pacific Ocean (i.e., west of 150° W) are as follows: 645 green; 76 leatherback; 390 loggerhead; and 2,337 olive ridley.

Below, we summarize the major U.S. commercial fisheries within the HCTT action area that result in sea turtle bycatch. The primary turtle species of concern for U.S. fisheries bycatch in the Pacific are leatherbacks and loggerheads, due to their critical conservation status (Moore et al. 2009).

[19] Accessible online at: https://www.wcpfc.int/node/29966

West Coast Longline Fishery

The west coast longline fishery operates in the north Pacific ocean, mainly from the U.S. EEZ west to 140 degree west longitude and from the equator to 35 degree north (NMFS 2016e). This fishery primarily targets bigeye tuna, although other tuna and non-tuna species are also caught and retained. As of 2016 there was only one boat participating in this fishery, although fishing effort is expected to increase in the future (NMFS 2016e). Sea turtle incidental take authorized over a ten-year period (starting in 2016) as provided for in the ITS of the 2016 biological opinion (NMFS 2016e) on this fishery is as follows:

- Green, East Pacific DPS and Central North Pacific DPS: one total (lethal or non-lethal).
- Leatherback: four total, including up to two lethal.
- Loggerhead, North Pacific DPS: one (lethal or non-lethal).
- Olive ridley: six total (lethal or non-lethal).

Hawaii Pelagic Longline

Domestic longline fishing around Hawaii consists of two separately managed fisheries: a deep-set fishery that primarily targets bigeye tuna and a shallow-set fishery that targets swordfish. The shallow-set fishery operates almost entirely north of Hawaii. The deep-set fishery operates primarily to the south of Hawaii between the equator and 35° N, although in some years this fishery expands northward and overlaps with the shallow-set fishery.

In 1999, the shallow-set longline fishery targeting swordfish was closed by court order due to high levels of sea turtle bycatch. Before the closure took effect, an estimated 417 loggerheads and 110 leatherbacks (McCracken 2000) were captured annually (with about 40% mortality) (Gilman et al. 2007) in Hawaii's longline fisheries (shallow and deep-set combined). Subsequent court orders led to regulations in 2001 prohibiting all Hawaii longline vessels from targeting swordfish until 2004. When the shallow-set fishery was reopened in 2004 it was restricted to considerably less fishing effort than pre-2001 levels. As a result, the deep-set fishery targeting tuna made up an increasingly larger proportion of Hawaii's longline fishing effort since 2004. A final rule published in 2004 (69Fed. Reg. 17329) established a limited shallow-set swordfish fishery and required the use of circle hooks with mackerel-type bait, a combination that had proven effective at reducing interactions with leatherback and loggerhead turtles in the Atlantic longline fishery (Watson et al. 2005). The use of circle hooks with mackerel-type bait reduced sea turtle interaction rates by approximately 90% for loggerheads and 83% for leatherbacks compared to the previous period 1994-2002 when the shallow-set fishery was operating without these requirements (Gilman et al. 2007). Annual sea turtle bycatch limits (17 loggerhead or 16 leatherback turtles) were also established for the swordfish fishery as part of the 2004 rule.

Based on analysis of reporting after 2004 and predictions of future vessel participation/fishery effort, a biological opinion (NMFS 2023f) on the continued operation of the Hawaii deep-set longline fishery anticipates the following mean annual captures of sea turtles and expected

mortalities (in parentheses) as a result of those interactions: 17 leatherback (6), 13 green (12), 9 North Pacific loggerhead (5), 71 olive ridley (65). Analysis conducted in the biological opinion on the continued operation of the Hawaii shallow-set longline fishery (NMFS 2019d) anticipates the following annual captures of sea turtles and expected mortalities (in parentheses) as a result of those interactions: 21 leatherback (3), 5 green (1), 36 North Pacific loggerhead (6), 5 olive ridley (1).

West Coast Drift Gillnet Fishery for Highly Migratory Species

The West coast drift gillnet fishery targets swordfish and thresher sharks in the U.S. EEZ and adjacent high seas off the coasts of California, Oregon, and Washington (NMFS 2013a). Leatherback sea turtles have been reported as bycatch in the California drift gillnet fishery. (Martin et al. 2015a) estimated leatherback sea turtle bycatch in this fishery for the 20-year period 1990-2009, with a total bycatch range of 104–242 leatherbacks (52–153 estimated deaths). Estimated leatherback entanglements declined each year over this time period, reaching low levels in 2001 after implementation of the Pacific Leatherback Conservation Area, which prohibits drift gillnet fishing in large portions of the historical fishing grounds, either seasonally or conditionally, to protect endangered leatherback and loggerhead sea turtle populations (66Fed. Reg. 44549; August 24, 2001). Since the time/area closure went into effect in 2001, interactions reduced in this fishery by approximately 80-90% (NMFS and USFWS 2020b). From 2001 through 2018, the estimated total leatherback bycatch in this fishery is 16 turtles (Carretta 2021a).

The biological opinion on the continued operation of the U.S. West Coast large mesh drift gillnet fishery (NMFS 2023h) expects the following sea turtle incidental take as total bycatch (and mortality or serious injury from that interaction) over the next five years: up to two (one serious/fatal) leatherback, up to one (one serious/ fatal) East Pacific green, up to three (one serious/ fatal) North Pacific loggerhead, up to one (one serious/ fatal) olive ridley.

Pot and Trap Fisheries

Leatherbacks have occasionally been documented entangled in pot and trap fisheries gear off California, and 1 leatherback was confirmed entangled in California fixed fishing gear in 2019. To reduce loggerhead bycatch, NMFS implements seasonal closures and additional closures during El Niño events (84Fed. Reg. 24398; May 28, 2019), when loggerheads are likely to forage on multiple prey items, including pelagic red crabs (*Pleuroncodes planipes*), off southern California (Welch et al. 2019). More recently, the fishery has adopted a 2-year bycatch cap of 2 loggerheads and increased observer coverage (30% in 2016 and 2017; 100% in 2018) (USFWS 2020).

Hook and Line

Based on a study of strandings data from 1984-2018, Brunson et al. (2022) identified interactions with Hawaii nearshore hook-and-line fishing gear as the primary threat to hawksbill sea turtles in that region. The fishing gear identified within the hook-and-line fisheries mainly consisted of

monofilament or Kevlar line and circle or J-hooks typically used by fishermen targeting nearshore coastal marine resources within State-managed waters (Honebrink and Hendricks 2016; Delaney et al. 2017). Other nearshore fishing gears contributing to hawksbill strandings include gill nets, trawls nets, and lobster and crab traps (Brunson et al. 2022). Høiberg et al. (2025) documented a steady increase in the proportion of stranded green sea turtles in Hawaii with injuries from either fishing hooks or lines since the early 1980's. In recent years (2020-2024), hook and line injuries combined accounted for 42% of green sea turtle strandings across the Main Hawaiian Islands (Høiberg et al. 2025), with 36% of green sea turtle fishing line injuries and 14% of hook injuries resulting in mortality (Høiberg et al. 2025).

5.10.2 Marine Mammal Fishery Interactions

Entrapment and entanglement in fishing gear is a frequently documented source of human-caused mortality for marine mammals (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). For some marine mammal populations, the impacts from fisheries likely have significant demographic effects (Read et al. 2006). Many marine mammals that die from entanglement in commercial fishing gear tend to sink rather than strand ashore, thus making it difficult to fully assess the magnitude of this threat. When not immediately fatal, entanglement or ingestion of fishing gear can impede the ability of marine mammals to feed and can cause injuries that eventually lead to infection and death (Wells et al. 2008; Cassoff et al. 2011; Moore and Van der Hoop 2012). Other sublethal effects of entanglement include increased vulnerability to additional threats, such as predation and ship strikes, by restricting agility and swimming speed. There are also costs likely to be associated with nonlethal entanglements in terms of energy and stress (Moore and Van der Hoop 2012).

In 1994, the MMPA was amended to formally require the development of a take reduction plan when bycatch exceeds a level considered unsustainable and would lead to marine mammal population declines if not mitigated. At least in part as a result of the MMPA bycatch amendment, estimates of bycatch in the Pacific declined by a total of 96% from 1994 to 2006 (Geijer and Read 2013). Cetacean bycatch declined by 85% from 342 in 1994 to 53 in 2006, and pinniped bycatch declined from 1,332 to 53 over the same time period.

The number of confirmed whale entanglements per year detected off the U.S. west coast from 1982-2017 are shown in Figure 65 (Saez et al. 2021). West coast entanglements are mostly reported off the coast of California. From 2000 to 2012, an average of eight large whale entanglements were observed and reported per year in California (Saez et al. 2013). Humpback whales and gray whales are the most commonly entangled cetacean species off California. There are some sperm whales, a few blue and a few fin whales also reported over this time frame. Confirmed reports of entangled animals likely represent only a small fraction of the total number of entanglements that are actually occurring.

Traps and pots are the most common fishing gears reported as entangling West Coast whales, accounting for about 45% of entanglements (Saez et al. 2013). Dungeness crab fishing gear is the

most common source of entanglements in recent years, of those incidents that could be traced to a specific fishery (Saez et al. 2021).

The number of large whale entanglements may be increasing over time. In 2016, 66 separate cases of entangled whales were reported off the coast of California, 51 of which were humpback whales (NMFS 2017a), this coincides with Pacific marine heat wave (see Section 5.2, Oceanic Temperature Regime). About 20% of reported entanglements in 2016 were from Southern California (Santa Barbara, Los Angeles, Orange, and San Diego counties), including ten humpbacks, two blue whales and one gray whale. There were 19 confirmed humpback whale entanglements off the coasts of Washington, Oregon, and California in 2017, 34 in 2018, 17 in 2019, and 10 in 2020, during which observations and reporting were likely affected by the COVID-19 pandemic (NOAA 2020; NMFS 2021b). One sperm whale entanglement was also reported in 2020 off Santa Barbara, California.

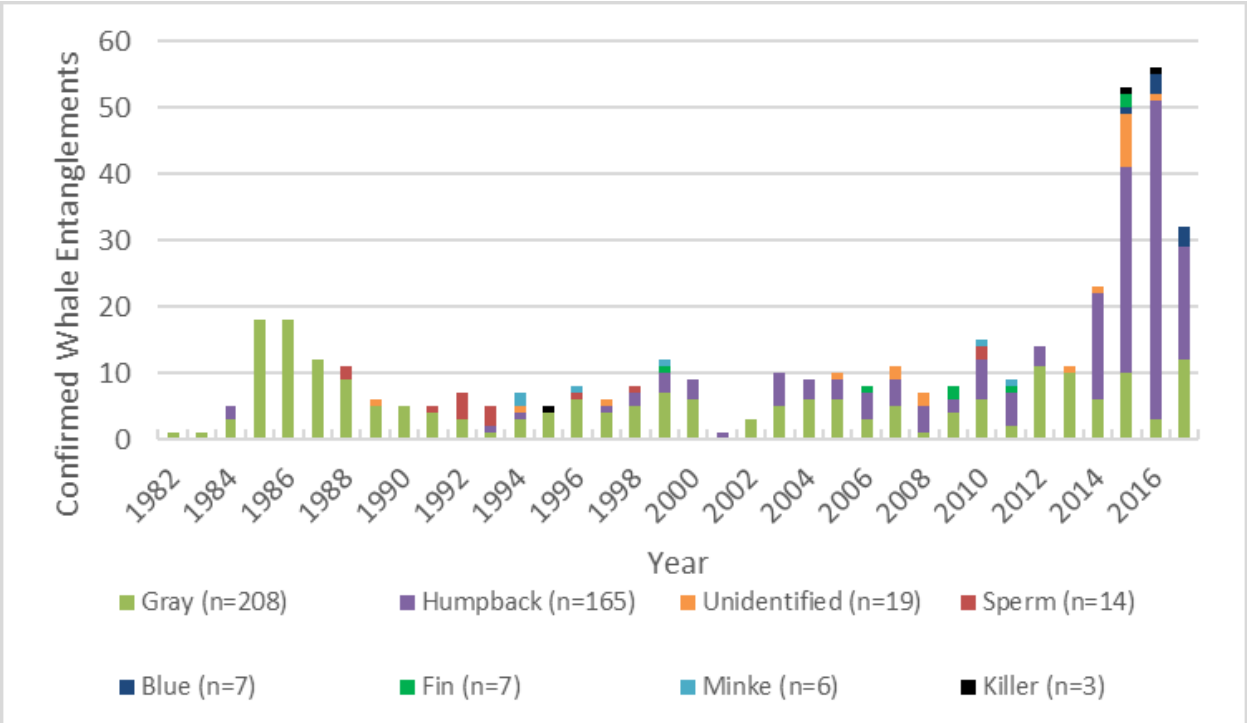


Figure 65. Confirmed Whale Entanglement Reports off the U.S. West Coast from 1982-2017 (N=429) by Year and Color-coded by Species (Saez et al. 2021)

From 2021 to 2023, 27-30 entangled large whales were confirmed per year off the coasts of Washington, Oregon, and California, or off the coast of other countries and states but entangled in U.S. West Coast commercial fishing gear.[20] This is still above the numbers of reported entanglements prior to 2014, but consistently below the high point of more than 50 reported

[20] https://storymaps.arcgis.com/stories/267aaa6e6a494d44be437b597f3993a3 Accessed 3/17/2025.

entanglements in 2016. Humpback whales continue to be the most common species entangled (16-18 per year), followed by gray whales which make up most of the remaining reports. There was one fin whale entanglement report in 2021 and 2022.

The estimated number of Eastern North Pacific blue whales seriously injured or killed due to commercial fisheries during 2017-2021 is 0.61 blue whales annually (Carretta et al. 2025). The estimated number of fin whales from the California/Oregon/Washington stock seriously injured or killed due to commercial fisheries during 2017-2021 is 0.64 fin whales annually (Carretta et al. 2025). The mean annual humpback whale mortality and serious injury observed and reported in California, Oregon, and Washington commercial fisheries for the period 2016-2020 is 20.9 whales (NMFS 2023e). The mean annual commercial fishery-related serious injury and mortality of sperm whales during 2017-2021 from the California/Oregon/Washington stock is 0.52 sperm whales per year (Carretta et al. 2025). These are likely underestimates of the serious injury and mortality of large whales in the region, because not all cases are detected and there is no correction factor available to account for undetected events.

Using photo-ID images of entangled humpback whales along the U.S. Pacific Coast between 1982 and 2017, Tackaberry et al. (2022) examine pre- and post-entanglement sighting histories to evaluate the deviation from the expected mortality rates (or lack of resighting) caused by entanglements. Entangled whales were resighted less often than the control group, suggesting a higher mortality rate after entanglement and there were indications of higher risk of entanglement for younger whales (Tackaberry et al. 2022).

The total number of confirmed humpback whales reported entangled in Hawaii from 2001 to 2023 was 154 or about seven per year (Lyman 2014; Cartwright et al. 2025). All but three of these reports (one sei whale and two sperm whales), were humpback whales. The most commonly reported gears associated with entanglements in Hawaii are fish pots (50%) and longlines (23%).

False killer whales in Hawaiian waters have been seen taking catches (depredation) from commercial longlines and trolling lines (Shallenberger et al. 1981; Nitta and Henderson 1993). Interactions with these fisheries operations can result in injury, including disfigurement to dorsal fins (Shallenberger et al. 1981; Zimmerman 1983; Nitta and Henderson 1993; Baird and Gorgone 2005; Forney and Kobayashi. 2007; McCracken and Forney 2010). NMFS published a final rule to implement the False Killer Whale Take Reduction Plan on November 29, 2012 (77Fed. Reg. 71260). The final rule includes gear requirements ("weak" circle hooks and strong branch lines) in the deep-set longline fishery, longline closure areas, and training and certification for vessel owners and captains in marine mammal handling and release. It was estimated that from 2017 to 2021, a mean of 0.1 MHI insular false killer whales were killed or seriously injured per year from interactions with the Hawaii deep-set longline fishery and zero per year from the shallow set longline fishery (Carretta et al. 2025). The 2023 stock assessment report (Carretta et al. 2023a) highlights the geographic overlap between various nearshore

fisheries and evidence of interactions with hook-and-line gear, including six MHI IFKWs stranded between 2010 and 2021.

Fishery interactions with Hawaiian monk seals can include direct interactions with gear (hooking or entanglement), competition for prey species, and the seals consuming discarded catch. Fishery interactions are a serious concern in the Main Hawaiian Islands, especially involving nearshore state managed commercial and recreational fisheries. Over the 30-year period between 1982 and 2012, approximately 11 Hawaiian monk seals have been observed entangled in fishing gear or other marine debris annually, with a total of nine documented deaths (Carretta et al. 2014). Gobush et al. (2017) individually identified 297 monk seals between 1988 and 2014 and recorded that 83 (28%) of these had at least one documented hooking or entanglement. Most individuals were aged two years or younger and a quarter of them were hooked or entangled multiple times. The proportion of monk seals alive one year after a documented fisheries interaction varied by age class and ranged between 76% and 84% (Gobush et al. 2017). Survival one year later for monk seals with a documented fisheries interaction versus matched controls (all age classes combined) was not significantly different.

The estimated minimum annual fishery-related mortality or serious injury for the 5-year period of 2018 to 2022 is 2.2 monk seals per year, which includes mortality from nearshore gillnets, hook-and-line gear, and mariculture (Carretta et al. 2025) (Carretta et al. 2025). In 2022, 22 seal hookings were documented, one of which were classified as serious, and 21 of which were classified as non-serious injuries, although eight of those would have been deemed serious had they not been mitigated as non-serious, injuries (Carretta et al. 2025).

No Guadalupe fur seals have been observed entangled in California gillnet fisheries between 1990 and 2017, although stranded animals have been found entangled in gillnets of unknown origin (Carretta 2021b). Guadalupe fur seals occasionally are observed hooked in the Hawaii shallow set longline fishery which has 100% observer coverage. Between 2018 and 2022 there were four observed mortalities or serious injuries involving this species from this fishery (Carretta et al. 2025). Unidentified fishery interactions, including generic gillnets and trawls of unknown origin, accounted for 32 observed Guadalupe fur seal mortalities or serious injuries from 2018 to 2022 (Carretta et al. 2025). Observed human-caused mortality and serious injury of Guadalupe fur seals likely represents a fraction of the true impacts because not all cases are reported or documented (Carretta et al. 2017).

In addition to the threats of entanglement and entrapment, fisheries operations can also result in changes to the structure and function of marine ecosystems that adversely affect marine mammals, including loss of prey species and alteration of benthic structure. Overfishing of many fish stocks results in significant changes in trophic structure, species assemblages, and pathways of energy flow in marine ecosystems (Jackson et al. 2001; Myers and Worm 2003). These ecological changes may have important, and likely adverse, consequences for populations of marine mammals (DeMaster et al. 2001). For instance, depletion of preferred prey could lead to a less nutritional diet and decreased reproductive success.

5.10.3 ESA-Listed Salmonids and Sturgeon

At the regional scale for salmon bycatch, Richerson et al. (2022) summarized both observed and estimated bycatch of salmon within west coast fisheries from 2002 to 2020. The data summarized came from NMFS observer programs and the Pacific States Marine Fisheries Commission electronic monitoring and catch monitoring programs. The focus of this report was on Chinook and coho salmon bycatch trends. This summary did not include a genetic analysis or identify the specific ESUs, so estimating how many ESA-listed fish were taken as bycatch was not discernible in this report. Chinook bycatch in these programs ranged from a low of approximately 2,100 Chinook in 2009, to approximately 26,000 Chinook in 2003. The hake fishery consistently had the highest bycatch of Chinook salmon in all years except 2002 and 2003 when the shoreside non-hake fisheries alone exceeded 15,000 Chinook (Richerson et al. 2022). By comparison, coho bycatch in these fisheries was far lower than Chinook, largely not exceeding 500 coho bycatch in any given year (Richerson et al. 2022). As with Chinook bycatch, coho bycatch was typically highest in the hake fishery data.

In a review of the California Set Gillnet Observer Program Observed Catch Summary from 2007 to 2017, NMFS (2022) identified eight Chinook (unknown ESU) and one unidentified salmon in 2011, and one Chinook (unknown ESU) in 2013 as potential ESA-listed bycatch. In a review of other species caught in the California halibut fishery from 2000 to 2020, Matthews et al. (2022) did not note any ESA-listed species, thought they did include that 0.06 metric tons (60 kgs [132 lbs.]) of unidentified sturgeon occurred as bycatch.

Richerson et al. (2020) summarized bycatch at the regional scale for green sturgeon in fishery sectors observed by the West Coast Groundfish Observer Program and the At-Sea Hake Observer Program from 2002 to 2019. As there is no visual distinction between the ESA-listed southern DPS and non-listed northern DPS fish while at sea, and tissue samples were not collected from all fish for genetic stock identification (GSI), green sturgeon bycatch compiles both listed and non-listed fish. However, a GSI analysis of a subset of the data indicated the proportions between the DPS's differed spatially, with 48% of green sturgeon caught off the Oregon and Washington coasts and 96% of individuals caught off the California coast assigned to the southern DPS (Richerson et al. 2020). Based on these proportions, Richerson et al., (2020) estimated that bycatch of southern DPS green sturgeon ranged between 0 and 24 fish annually from 2002 to 2015, and 0–12 fish annually from 2015 to 2019. Bottom trawl fisheries resulted in substantially more bycatch of green sturgeon than other commercial fisheries.

Gustafson et al., (2023) summarized the observed and estimated bycatch at the regional scale for eulachon in west coast commercial fisheries from 2002 to 2021. Analysis of this 19-year data set indicates an estimated total of 20,396 eulachon were captured in the groundfish fishery. However, in 2021 alone, 23,820 eulachon were caught as bycatch, surpassing the previous 19 years combined (Gustafson et al. 2023). Further, the authors note that eulachon bycatch in all U.S. West Coast groundfish fisheries increased from an estimated 792 in 2018 and 2,663 in 2019, to an estimated 8,528 eulachon in 2020 and 23,820 eulachon in 2021. The combined 2020

and 2021 estimated bycatch represents about 72% of the 2002–2021 total. As with green sturgeon, there is no visual distinction between the ESA-listed southern DPS and non-listed northern DPS eulachon while at sea, and tissue samples were not collected from all fish for GSI.

In summary, west coast observer programs, electronic monitoring programs, and catch monitoring programs have documented, as bycatch, various fish species with either ESUs or DPSs listed under ESA, including Chinook and coho salmon, green sturgeon and eulachon (Richerson et al. 2020; Richerson et al. 2022; Gustafson et al. 2023). No other ESA-listed fish species were included as bycatch in the most recent summary of west coast fisheries (Sommers et al., 2023).

5.10.4 ESA-listed Elasmobranchs

Scalloped hammerhead sharks are both targeted and taken as bycatch in many global fisheries (e.g., bottom and pelagic longlines, coastal gillnet fisheries, artisanal fisheries). This species is highly desired for the shark fin trade because of its fin size and high fin ray count. In the United States, scalloped hammerhead sharks are mainly caught as bycatch in longline and coastal gillnet fisheries and are known to suffer high post-release mortality rates (76Fed. Reg. 72891). In the Pacific, shark bycatch occurs primarily in the Hawaii-based pelagic longline fishery. An observer program has been in place since 1995 with targeted coverage of 25% in the deep-set sector and 100% in the shallow-set sector. Observer data from 1995-2006 indicated a very low catch of scalloped hammerhead sharks in this fishery (56 individuals on 26,507 sets total, both fishery sectors combined) (Miller et al. 2014). Hawaii is in the range of the Central Pacific DPS of scalloped hammerheads, which is not listed. If the deep-set fishery operates south of Hawaii, there is potential for exposure to the threatened Indo-West Pacific DPS below 10° N. Scalloped hammerheads are also occasionally caught in U.S. recreational fisheries, although recreational catch estimates are often unreliable due to the rare event nature of capture and species identification issues.

The most significant threat to giant manta ray populations is commercial fishing. Giant manta rays are a targeted species for the mobulid gill raker market. Gills from mobuilds (i.e., rays of the genus Mobula, including *Manta* spp.) are dried and sold in Asian dried seafood and traditional Chinese medicine markets (O'Malley et al. 2017). In addition to the threat from directed fishing, giant manta rays are also captured incidentally in industrial purse seine, longline and artisanal gillnet fisheries. Incidental bycatch is a particular concern in the eastern Pacific Ocean, and the Indo-Pacific (Miller 2016). From 2010-2016, an average of 22 giant manta rays have been caught annually as bycatch in the Hawaii deep-set longline fishery (based on extrapolated estimates from observer covered trips) (Kapur and Yau 2018). An average of one giant manta ray has also been reported annually as bycatch in the Hawaii shallow-set longline fishery during this time period (based on 100% observer coverage) (Kapur and Yau 2018),

Bycatch-related mortality in commercial longline fisheries remains the largest source of mortality of oceanic whitetip sharks (Young and Carlson 2020) and the most significant threat to this species (NMFS 2023g). Based on a historical catch reconstruction conducted by Tremblay-

Boyer et al. (2019), catches of oceanic whitetip shark in the Western and Central Pacific Ocean longline fishery peaked around 2001 at about 540,000 individuals and declined to only about 80,000 individuals in 2016. Walsh and Brodziak (2022) estimated 9,501 oceanic whitetip sharks were caught in Hawaiian longline fisheries (deep-set and shallow-set combined) from 1994-2019 (or about 594 per year) over a total of 90,139 longline sets. Although the oceanic whitetip shark has been prohibited from retention by all the relevant major tuna Regional Fishery Management Organizations, oceanic whitetip fins were still commonly found in the Hong Kong fin market in a survey conducted from 2014-2016 (Cardeñosa et al. 2018).

Reported worldwide catches for oceanic whitetip shark for five years of available data (2012-2017) ranged from 62-519 metric tons per year (NMFS 2023g). Spatiotemporal predictions of oceanic whitetip shark catches indicated that higher catches occurred during the boreal spring (Apr-Jun) with higher catches in the Humboldt current (79− 87°W; 10−16°S), along the central Eastern Pacific between 2− 8°N from 105°W to the westernmost of the Inter-American Tropical Tuna Commission management area, north- west to French Polynesia (147°W, 5°S); and along of central eastern Pacific between 2− 8°S from 111-135°W (Díaz-Delgado et al. 2021).

In the eastern Pacific, the oceanic whitetip shark is caught on a variety of gear, including pelagic longline and purse seine gear targeting tunas and swordfish. While the range of the oceanic whitetip in the eastern Pacific is noted as extending as far north as southern California waters, based on the available data, the distribution of the species appears to be concentrated in areas farther south, and in more tropical waters (Young et al. 2017). Observer data of the West-Coast based U.S. fisheries further confirms this finding, with oceanic whitetip sharks generally not observed in the catches. For example, in the California/Oregon drift gillnet fishery, which targets swordfish and common thresher sharks and operates off the U.S. Pacific coast, observers recorded zero oceanic whitetip sharks in 8,698 sets conducted over the past 25 years (from 1990-2015) (Young et al. 2017).

From 2004 to 2016, approximately 32M–47M hooks were set annually in the Hawaii deep-set longline fishery and from 2017 to 2022, approximately 53M to 63M hooks were set annually (NMFS 2023h). In a 2023 opinion, future fishing effort was expected to increase from past fishing effort based on an increase in vessels entering the fishery. It was estimated that over a 5-year period, incidental take through capture of up to 181 giant manta rays, 10,589 oceanic whitetip sharks, and 14 Indo-West DPS scalloped hammerheads (NMFS 2023f). This fishery has a mandatory observer program, although percent observer coverage for fishing vessels often does not exceed 20% (NMFS 2023f; NMFS 2023g; NMFS 2024c). In 2022, a total of 942 (698 released alive) oceanic whitetip sharks and 50 (48 released alive) giant manta rays were reported as bycatch (NMFS 2023g). In 2023, a total of 904 (634 released alive) oceanic whitetip sharks and 6 (4 released alive) giant manta rays were reported as bycatch (NMFS 2023f). No other ESA-listed fish were reported as bycatch.

In a 2019 opinion, future fishing effort in the shallow-set longline fishery was expected to be consistent with recent fishing effort; 2,000 long line sets annually (approximately 500,000-1.8M

hooks annually) (NMFS 2019d). Mortality of 32 oceanic whitetip was estimated annually and hooking or entanglement of up to 13 (mean=7) giant manta rays with mortality of up to 4 giant manta rays in any given year (NMFS 2019d). There are at-sea observers aboard every shallow-set longline fishing trip. In 2022, a total of 108 (94 released alive) oceanic whitetip sharks and 6 (all 6 released alive) giant manta rays were reported as bycatch (NMFS 2023f). In 2023, a total of 26 (all 26 released alive) oceanic whitetip sharks and 2 (both released alive) giant manta rays were reported as bycatch (NMFS 2023f). No other ESA-listed fish were reported as bycatch.

Effective May 31, 2022, NMFS formalized a final rule for gear modifications to increase the survival of oceanic whitetip sharks captured as bycatch (87Fed. Reg. 25153). This final rule prohibits the use of steel wire leaders within 1 m of any hook deployed in the Hawaii deep-set longline fishery, and other gear modifications in the longline fisheries operating under the Fishery Ecosystem Plan for Pelagic Fisheries of the Western Pacific, including the Hawaii deep-set and shallow-set longline fisheries and American Samoa longline fishery. These measures are anticipated to increase their bycatch survival rate by 30%.

5.11 Aquaculture

Aquaculture supplies more than 50% of all seafood produced for human consumption globally.[21] Aquaculture systems are diverse, ranging from highly controlled land-based systems to open water cages that release wastes directly into the environment. Most of the aquaculture that currently operates in the US occurs on land, in tanks or ponds, and in the inshore or nearshore marine environment (Upton 2020). Species produced in the marine environment include seaweeds, bivalve molluscs, echinoderms, crustaceans, and finfish (Langan 2004).

Marine aquaculture is expected to expand in the U. S. EEZ due to increased demand for domestically grown seafood, coupled with improved technological capacity to farm in the open ocean. The National Offshore Aquaculture Act of 2005 (S. 1195) promoted offshore aquaculture development within the EEZ and established a permitting process that encourages private investment in aquaculture operations, demonstrations, and research. As a result of the May 2020 Executive Order on Promoting American Seafood Competitiveness and Economic Growth (85Fed. Reg. 28471), the Department of Commerce needs to identify geographic areas and evaluate locations suitable for commercial aquaculture to become Aquaculture Opportunity Areas. One of the areas considered is off southern California, which could support multiple aquaculture farm sites, including finfish, shellfish, and/or seaweed (NMFS 2020c). Currently, there is nearshore aquaculture in California; mussel farms in the Santa Barbara Channel and off Long Beach, with a permit pending for significant expansion of mussel farming off the coast of Ventura.[22] The first commercial-scale offshore aquaculture project in U.S. Federal waters has been proposed 4.5 statute mi (7.2 km) west of Mission Bay in San Diego, California.

[21] https://www.fisheries.noaa.gov/topic/aquaculture Accessed 3/3/2025

[22] https://caseagrant.ucsd.edu/our-work/discover-california-seafood/aquaculture-california#:~:text=Currently%2C%20aquaculture%20facilities%20that%20produce,with%20sturgeon%20farmed%20near%20Sacramento. Accessed 3/3/25.

Submersible sea cages would be deployed and the project could have potential impacts on marine biological resources and water quality. Mitigation would include the use of proper mesh size netting and the construction of the facilities away from known seal and sea lion haulout areas to minimize marine mammal interactions (U.S. Navy 2022).

Farming the sea is a part of Hawaii's rich oceanic heritage and the state has been at the forefront of aquaculture research and technology (HDOA 2018). Hawaii is the first state to successfully operate commercial open ocean aquaculture cages in nearshore (State) waters of the U.S. In 2011, Hawaii's total aquaculture sales were valued at $40 million, an increase from $10 million in 2010. Algae sales accounted for 63% of the value, ornamental category 6%, finfish 4%, and shellfish 1%, with the remaining 26% from sales of seedstock, broodstock and fingerlings.

Aquaculture has the potential to impact ESA-listed 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 (Llyod 2003; Clement 2013; Price and Morris 2013; Price et al. 2016; Price et al. 2017). Potential negative effects from finfish culture is enrichment of the water column with dissolved nutrients, resulting from the decomposition of uneaten feed, and from metabolic wastes produced by the fish (Langan 2004). Aquaculture gear can be similar to gear used in commercial fisheries, and therefore potential for similar impacts. Species of baleen whales are not evolved to echolocate and rely on visual and audio queues, which may put them at higher risk of entanglement (Llyod 2003). There are reports of cetacean interactions globally with aquaculture gear: humpback whales is Australia, Canada and Iceland, Bryde's whales in New Zealand, right whales in South Korea, Argentina, and the North Atlantic Ocean (Price et al. 2016). There are three known incidents involving leatherback sea turtles being entangled in mussel ropes in Notre Dame Bay, Newfoundland from 2009 through 2013 (Price et al. 2016). One leatherback was documented entangled in shellfish aquaculture gear in the Greater Atlantic Region of the U.S.. This animal was entangled in the vertical line associated with the anchoring system. Despite these reported incidents, collectivley the available data suggests that interactions and entanglements of protected species with aquaculture gear are rare (Price et al. 2016). This may indicate operations pose low risk of ESA-listed marine mammals but that is difficult to discern because gobally the number and density of aquaculture farms are relatively low, and thus the probability of interactions may be low.

It is generally thought that echolocating marine mammals (toothed whales, dolphins and porpoises) can effectively perceive mussel and fish farms and, in most cases, navigate through or around them (Llyod 2003; Markowitz et al. 2004). Aquaculture gear has shown the potential for behavioral effects (attraction or avoidance) on marine mammals based on a study of bottlenose dolphins around fish cages in Italy (reviewed in Callier et al. 2018).

Pinnipeds and sharks may be attracted to fish culture operations and try to find a way to get inside the nets/cages to opportunistically feed. There was a case of a monk seal mortality associated with fish culture in 2017, suspected to have gotten stuck inside a holding pen, but no

reports of serious injuruy or mortality have occurred since then (Carretta et al. 2025). The company growing fish in the open water cages in Hawaii has partnered with NMFS to study monk seal interactions with the culture operations, using underwater video cameras to collect footage and artificial intelligence to analyze the data.[23] The footage and data is being used to make changes to gear and operations to reduce the potential for negative interactions.

5.12 Ongoing Military Training and Testing Activities in the Action Area

Military readiness activities have been conducted in the HCTT action area for decades. Ongoing U.S. Navy training and testing activities in the action area are discussed here as part of the environmental baseline. The Navy categorizes training exercises and testing activities into functional warfare areas called primary mission areas. Training exercises fall into primary mission areas: air warfare, surface warfare, amphibious warfare expeditionary warfare, anti-submarine warfare, mine warfare, and electronic warfare. Details regarding primary warfare mission activities can be found in the *Hawaii-Southern California Training and Testing Phase III 2018 Final Environmental Impact Statement/Overseas Environmental Impact Statement (FEIS/OEIS).* Conservation measures are employed to mitigate activities that are likely to adversely affect ESA listed species, such as visual observers and implementing protective geographic mitigation areas.

5.12.1 Summary of Hawaii-Southern California Training and Testing Phase III Biological Opinion

In December 2018, we completed a section 7 consultation on U.S. Navy HSTT Phase III Training and Testing Activities. We consulted with the Navy and 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 for HSTT activities. On October 26, 2023, reinitiated formal the HSTT Phase III consultation and we issued a revised opinion and ITS on June 3, 2024 (NMFS 2024f). The following levels of incidental take are expected annually from acoustic stressors in the form of harassment (TTS or behavioral) and harm (PTS) for HSTT Phase III training and testing activities in the action area, by species:

- Blue Whale: 1,229 TTS, 806 behavioral, 1 PTS
- Fin Whale: 1,417 TTS, 856 behavioral, 1 PTS
- Mexico DPS Humpback Whale: 920 TTS, 198 behavioral, 1 PTS
- Central America DPS Humpback Whale: 594 TTS, 282 behavioral
- Sei Whale: 173 TTS, 73 behavioral
- Western North Pacific DPS Gray Whale: 4 TTS, 2 behavioral
- Sperm Whale: 86 TTS, 4,903 behavioral
- MHI IFKW*:* 17 TTS, 572 behavioral
- Hawaiian Monk Seal: 62 TTS, 143 behavioral, 1 PTS
- Guadalupe Fur Seal: 15 TTS, 1,442 behavioral

[23] https://www.fisheries.noaa.gov/feature-story/ai-meets-aquaculture-study-hawaiian-monk-seal-interactions-net-pens Accessed 3/4/25.

- Central North Pacific DPS Green Sea Turtle: 20 TTS, 1,831 behavioral, 7 PTS, 1 non-auditory injury
- Hawksbill Sea Turtle: 21 behavioral
- Olive Ridley Sea Turtle: 96 behavioral
- North Pacific DPS Loggerhead Sea Turtle: 182 behavioral
- Leatherback Sea Turtle: 193 behavioral

The ITS for the HSTT Phase III reinitiated opinion anticipated up to two large whale vessel strikes over the final 2.5 year period of the MMPA rule, and each subsequent 2.5 year period into the reasonably foreseeable future. During each 2.5 year period, up to one blue whale, one humpback whale – Mexico DPS, and one sei whale vessel strike is exempted and up to two fin whale vessel strikes are exempted. The total number of vessel strikes exempted of these species combined is two during each 2.5 year period, and all strikes are expected to result in mortality.

Vessel strikes are also anticipated to result in non-lethal injuries (injuries) and mortalities for the following sea turtles over a 7-year period:

- Central North Pacific DPS Green Sea Turtle: 56 mortalities, 21 injuries
- Eastern Pacific DPS Green Sea Turtle: 3 mortalities, 1 injury
- Hawksbill Sea Turtle: 3 mortalities, 1 injury
- Olive Ridley Sea Turtle: 1 mortality, 1 injury

5.12.2 Surveillance Towed Array Sensor System Low Frequency Active Sonar

The Navy's SURTASS sonar system has a vertical line array of 18 elements operating between 100 and 500 Hz. The typical low-frequency active sonar signal is not a constant tone but consists of various waveforms that vary in frequency and duration. A complete sequence of sound transmissions (waveforms) is referred to as a wavetrain (also known as a ping). These wavetrains last between 6 and 100 seconds, with an average length of 60 seconds. Within each wavetrain, a variety of signal types can be used, including continuous wave and frequency-modulated signals. The duration of each continuous-frequency sound transmission within a wavetrain is no longer than 10 seconds. The interval between transmissions varies between 6 and 15 minutes. SURTASS low-frequency active sonar has a coherent low frequency signal with a duty cycle of less than 20%. The NMFS 2019 opinion (NMFS 2019c) estimated the following levels of incidental takes by harassment (TTS and behavioral) from exposure to Navy SURTASS LFA sonar over a 7-year period during training and testing activities in a broad action area, for the following ESA-listed species that occur within the HCTT action area:

- blue whale: 763
- fin whale: 20,871
- sei whale: 25, 944
- Western North Pacific DPS gray whale: 3
- sperm whale: 12,483
- MHI IFKW*:* 7
- Hawaiian monk seal: 79

The 2019 opinion also exempted incidental take due to the potential for harassment (TTS and behavioral) for the following sea turtle species that overlap with HCTT action area: Central North Pacific DPS green, hawksbill, leatherback, olive ridley, and North Pacific DPS loggerhead. No instances of PTS or mortality was expected for sea turtles. The number of sea turtles harassed from each species was not estimated because there was not enough information on those species occurring in the broad SURTASS action area. Take will be considered exceeded if activity levels (i.e., the number of LFA sonar transmission hours) as proposed are exceeded.

5.12.3 Point Mugu Sea Range

During ESA section 7 consultation with the Navy for testing and training activities on the PMSR, for 2022 to 2029, it was determined that the use of explosives would likely result in adverse effects to ESA-listed species (NMFS 2022b).

The following estimated takes, in the form of harassment (TTS or behavioral) and harm (PTS), are expected to occur annually in the PMSR action area for the following listed species:

- blue whale: 4 TTS, 7 behavioral
- fin whale: 7 TTS, 14 behavioral, 1 PTS
- Mexico DPS humpback whale: 4 TTS, 7 behavioral
- Central America DPS humpback whale: 1 behavioral
- sperm whale: 1 TTS, 1 behavioral
- Guadalupe fur seal: 1 TTS, 1behavioral
- leatherback sea turtle: 1 behavioral

5.13 Vessel Approach

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; Blane and Jaakson 1994; Evans et al. 1994).

Studies investigating the behavioral responses of cetaceans to vessels suggest that individual whales experience stress responses to approaching vessels. While this type of stimulus is often stressful, the fitness consequences of this stress on individual whales remains unknown (Baker et al. 1983b; Baker and Herman 1987). Beale and Monaghan (2004) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches.

Baker *et al.* (1983b) described two responses of whales to vessels: (1) *horizontal avoidance* of vessels 2,000 to 4,000 m away characterized by faster swimming and fewer long dives; and (2) *vertical avoidance* of vessels from 0 to 2,000 m away during which whales swam more slowly, but spent more time submerged. Watkins *et al.* (1981) found that both fin and humpback whales

appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 km from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels (Bauer 1986). Bauer (1986) and Bauer and Herman (1986) noted changes in humpback whale respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels.

Studies of other baleen whales, specifically bowhead and gray whales, document similar patterns of behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Malme et al. 1983; Richardson et al. 1985). For example, studies of bowhead whales revealed that they orient themselves in relation to a vessel when the engine is on, and exhibit significant avoidance responses when the vessel's engine is turned on even at a distance of about 900 m (3,000 ft). Jahoda *et al.* (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They found that close vessel approaches caused the whales to stop feeding and swim away from the approaching vessel. The fin whales studied also tended to reduce the time they spent at the surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. Whales that had been disturbed while feeding remained disturbed for hours after the exposure ended.

Whale Watching

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, whale watching (either commercially or on a private vessel) has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close and strikes a whale (New et al. 2015). Another concern is that preferred habitats may be abandoned if disturbance levels from whale watch boats are too high.

Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS 2006). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity. Oliveira et al. (2022) tagged 24 sperm whales off the Azores to evaluate changes in movement and acoustic parameters related to foraging effort and dives, locomotion effort, and non-foraging behavior compared during periods of exposure and non-exposure to whale watching vessels. While no significant changes were found in foraging effort and overall dive behavior between non-exposure and exposure periods, variation in body pitch during non-foraging periods increased during exposure. These results suggest increased movements during resting activity, which may carry energetic costs to whales and impact their health and fitness (Oliveira et al. 2022). Disturbance by whale

watch vessels has also been noted to cause newborn calves to separate briefly from their mothers' sides, which leads to greater energy expenditures by the calves (NMFS 2006).

Au and Green (2000) concluded that it is unlikely that the levels of sounds produced by whale watching boats in Hawaii would have any grave effects on the auditory system of humpback whales. Rey-Baquero et al. (2021) modeled humpback whales' communication space during ambient noise and with 1-2 boats traveling slowly. The most common peak frequency of this stock's song (350 Hz) was used in the model, and, along with a whale's location along the coast, informed calculations of transmission loss. They found that the infrequent addition of whale watching boat noise could temporarily reduce the "very audible area" (> 10 dB SNR) in humpback whale song's commonly used peak frequency (350 Hz) by 63%. Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from this activity (NMFS 2006) (New et al. 2015).

By regulation, humpback whales cannot be approached closer than 100 yd (90 m) by vessels in Hawaiian waters (50 CFR 224.103). The only exception to these approach restrictions is for researchers who hold a scientific research permit authorized by NMFS. For all other cetaceans and for monk seals the recommended distance for observation is 50 yd when the animal is on land or in the water. Other guidelines have been issued by NMFS to minimize the impacts of wildlife viewing on marine mammals, including maximum vessel speeds, proper vessel positioning, limiting noise levels, and the use of extra caution in the vicinity of mothers and their young.

In Hawaii, most of the whale watching industry is based around humpback whales which winter in the islands from mid-December to the end of April (Hoyt 2001). Maui is the primary location for boat-based whale watching, but whale watching operations are located at most major harbors around the state. The whale watching industry in Hawaii contributes approximately $20 million in total revenues per year.

In the Southern California portion of the action area, whale watching companies offer blue whale tours that leave from San Diego Bay from about mid-June through September. Whale watching around the Channel Islands is a popular activity, particularly from March through May during the annual gray whale northward migration. In Santa Barbara and Ventura Counties, whale watching trips are offered year round, with the southern migration of gray whales between late December and February and the northern migration between February and mid-May (O'Connor et al. 2009). The peak of activity is between January and March and particularly during the northern migration, as the whales travel much closer to the shore and often more slowly as the mothers are travelling with calves. Blue and humpback whale watching occurs between June and November, although the highest concentration of whales usually occurs between June and September. The Santa Barbara Channel has very reliable sightings of blue whales during this time (O'Connor et al. 2009).

Farther north off California, much of the whale watching originates from San Francisco Bay with whale watching concentrated in the Greater Farallones National Marine Sanctuary and Monterey

Bay. In part due to weather, offshore whale watching generally occurs from April through November. Some of the more commonly observed whales within these areas are humpback whales, gray whales, and blue whales (NOAA National Centers for Coastal Ocean Science 2007; Burrows et al. 2012; Dransfield et al. 2014).

We have no information regarding the specific effects of whale watching operations within the action area. We anticipate that at least some short-term effects from whale watching, as described above, are affecting ESA listed whales within the action area, although the regulations and mitigation measures in place likely reduce those effects to some extent.

5.14 Vessel Strike

Marine habitats occupied by ESA-listed species often feature both heavy commercial and recreational vessel traffic. Vessel strikes are a threat to several taxa of large air breathing marine vertebrates, including whales and sea turtles. The International Whaling Commission noted that human-induced mortality caused by vessel strikes can be an impediment to cetacean population growth (IWC 2017). Most whales killed by vessel strike likely end up sinking rather than washing up on shore. It is estimated that only 17% of vessel strikes of whales are actually detected (Kraus et al. 2005). Therefore, it is likely that the number of documented cetacean mortalities related to vessel strikes is much lower than the actual number of mortalities associated with vessel strikes.

Various types and sizes of vessels have been involved in ship strikes with large whales, including container/cargo ships/freighters, tankers, steamships, U.S. Coast Guard vessels, Navy vessels, cruise ships, ferries, recreational vessels, fishing vessels, whale-watching vessels, and other vessels (Jensen and Silber 2004). The majority of vessel strikes of large whales occur when vessels are traveling at speeds greater than approximately ten knots, with faster vessels, and especially if they are large vessels (80 m or greater), being more likely to cause serious injury or death (Laist et al. 2001; Jensen and Silber 2004; Vanderlaan and Taggart 2007; Conn and Silber 2013). Injury is generally caused by the rotating propeller blades, but blunt injury from direct impact with the hull also occurs. Injuries to whales killed by vessel strikes include huge slashes, cuts, broken vertebrae, decapitation, and animals cut in half (Carillo and Ritter 2008). Conservation strategies try to minimize ship strikes by avoiding higher risk areas, such as re-routing shipping lanes, and implementing vessel speed limits in areas where collisions are known to occur.

The west coast of the U.S. has some of the heaviest ship traffic associated with some of the largest ports in the country, including Los Angeles/Long Beach, San Francisco, Seattle, and the Columbia River. Blue, fin, humpback, and gray whales are the most vulnerable species to ship strikes because they migrate along the coast and utilize coastal areas for feeding. NMFS declared an Unusual Mortality Event (UME) for the fall of 2007, because at least five blue whales were struck and killed by vessels. The magnitude of this threat for large whales populations along the U.S. West Coast could be considerably larger than indicated based on reported incidents due to the unknown number of vessel strikes that go undocumented (NMFS 2011). For example, Rockwood et al. (2017) estimated ship strike mortality of whales using an encounter theory

model that considered whale density, vessel traffic characteristics, and whale movement patterns. Results indicated that 74% of blue whale, 82% of humpback whale, and 65% of fin whale known vessel strike mortalities occur in the shipping lanes associated with the ports of San Francisco and Los Angeles/Long Beach.

The NMFS Marine Mammal Health and Stranding Response Program recorded 57 whale strikes off California between 2017 and May 2024(Greenman 2024). Twenty of these strikes were off Southern California and 37 off Northern California. The majority struck were gray whales (51%), followed by fin whales (18%) and humpback whales (12%); see Figure 66 below.

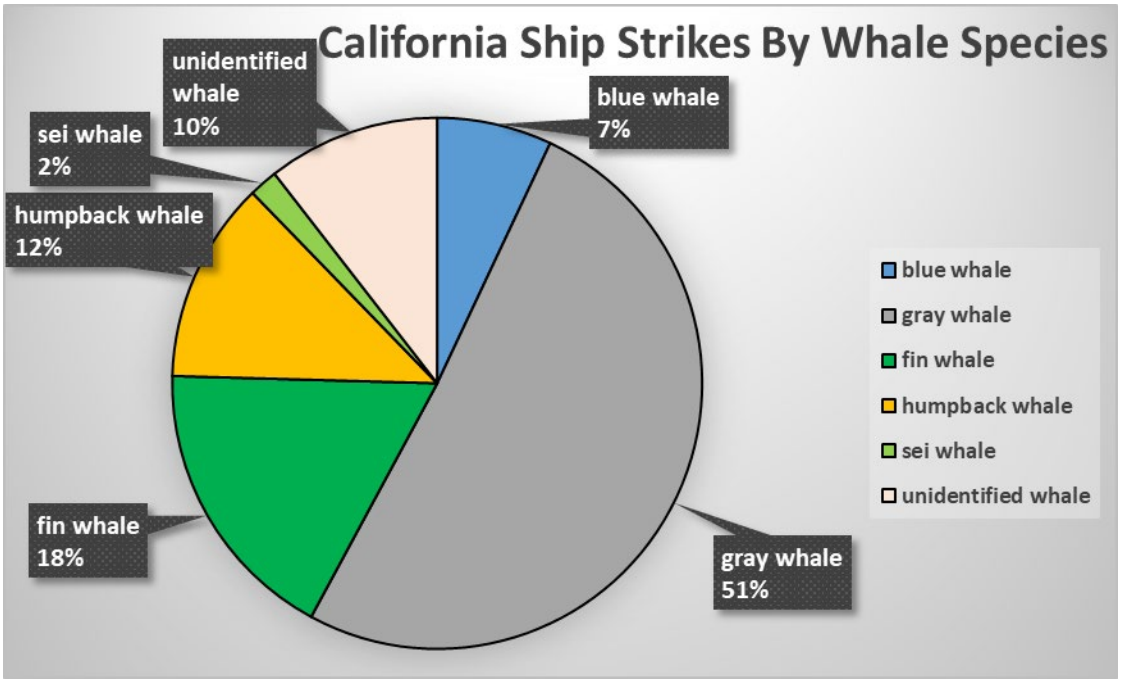


Figure 66. California Ship Strikes by Whale Species (2017–2024)

The Rockwood et al. (2017) model estimated 18 blue whales, 43 fin whales, and 22 humpback[24] whales are fatally struck by vessels on an annual basis along the U.S. West Coast. These estimates include only the seasonal period of July-November when large whales are most likely to be present and based on an assumption of a moderate level of vessel avoidance. A comparison of the annual average vessel strikes of 0.6 blue whales per year, observed over the period 2017-2021 (Carretta et al. 2025), versus the estimated 18 vessel strikes per year indicates that the rate of detection for blue whale vessel strikes is approximately 3%. Similarly, the average observed annual mortality and serious injury during 2017-2021 due to vessel strikes is 1.6 fin whales per year (Carretta et al. 2025) versus the estimated annual deaths of 43, suggests a reporting rate of 3.7%. The mean annual observed vessel strike cases of humpback whales per year observed off

[24] The number of vessel strikes attributable to each humpback whale DPS is unknown.

the U.S. West Coast waters during 2016-2020 was 2.6 deaths/serious injuries (Carretta 2022), compared to the estimated value of 22 per year, implies that vessel strike counts from opportunistic observations represent 11% of overall incidents.

No vessel strikes or serious injuries have been documented during the most-recent available 5-yr period (2017-2021) of either sei whales or sperm whales (Carretta et al. 2023b). We are not aware of any modelled estimates of annual strikes for those species in this area and therefore unable to evaluate how many incidences may occur unobserved.

An Australian naval vessel fatally struck two fin whales off southern California in May 2021. The two carcasses had become lodged on the hull of the vessel and were dislodged when the vessel pulled into a pier in San Diego, California (Associated Press 2021). From 2021 through August of 2025, the U.S. Navy reported striking five large whales of unknown species within the SOCAL RC. These five vessel strikes fall within the scope of the Navy's action for the 2018 HSTT Phase III consultation. Military vessel strikes of large whales in the HSTT action area are discussed in more detail in our marine mammal vessel strike effects analysis below (Section 6.1.3).

In March 2024, a dead fin whale was discovered off of Pier 10 in Naval Station San Diego within the Navy's security barrier. The security barrier, which consists of a series of connected floating sections, is intended to discourage unauthorized boat entry to the piers. The necropsy indicated that vessel strike was the most likely cause of death. Given the whale's location, this could have been the result of a strike by a U.S. Navy vessel. However, based on the information available, including information provided by the Navy regarding vessel activity during that timeframe (B. Colbert, U.S. Navy Office of the Chief of Naval Operations, personal communication email titled "Fin Whale Found at Naval Station San Diego" sent to J. Viezbicke, NMFS California Stranding Network Coordinator, April 2, 2024), we cannot determine whether this whale was struck by a Navy vessel during HSTT activities or was struck by a commercial or other vessel and drifted into the Navy pier area. On September 12, 2025, a Navy Lookout reported a whale carcass adjacent to the port quarter of a transiting vessel in the PMSR. There were no observations of whales or other indicators prior to the discovery, and sailors onboard did not feel a shudder or other physical indicator of a strike. Based on the information provided, we cannot conclude that this stranding was the result of a vessel strike, let alone the result Navy HCTT vessel strike. Given the context of these two strandings (i.e., March 2024 and September 2025), and associated high degree of uncertainty, we have not included these events as records of vessel strikes that fall within the scope of the Navy's HSTT Phase III or PMSR actions.

Impact from a boat hull or outboard motor, or cuts from a propeller can kill or severely injure turtles. Many recovered turtles display injuries that appear to result from interactions with vessels and their associated propulsion systems (Work et al. 2010). Turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases (Hazel et al. 2007). Results from a study by Hazel et al. (2007) suggest that green turtles cannot consistently avoid being struck by vessels moving at relatively

moderate speeds (i.e., greater than 4 km per hour). Vessel strikes have been identified as one of the important mortality factors in several near shore turtle habitats worldwide (Denkinger et al. 2013).

The NMFS Southwest Fisheries Science Center maintains a database of green sea turtle strandings in San Diego Bay. From 2017-2021, a total of 38 green sea turtle strandings were reported in the bay, 11 of which showed signs of a vessel strike (Erin LaCasella, NMFS Southwest Fisheries Science Center, unpublished data provided to the Navy on July 27, 2022). Since it is possible that a vessel strike can occur post-mortem, vessel strike may not have been the proximal cause of death in all 11 stranded turtles exhibiting vessel strike wounds. In general, sea turtle strandings data represent a minimum mortality rate since the majority of carcasses never reach shore and are either undetected or unreported. Previous studies suggest that the stranding probability of a sea turtle that dies at sea usually does not exceed 10-20% of total at sea mortality, even in nearshore waters (Epperly et al. 1996; Hart et al. 2006; Mancini et al. 2012).

Vessel strikes were identified as a source of mortality for green sea turtles in Hawaii waters, although reported incidence rates among stranded turtles are not as high as in the southeastern United States. Chaloupka et al. (2008) reported that 2.5% of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by vessel strike. From 2008 through 2018, 176 stranded green sea turtles in Hawaii were documented with boat strike injuries, representing approximately 5% of all sea turtle strandings over this time period (Kelly 2020). Brunson et al. (2022) analyzed hawksbill sea turtle strandings data collected between 1984 and 2018 in the Hawaiian Islands. Vessel strike was determined to be the cause of death for three out of the 72 (i.e., 4.2%) strandings with associated impacts.

5.15 Invasive Species

The introduction of non-native species is considered one of primary threats to at-risk species (Anttila et al. 1998; Wilcove and Chen 1998; Pimentel et al. 2004; Dueñas et al. 2018; Noss et al. 2020; Costante 2021). 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 database; invasive species were the only cited cause in 20% of those cases. Invasive species consistently rank as one of the top threats to the world's oceans (Raaymakers and R. Hilliard 2002; Raaymakers 2003; Terdalkar et al. 2005; Pughiuc 2010; Hughes et al. 2020; Occhipinti-Ambrogi 2021; Salimi et al. 2021).

When non-native plants and animals are introduced into habitats where they do not naturally occur, they can have significant impacts on ecosystems and native fauna and flora. Aquatic non-native species can be introduced through a variety of ways, including but not limited to, aquarium and pet trades, recreation, ballast water discharges from ocean-going ships, hull fouling, stocks used for fishery enhancement or escape from aquaculture. 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).

Non-native species can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may also displace food

items preferred by native predators, disrupting the natural food web. An example of indirect predatory effects caused by an invasive species is the European green crab, which has invaded both the east and west coasts of the U.S., resulting in trophic scale effects to ecosystems in both regions (Grosholz and Ruiz 1996). Invasive plants can cause widespread habitat alteration, including native plant displacement, changes in benthic and pelagic animal communities, altered sediment deposition, altered sediment characteristics, and shifts in chemical processes such as nutrient cycling (Grout et al. 1997; Wigand et al. 1997; Ruiz et al. 1999). Introduced seaweeds can alter habitat by colonizing previously unvegetated areas, while algae can form extensive mats that exclude most native taxa, dramatically reducing habitat complexity and the ecosystem services provided by it (Wallentinus and Nyberg 2007). Invasive algae can alter native habitats through a variety of impacts including trapping sediment, reducing the number of suspended particles that reach the benthos for benthic suspension and deposit feeders, reducing light availability, and adverse impacts to foraging for a variety of animals (Gribsholt and Kristensen 2002; Britton-Simmons 2004; Levi and Francour 2004; Sanchez et al. 2005). Pathogens and species with toxic effects not only have direct effects on listed species, but also may affect essential critical habitat features or indirectly affect the species through ecosystem-mediated impacts. There are a number of non-native species that can expel toxins at low levels, becoming problematic when their population grows very large, releasing cumulatively large amounts of toxins.

As of 2013 there were 54 documented marine invasive species in San Diego Bay including tunicates (nine species), amphipods (eight species), polychaetes (six species), moss animals (six species), molluscs (five species), and isopods (four species) (Navy 2013b). Several of these invasions have resulted in ecosystem level effects. Dense mats of the Japanese mussel (*Musculista senhousia*) can crowd out native clams and inhibit growth of native eelgrass (Reusch and Williams 1998; Reusch and Williams 1999). *Caulerpa taxifolia, Caulerpa prolifera* and *Codium fragile tomentosoide* are invasive green algal species found in Southern California (Dobroski et al. 2015; Gagnon et al. 2015). *Sargassum muticum* (Japanese wireweed) and *Sargassum horneri* (devil weed) are invasive brown algal species found in Southern California (Dobroski et al. 2015; Marks et al. 2015). *Undaria pinnatifida* (or wakame), an edible seaweed native to Japan, has spread rapidly along the California coast (Kaplanis et al. 2016). The ecosystem level changes produced by these invasive species could have detrimental impacts on ESA-listed green sea turtle habitat and prey, although no studies are known to have specifically addressed this issue.

There are a total of 333 non-native species, and another 130 cryptogenic species (i.e., unknown origin), documented as part of the marine and estuarine biota of the six largest Hawaiian islands from Kauai to Hawaii (Carlton and Eldredge 2015). The greatest proportion of non-native and cryptogenic species are found in the majors harbors of Oahu, which receive the large majority of all vessel traffic in the Hawaiian Islands (Coles and Eldredge 2002). Approximately 20% of the benthic algae, fish, and macroinvertebrate species found these harbors are either non-native or cryptogenic. Algal species have become nuisance invaders of many Hawaiian reefs (Smith et al. 2002). One study has shown that green sea turtles have modified their feeding behavior over the

past 36 years in the greater Kāneʻohe Bay to include the increasing abundance of non-native algae growing, becoming more than half of their diet (Russell and Balazs 2015). With the exception of Kāneʻohe Bay, the largest embayment in Hawaii with a history of urban impact, few nonindigenous fish or invertebrates have been detected on Hawaiian reefs (Coles and Eldredge 2002). Since at least 2001, non-indigenous species of octocorals have been identified in Hawaiian waters (Kahng and Grigg 2005; Kahng 2006). In May 2020, three new nonindigenous species of coral (*Montipora spp.*) and an anemone (*Anemonia manjano*) were confirmed in Kāneʻohe Bay (Division of Aquatic Resources 2021). Even more recently, colonies of the invasive octocorals *Unomia stolonifera* and *Capnella cf. spicata* were detected in the southern region of the Main Channel at Joint Base Pearl Harbor-Hickam, Hawaii, in the vicinity of Bishop Point (Miller et al. 2023). Changes in coral or other benthic communities and the related inhabitant fish species could be impacting ESA-listed sea turtles and Hawaiian monk seals in Hawaii through a variety of ways, although there is a lack of available information to evaluate this.

5.16 Parasites and Diseases

Fibropapillomatosis is a neoplastic disease that can negatively impact ESA-listed sea turtle populations. Fibropapillomatosis 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). Fibropapillomatosis 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). Prevalence rates as high as 45 to 50% have been reported within some California green turtle populations (Jones et al. 2015; Hargrove et al. 2016). Fibropapillomatosis primarily affects medium-sized immature turtles in coastal foraging pastures. Disease prevalence rose in the 1980s, spreading substantially in both Florida and Hawaii in green sea turtle populations (Work and Balazs 2013; McCorkle 2016).

Fibropapillomatosis is characterized by both internal and external tumorous growths, which can range in size from very small to extremely large. Large tumors can interfere with feeding and essential behaviors, and tumors on the eyes can cause permanent blindness (Foley et al. 2005). Renan de Deus Santos et al. (2017) assessed stress responses (corticosterone, glucose, lactate, and hematocrit) to capture and handling in green sea turtles with different fibropapillomatosis severity levels. Their findings suggest that moderate fibropapillomatosis severity may affect a turtle's ability to adequately feed themselves (as evidenced by poor body condition), and advanced-stage fibropapillomatosis severity may result in an impaired corticosterone response. Expression of fibropapillomatosis differs across ocean basins and to some degree within basins (Hargrove et al. 2016). In Hawaii, tumors have been reported on the internal organs of green sea turtles and oral tumors are common and often severe (Hargrove et al. 2016).

While fibropapillomatosis can result in reduced individual fitness and survival, documented mortality rates in Hawaii have been low and do not seem to impede population growth in places that are intensively monitored (e.g., Florida and Hawaii) as evidenced by increasing nesting trends (Hargrove et al. 2016). However, fibropapillomatosis cannot be discounted as a potential

threat to sea turtle populations (particularly green turtles) as the distribution, prevalence rate, severity, and environmental co-factors associated with the disease have the capacity to change over time (Jones et al. 2015).

Environmental factors likely play a role in the development of fibropapillomatosis. Most sites with a high frequency of fibropapillomatosis tumors are areas with some degree of water quality degradation resulting from altered watersheds (Hargrove et al. 2016). Despite there being a strong positive correlation between the prevalence of fibropapillomatosis in green turtle populations and areas with degraded water quality, it is difficult to identify one specific causal contaminant or a combination of such working synergistically to lead to fibropapillomatosis formation. Results from a study by Yan et al. (2018) suggest that PCBs may play a role in fibropapillomatosis in Hawaiian green turtles.

Infectious diseases and parasites are a threat to many cetacean populations worldwide. Genital papillomatosis has been observed in sperm whales from Iceland (Lambertsen et al. 1987). Jauniaux et al. (2000) reported evidence for morbillivirus infection in fin whales stranded on the Belgian and French coastlines. Cetacean morbilliviruses and papillomaviruses as well as *Brucella spp*. and *Toxoplasma gondii* are thought to interfere with population abundance by inducing high mortalities, lowering reproductive success, or by synergistically increasing the virulence of other diseases (Van Bressem et al. 1999). Anthropogenic environmental changes may increase the prevalence and severity of infectious illnesses and disease in cetaceans. A high prevalence of traumatic injuries or even minor skin lacerations from other stressors (e.g., vessel strike, fisheries interactions), in combination with a compromised immune system create ideal targets for opportunistic pathogens.

Cetaceans have evolved with a group of parasites belonging to the genus *Crassicauda* (order Spirurida) (Lambertsen 1992). 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. Infections caused by *Crassicauda boopis* are especially pathogenic. This giant worm infects blue whales, humpback whales, and fin whales (Lambertsen 1992).

Several species of helminth parasites were found in 100 blue whale fecal samples, collected during the winter in the Gulf of California from 1993-2014 (Flores-Cascante et al. 2019), indicating gastrointestinal tract infections. Additionally, *Diphyllobothrium* sp. eggs were also identified in the blue whale feces (Flores-Cascante et al. 2019). However, parasites are not known to have population-level effects for any blue whale population (NMFS 2020d).

The potential population-level impact of infectious disease on Hawaiian monk seals could be severe given their critically endangered status, very low genetic diversity, and that this population has not been previously exposed to many diseases due to the isolation of the Hawaiian Archipelago (PIFSC 2018). Monk seals in the main Hawaiian Islands are often in close proximity to areas of human activity, domestic and feral animals, and agricultural areas, thus increasing the probability of infectious disease transmission. Infectious diseases that pose a risk to the monk seal population include distemper viruses, West Nile virus, *Leptospira spp*., and *Toxoplasma gondii* (PIFSC 2018). Risk factors for Hawaiian monk seals include cetaceans and

non-native pinniped species that carry morbillivirus into Hawaiian waters and interactions between monk seals and infected dogs.

Toxoplasmosis was first identified infecting a wild Hawaiian monk seal carcass examined in 2004 with disseminated disease and intra- and extracellular tachyzoites and tissue cysts in affected organs (Honnold et al. 2005). Barbieri et al. (2016) reported seven additional cases (eight total) and two suspect cases of protozoal-related mortality in Hawaiian monk seals between 2001 and 2015, including the first record of vertical transmission in this species. *Toxoplasma gondii* was the predominant apicomplexan parasite identified and was associated with 100% of confirmed protozoal-related mortalities (n = 8), and 50% of suspected cases (Barbieri et al. 2016). During the past five years of available data (2017-2021) seven monk seal deaths (representing a minimum average of 1.4 deaths per year) have been directly attributed to toxoplasmosis (Carretta et al. 2025). Five of the seven deaths involved female seals. The number of deaths from this pathogen are likely underrepresented, given that more seals disappear each year than are found dead and examined (Harting et al. 2021), and the potential for chronic infections remains poorly understood in this species. Felids are the definitive host for *T. gondii*, it can only reproduce in cats and the parasite is shed from cat feces, therefore, the threat of toxoplasmosis is a growing concern given the increasing overlap between humans, cats, and Hawaiian monk seals (Carretta et al. 2025). Furthermore, *T. gondii* can be maternally transmitted to the fetus. Barbieri et al. (2016) includes the report of the first known case of transplacental transmission of this disease in monk seals.

Although the pathogen has not been associated with phocid mortality in the North Pacific to date, morbilliviruses have caused mass die offs of wild phocid populations in other parts of the world (PIFSC 2017). In 2016, NOAA developed the Hawaiian Monk Seal Vaccination Research and Response Plan to proactively address the threat of infectious diseases in this population, particularly for morbillivirus and West Nile virus infections.

Studies of Guadalupe fur seals stranding off the coast of California have reported finding hemorrhagic gastroenteritis, nematodes, cestodes (Gerber et al. 1993), septicemia, and bacterial pneumonia (Hanni et al. 1997a) in stranded animals. In addition, a parasite (*Parafilaroides decorus*) commonly found in California sea lions was documented for the first time in 2015 in a male yearling Guadalupe fur seal that stranded in Santa Cruz, California. Guadalupe fur seals have been diagnosed with other parasites, including *Toxoplasma gondii*, *Sarcocystis neurona*, and gastrointestinal parasites. The most common gastrointestinal helminths were tapeworms, nematodes/ascarids, hookworms, and anthocephalans. The diatom genus *Pseudo-nitzschia* produces the neurotoxin domoic acid. While domoic acid intoxication is not well known in Guadalupe fur seals, domoic acid has been detected in market squid (*Loligo opalescens*) and Humboldt squid (*Dosidicus gigas*), two species Guadalupe fur seals prey upon (reviewed in McCue 2021).

There is also no information to indicate that disease is a factor affecting populations of scalloped hammerhead sharks, oceanic whitetip sharks, or giant manta rays (Miller et al. 2014; Miller and Klimovich 2017; Young et al. 2017). Most elasmobranch species likely carry a range of external parasites including cestodes, nematodes, leeches, copepods, and amphipods but there are no

studies suggesting parasites are negatively affecting the fitness or survival of these species (Miller et al. 2014; Miller and Klimovich 2017; Young et al. 2017). At least some oceanic whitetip sharks are infected with highly pathogenic *Vibrio harveyi*. This bacterium is known to cause deep dermal lesions, gastro-enteritis, eye lesions, infectious necrotizing enteritis, vasculitis, and skin ulcers in marine vertebrates (Austin and Zhang 2006). *Vibrio harveyi* is considered to be more serious in immunocompromised hosts, and therefore may act synergistically with the high pollutant loads that oceanic whitetip sharks potentially experience to create an increased threat to the species (Young et al. 2017). However, there is no additional information available regarding the magnitude of impact these parasites may have on the health of oceanic whitetip populations (Young et al. 2017).

5.17 Scientific Research and Permits

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. As required by regulation, 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. Section 10(a)(1)(A) permits are also issued to research facilities and educational display facilities for the captive research and educational display of ESA-listed species.

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 ESA 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 2019a). Eleven new permits were issued under the cetacean research programmatic in 2023. Thirty seven permit holders conducted research, a total of 4,693 takes (only 3.4% of what was authorized) were reported for ESA-listed cetaceans. The majority (96%) were for Level B harassment and 4% were for Level A harassment, including 131 biopsy takes and 57 tagging takes (NMFS 2024a). 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 2019a).

Between 2009 and 2020, 26 research takes of Guadalupe fur seals were reported to the Permits Division. These included takes from vessel survey disturbance, sample collection, capture, morphometrics, and tagging. Hawaiian monk seal research and enhancement activities conducted by the Pacific Islands Fisheries Science Center from 2019 to 2024 include the following authorized procedures (not comprehensive): photo and video, manned and unmanned aerial surveys, capture and handling, sedation, tagging, marking, sampling (e.g. blood, blubber) and health surveys, vaccinations, disentanglements, translocations (e.g. for pup safety or dangerous adult male removals), humane euthanasia for moribund seals, and necropsies.

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. Mortality is rarely authorized by the Permits Division in sea turtle research permits and no lethal take was authorized for sea turtle research in the Pacific Ocean basin since 2007-2017. 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 (NMFS 2017b). The sea turtle research programmatic consultation allows for the authorization of up to the following number of sea turtle mortalities within the Pacific Ocean basin every ten years: nine green sea turtles (Central West Pacific, Central South Pacific, Central North Pacific, East Pacific DPSs combined); ten hawksbill; two leatherback; 12 loggerhead (North Pacific DPS); and eight olive ridely (NMFS 2017b). This programmatic also includes an ITS that allows for one green sturgeon Southern DPS lethal take every ten years and one scalloped hammerhead Eastern Pacific DPS lethal take every ten years. Through 2023, no sea turtle mortalities have been reported or any incidental take of other protected species by researchers operating in the Pacific Ocean since the programmatic biological opinion took effect in 2018. The percentage of takes used compared to what was authorized ranged from 0.8 to 21.8% by species in the Pacific, the majority of takes were green sea turtles (126) followed by loggerheads (38) (NMFS 2025b).

5.18 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 species considered in this consultation. Some of these stressors result in mortality or serious injury to individual animals (e.g., vessel strike, whaling, bycatch and entanglement), whereas others result in non-lethal impacts (e.g., vessel noise or whale watching) or more indirect impacts (e.g., a fishery that impacts prey availability).

We consider the best indicator of the environmental baseline on ESA-listed resources to be the status and trends of those species. As noted in Section 4.2, 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 affecting 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 limiting their recovery. However, it is also possible that their populations are at such low levels (e.g., due to historic hunting, fishing, harvesting, 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 4.2 Status of the Species and Critical Habitat Likely to be Adversely Affected by the Proposed Action.

6 ANALYSIS OF EFFECTS

The ESA section 7 regulations (50 CFR §402.02) define effects of the action as: "all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action but that are not part of the 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."

To understand the effects of the action to ESA-listed species and critical habitats, we employ a stressor-exposure-response analysis. The stressors resulting from the proposed action were identified in Section 2.5. In Section 4.1.1, we discussed those stressors that are not likely to adversely affect (NLAA) any ESA-listed species. The remaining stressors, which we will analyze in this section for each species, are:

1) Acoustic stressors from sonar and other transducers;
2) Acoustic stressors from air guns;
3) Explosive stressors in water; and
4) Physical disturbance and strike stressors from vessels and USVs;

Our analysis of the effects of the stressors listed above is organized in this section by species group (i.e., marine mammals, sea turtles, fish, and invertebrates) because, in general, we expect species within each group to respond in similar ways to these stressors. Where different responses by a species or subgroup (e.g., mysticetes versus odontocetes) within a species group have been identified or documented in the literature, we discuss those difference in our analysis and consider them in our effects determinations.

The following analysis is structured to first assess the exposure of ESA-listed species to the stressors above, followed by a separate assessment of the responses of ESA-listed species to that exposure. Species that we previously determined as not likely to be adversely affected (NLAA) by any of the stressors resulting from the proposed action (see Section 4.1.2) are not included in our analysis in this section.

We follow the same structure (i.e., an exposure analysis followed by a response analysis) to analyze the effects of the proposed action on the essential PBFs of critical habitat. Critical habitats that we previously determined are not likely to be adversely affected (NLAA) by any of the stressors resulting from the proposed action (see Section 4.1.3) are not included in our analysis in this section.

At the end of this section we provide a table summarizing all of our effects determinations for each stressor category and ESA-listed species/critical habitat combination (Table 61).

6.1 Marine Mammals

This section addresses the effects of acoustics (i.e., from sonar and other transducers, air guns, and weapons firing noise), explosives, and vessel strike stressors on ESA-listed marine mammals. Section 6.1.1 starts with an overview of the methodology for the Navy's quantitative analysis of the effects of sonar, air guns, and explosives from training and testing activities on marine mammals when they are in the water. In 6.1.2 we address the effects of weapons firing noise to ESA-listed pinnipeds when they are hauled out on land.

6.1.1 Acoustic and Explosive Effects Analysis

To estimate impacts from acoustic and explosive stressors associated with the proposed training and testing activities, the Navy performed a quantitative analysis to estimate the number of instances that ESA-listed marine mammals would be exposed to acoustic effects 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. NAEMO is also used to quantify acoustic impacts to ESA-listed sea turtles, which is discussed in Section 6.2.1.

An overview of the quantitative analysis approach 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 IV Training and Testing* (Navy 2024h). NMFS verified the methodology and data used by the Navy in this analysis and, unless otherwise specified in this opinion, we accept the Navy's quantitative analysis on exposure of marine mammals and sea turtles to sound generated by the proposed action. We consider the modeling conclusions from this analysis to represent the best available data on exposure of marine mammals and sea turtles to acoustic stressors from the proposed action, and the estimated impacts (e.g., mortality, non-auditory injury, auditory injury [AINJ], TTS, significant disruption of behavior) resulting from this analysis are reasonably certain to occur.

Navy Acoustic Effects Model (NAEMO)

NAEMO is used to assess the level of behavioral disturbance and physiological impacts (e.g., TTS and AINJ) predicted for individual marine mammals and turtles likely to be in the vicinity of training and testing activities due to the emission of sounds and impulses. NAEMO calculates

sound energy propagation from sonars and other transducers (as well as airguns and explosives) during naval activities and the sound received by animat dosimeters. Animat dosimeters are virtual representations of marine mammals distributed in the area around the modeled naval activity. Each of the animat dosimeters records its individual sound "dose." The animats do not represent actual animals, but rather allow for a statistical analysis of the number of instances during which marine mammals 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 that may be affected. Some individuals may be exposed more than once per year but the model does not estimate whether a single individual is exposed multiple times.

Marine mammal data input to the NAEMO include densities, group size, stock estimates, and depth distribution. Marine mammals are distributed within the action area with multiple iterations generated for each species to account for statistical uncertainty in the density estimates. Because 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 bases the distribution of animats over the action area on the density values in the *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area* (Navy 2025n), and distributes animats in the water column proportional to the known time that species spend at varying depths. NAEMO accounts for depth distributions by changing the depth of each animat 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 at each depth in the water column. Dive profiles for each species are provided in the Navy technical report *Dive Distribution and Group Size Parameters for Marine Species Occurring in the U.S. Navy's Atlantic and Hawaii-California Training and Testing Study Areas* (Navy 2024a).

Physical environment data plays an important role in acoustic propagation of underwater sound sources used in the impact modeling process (Navy 2024h). 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 could be affected by the aspects of the proposed activity that generate sound (Navy 2024h).

The model estimates the impacts caused by individual training and testing events. Each modeling scenario is connected to a modeling area to create a modeling event. The platforms within the

scenario are randomly distributed throughout the modeling area. The modeling areas can be quite large (i.e., range complexes and operation areas) and platforms included in the same modeling event are intended to interact during the exercise. During any individual modeled event, impacts on individual animats are considered over 24-hour periods. A Monte Carlo simulation approach which involves repeated random sampling is used to account for the variability inherent in many factors of training and testing events, such as the precise location of modeling area, platform location and movement, and instantaneous distributions of marine mammals (Navy 2024h).

The Navy, with input from NMFS, has continually sought to update and improve on NAEMO. The notable changes to the model from Phase III to Phase IV include (Navy 2024h):

- Updated density estimates and environmental data based on the best available science.
- The acoustic source binning process has been updated to more accurately group similar sources and use the parameters of the bin for propagation.
- Non-impulsive broadband sources were binned into broadband source bins and broken out into sub-bins at every octave within the frequency range. Non-impulsive sources are considered broadband in NAEMO if the bandwidth spans more than two octaves. Each sub-bin was propagated and simulated separately, more accurately capturing the frequency spectrum associated with these sources. In Phase III, broadband sources were modeled at a single frequency, the geometric mean of the low and high range, which did not provide as comprehensive a representation of the frequency profile.
- An animal avoidance process was developed and integrated into NAEMO for marine mammals. "Avoidance" refers to a marine mammal avoiding a high-intensity sound exposure by moving horizontally out of the immediate injury zone for subsequent exposures. The onset of avoidance was based on the behavioral response functions. Avoidance speeds and durations were informed by a review of available exposure and baseline data. This process reduces sound exposure level (SEL) accumulation for a given animat (virtual animal) by reducing the received sound pressure levels (SPL) of individual exposures based on a spherical spreading calculation from sources on each unique platform in an event. The SEL reduction could lower an animat's calculated value below the TTS or AINJ species-specific threshold, reducing the total number of auditory effects. An avoidance factor was only calculated for non-impulsive source (e.g., sonar and transducers) modeling but could not be used for impulsive modeling due to a lack of data on certain source count spacing. In Phase III avoidance was not modeled in NAEMO. Instead, 95% of the permanent threshold shift (PTS), now referred to as AINJ, predicted by NAEMO were assumed to be reduced to TTS due to avoidance (no further reduction below TTS was calculated for Phase III).
- The similitude equation used for impulsive modeling has been updated to reflect the most current version applicable for underwater explosions.

Density Estimates

For most cetacean species, abundance is estimated using line-transect surveys or mark-recapture studies. The result provides one single density estimate value for each species across broad geographic areas. This is the general approach applied in estimating cetacean abundance in NMFS' marine mammal stock assessment reports. Although the single value provides a good average estimate of abundance (total number of individuals) for a specified area, it does not provide information on the species distribution or concentrations within that area, and it does not estimate density for other timeframes or seasons that were not surveyed. More recently, habitat modeling has been used to estimate cetacean densities. These models estimate cetacean density as a continuous function of habitat variables (e.g., sea surface temperature, seafloor depth, etc.) and thus allow predictions of cetacean densities on finer spatial scales than traditional line-transect or mark recapture analyses. Within the geographic area that was modeled, densities can be predicted wherever these habitat variables can be measured or estimated.

To characterize the marine species density for large areas such as the HCTT action area, the Navy compiled data from several sources. The Navy developed a protocol to select the best available data sources based on species, area, and time (season). The resulting Geographic Information System database called the Navy Marine Species Density Database (NMSDD) includes seasonal density values for every marine mammal species present within the HCTT action area. This database is described in the technical report titled *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area* (Navy 2025n), hereafter referred to as the density technical report.

Updates and improvements to the NMSDD for HCTT Phase IV included the use of additional systematic survey data, improvements to habitat modeling methods used to estimate species density, updated abundance estimates, and more detailed seasonal distribution information. New survey data collected by NMFS within the EEZ of the Hawaiian Islands during the summer and fall of 2017 and the winter of 2020 supported the derivation of updated cetacean density estimates from both design- and model-based analyses (Navy 2025n).The winter sighting data also enabled an examination of seasonal differences in the abundance and distribution of cetaceans and supported the development of a new habitat-based density model for humpback whale that provided monthly density predictions for this species. A new habitat-based density model was also developed for sperm whales. A new abundance estimate for the Hawaiian monk seal was published in the 2020 stock assessment report (Carretta 2021b) and was used to update density estimates for this species in Hawaiian waters. In addition, Hawaiian monk seal densities were estimated for individual islands using island-specific abundances instead of calculating separate uniform densities for the Main Hawaiian Islands (MHI) and Northwest Hawaiian Islands (NWHI).

The California Current Ecosystem habitat-based density models were updated based on additional survey data collected in summer and fall of 2014 and 2018 off the U.S. West Coast (Navy 2025n). The model-based analyses provided updated density estimates for summer and

fall for most cetacean species occurring off California and within the HCTT action area. Cetacean habitat-based density models specific to winter/spring were developed for waters off Southern California using 2005-2020 California Cooperative Oceanic Fisheries Investigations shipboard survey data. Density estimates for all pinniped species off California were revised by integrating updated abundance estimates, seasonal haul-out factors (if applicable), and more localized distribution areas defining new strata (Navy 2025n).

To improve density estimates for Phase IV, the Navy funded an analysis to develop habitat-based density models for the Southern California Current, from Point Conception to the tip of the Baja California Peninsula (Navy 2025n). These models provided the first spatially-explicit estimates of average species density and abundance for waters off the Baja California Peninsula, finer-scale density predictions, and improved, spatially explicit estimates of uncertainty for the following ESA-listed marine mammals: sperm whale, blue whale, fin whale, and humpback whale (Navy 2025n). Phase III density estimates for Guadalupe fur seals were extrapolated along the Baja California Peninsula, Mexico. For Phase IV, density estimates were refined by defining strata based on recent survey data as well as migration and haul-out behavior.

NMFS recognizes two stocks of gray whales in the North Pacific: the larger Eastern North Pacific stock and the highly endangered Western North Pacific stock. While it is possible that sightings of western population animals might be included in the data used to estimate gray whale density in the Eastern North Pacific, given the current paucity of data regarding the western population, as well as the very low population numbers, separate density estimates for the western population were not included in the NMSDD Phase IV (Navy 2025n). For purposes of the HCTT acoustic effects analysis, it was assumed that 1% of the gray whales located in SOCAL RC/NOCAL RC are from the ESA-listed Western North Pacific DPS (Chip Johnson, Navy US Pacific Fleet, pers. comm. to Ron Salz, NOAA Fisheries, Office of Protected Resources, December 16, 2024).

Criteria and Thresholds

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 training and testing activities.

Non-auditory Injury and Mortality

The Navy's quantitative analysis assessed the potential for HCTT activities involving in-water explosives to physically injure or kill marine mammals. The effects of underwater explosions on marine mammals depend on a variety of factors including animal size and depth; charge size and depth; depth of the water column; and distance between the animal and the charge. The gas-containing organs (lungs and gastrointestinal tract) are most vulnerable to primary blast injury from barotrauma. Severe injuries to these organs are presumed to result in mortality (e.g., severe

lung damage may introduce air into the cardiopulmonary vascular system, resulting in lethal air emboli) (Navy 2025c).

Two metrics are used to assess injurious exposures to explosives: peak pressure and impulse. The impulse criterion relies on two variables: the mass of an animal (M) and its depth (D) below the water surface. The impulse is calculated over the lesser of the duration of the initial positive pressure or 20% of the estimated lung resonance period (Navy 2025c). In Phase IV, the thresholds for non-auditory injury and mortality are anchored to the lowest exposures with an effect observed in the underlying experimental data sets, rather than the mean value as in Phase III. The criteria used for Phase IV follow a similar methodology as past Navy explosive impact analyses (Finneran and Jenkins 2012; Navy 2017; Navy 2024d). Table 32 compares the Phase III and Phase IV thresholds for mortality and injury.

Table 32. Comparison of Phase III and Phase IV Mortality and Injury Thresholds

Impact Category	Phase III	Phase IV
Mortality (Impulse)	$144M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$ Pa-s	$103\mathrm{M}^{1/3}\left(1+\frac{\mathrm{D}}{10.1}\right)^{1/6}$ Pa-s
Injury (Impulse)	$65.8M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$ Pa-s	$47.5M^{1/3}\left(1+\frac{D}{10.1}\right)^{1/6}$ Pa-s
Injury (Peak Pressure)	243 dB re: 1 µPa SPL peak	237 dB re: 1 µPa SPL peak

Notes: dB re: 1 µPa: decibels referenced to 1 micropascal; Pa-s: pascal second; SPL: sound pressure level; D: depth of animal (m); M: mass of animal (kilograms).

Auditory Impacts

Exposure to sound with sufficient duration and sound pressure level (SPL) may result in an elevated hearing threshold (i.e., a loss of hearing sensitivity), called a noise-induced threshold shift (NITS). If the hearing threshold eventually returns to normal, the NITS is called a temporary threshold shift (TTS); otherwise, if thresholds remain elevated after some extended period of time, the remaining NITS is called a permanent threshold shift (PTS).

The Navy, in coordination with NMFS, established marine mammal acoustic thresholds that identify 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 some degree of TTS or AINJ (which includes PTS). 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 marine mammal criteria and threshold development is included in the Navy's technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV)* (Navy 2025c). The thresholds used by the Navy were developed by compiling and synthesizing the best available scientific information 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 HCTT activities.

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 impacts to different marine mammal groups (based on hearing sensitivity) as a result of exposure to noise from two different types of sources: impulsive (e.g., explosives and air guns) and non-impulsive (e.g., sonar and other transducers).

Marine mammals were divided into groups for analysis. For purposes of the Phase IV NAEMO analysis, group names were changed from Phase III to be consistent with Southall et al. (2019b). The low-frequency cetacean group (i.e., mysticetes) was split into two groups (i.e., very-low frequency group and low frequency group[25]) with the upper hearing limit for the LF group increased from Phase III to match recent hearing measurements in minke whales (Houser et al. 2024). Houser et al. (2024) temporarily held two adolescent minke whales in pens for hearing tests, which showed that minke whales are sensitive to sound frequencies as high as 45 to 90 kHz. This study challenged the long-standing notion that all mysticetes are low-frequency hearing specialists. For Phase IV, the mid-frequency cetacean group is now designated as the high-frequency cetacean group; and the group previously designated as high-frequency cetaceans is now the very high-frequency cetacean group.

The five groups relevant to the effects analysis for the ESA-listed marine mammals considered in this opinion are: very-low frequency (VLF) cetaceans (e.g., fin and blue whales); low-frequency (LF) cetaceans (e.g., sei, gray and humpback whales); high-frequency (HF) cetaceans (e.g., sperm whales and false killer whales); phocid carnivores in water (PCW) and phocid carnivores in air (PCA) such as Hawaiian monk seals, and otariids and other non-phocid marine carnivores in water (OCW) and air (OCA) such as Guadalupe fur seals.

Other notable changes from Phase III include significantly lower TTS/AINJ thresholds for the OCW group based on new TTS data for California sea lions (Kastelein et al. 2020; Kastelein et al. 2021) and slightly lower TTS/AINJ thresholds at high frequencies for the PCW based on new TTS data for harbor seals (Kastelein et al. 2020; Kastelein et al. 2021).

The Navy's acoustic impact analyses uses auditory weighting functions to capture the frequency dependent nature of the effects of noise. Auditory weighting functions are mathematical

[25] Note: This approach differs from NMFS's 2024 Update to Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing NMFS. 2024f. Reinitiation of the Biological and Conference Opinion on (1) U.S. Navy Hawaii-Southern California Training and Testing, 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 Hawaii-Southern California Training and Testing Through December 2025. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce., which considers all baleen whales as part of the low frequency cetacean hearing group.

functions used to emphasize frequencies where animals are more susceptible to noise exposure and deemphasize frequencies where animals are less susceptible. A frequency-dependent weighting function (Figure 67) and numeric thresholds for the onset of temporary threshold shift (TTS) and the onset of auditory injury (AINJ) were estimated for each marine mammal hearing group. New weighting functions and TTS/AINJ thresholds were developed for Phase IV based on the availability of new data regarding marine mammal hearing and the effects of noise on marine mammal hearing.

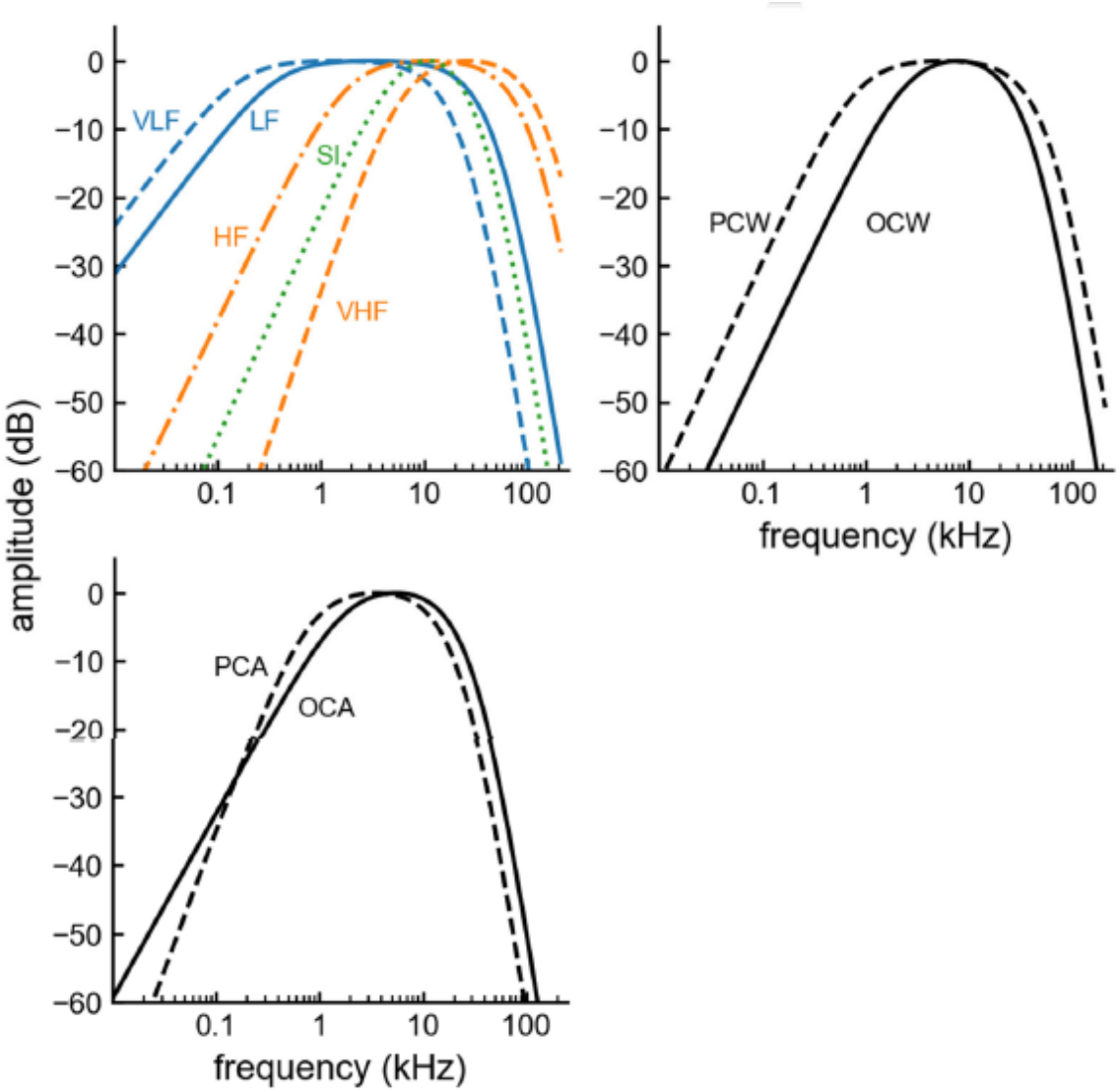


Figure 67. Navy Phase IV Weighting Functions by Marine Mammal Species Group (amplitude in dB on y-axis; frequency in kHz on x-axis)

The onset of TTS is defined as a TTS of 6 dB measured approximately 2–5 minutes after exposure. A TTS of 40 dB is used as a proxy for the onset of AINJ, which include PTS. For non-impulsive exposures, onset levels are specified in frequency-weighted sound exposure level

(SEL); for impulsive exposures, dual metrics of weighted SEL and unweighted peak sound pressure level (SPL) are used.

Figure 68 and Figure 69 compare exposure functions across species groups, for non-impulsive and impulsive exposures, respectively.

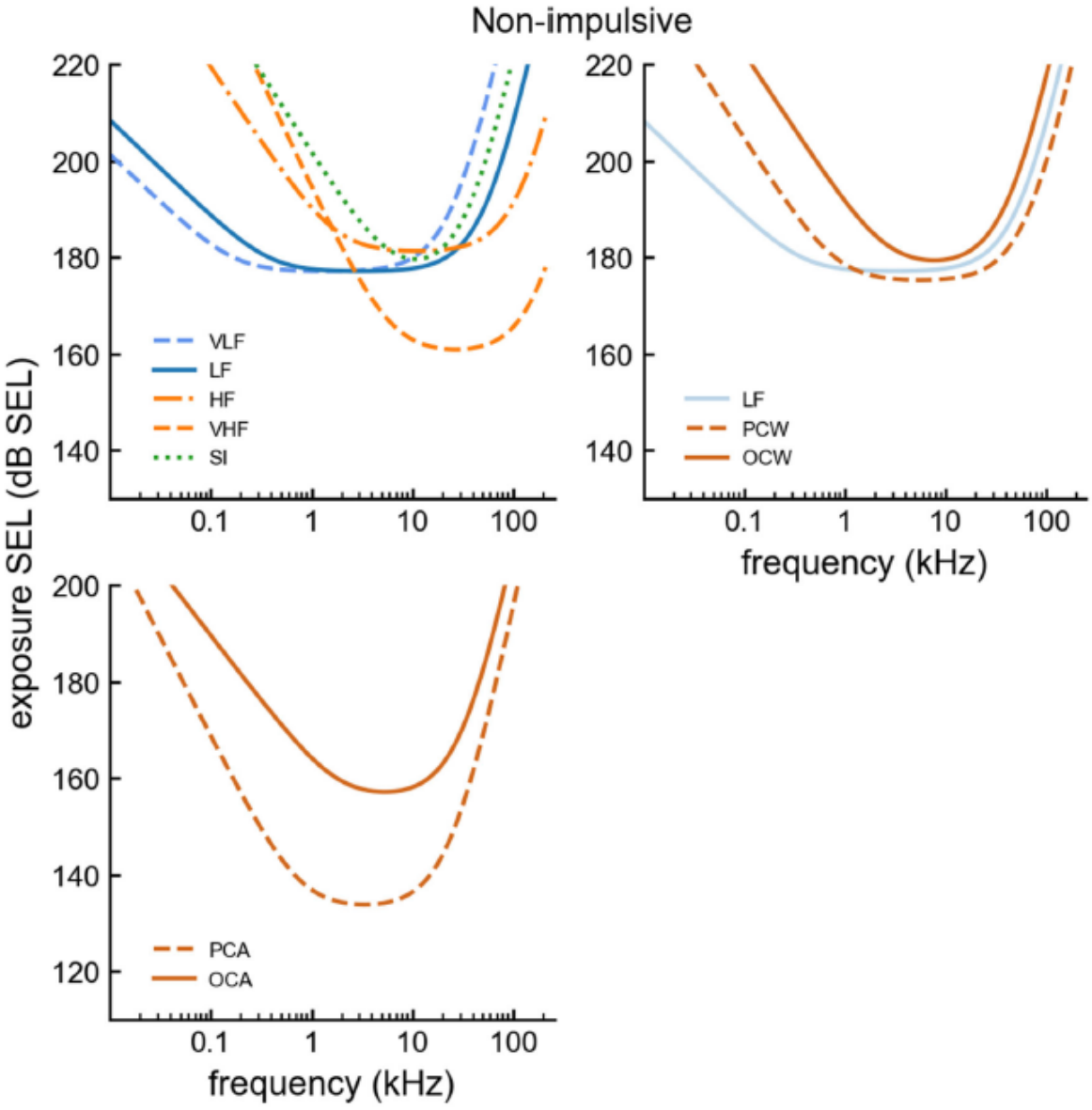


Figure 68. Comparison of Navy Phase IV TTS Exposure Functions for Sonars and Other Non-impulsive Active Acoustic Sources across Species Groups (exposure SEL in dB on y-axis; frequency in kHz on x-axis)

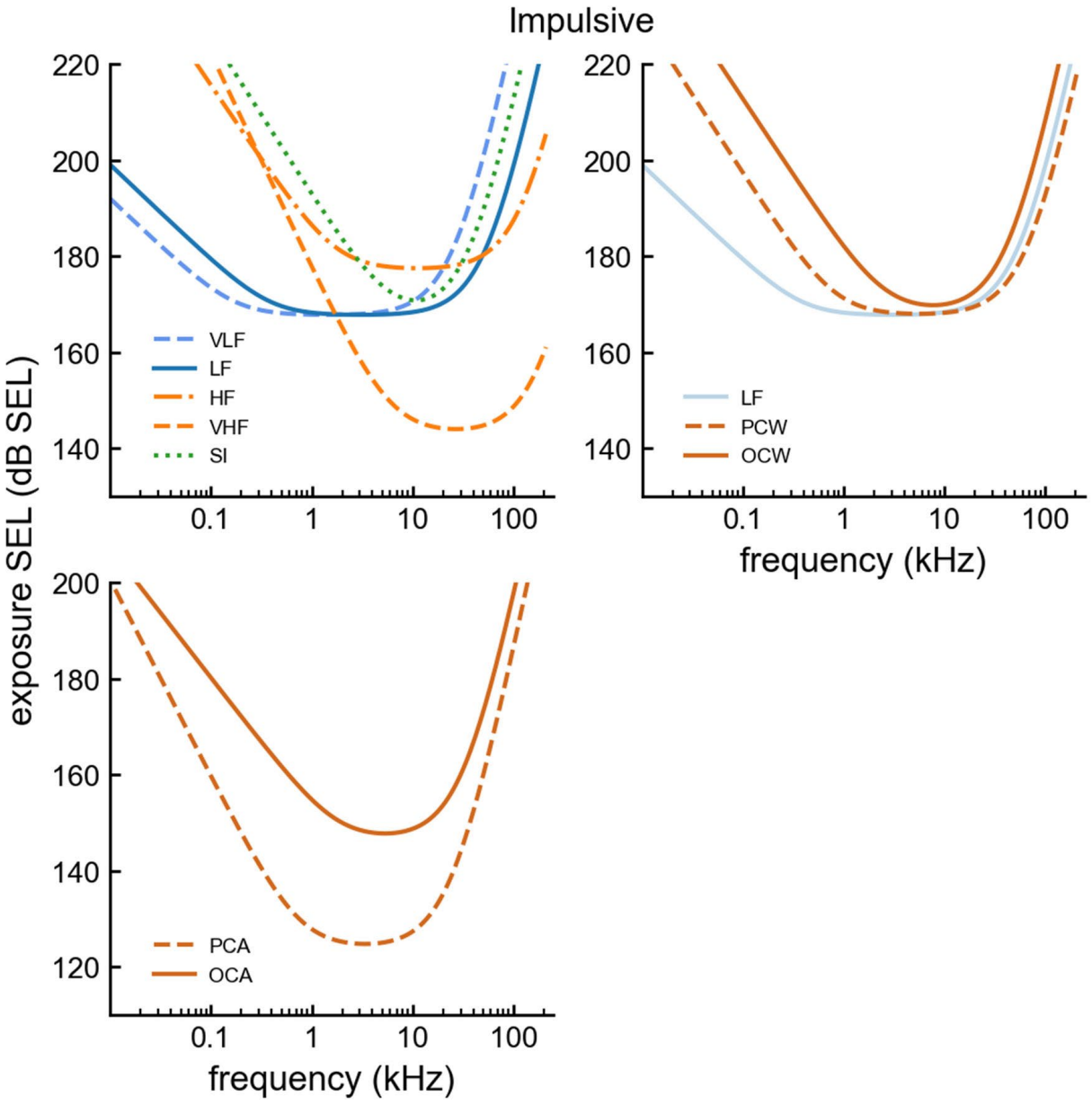


Figure 69. Comparison of Navy Phase IV TTS Exposure Functions for Explosives, Impact Pile Driving, Air Guns and Other Impulsive Acoustic Sources Across Species Groups (exposure SEL in dB on y-axis; frequency in kHz on x-axis)

Table 33 summarizes the weighted TTS and AINJ thresholds by source type (i.e., impulsive and non-impulsive) and by marine mammal hearing group (i.e., VLF-very low frequency cetaceans, LF-low frequency cetaceans, HF-high frequency cetaceans, VHF – very high frequency cetaceans, OCW- otariids and other non-phocid marine carnivores in water, PCW - phocid carnivores in water, SI – sirenians, OCA - otariids and other non-phocid marine carnivores in air, and PCA - phocid carnivores in air).

Table 33. Summary of Phase IV TTS/AINJ Thresholds (SEL thresholds are in dB re: 1 µPa²s underwater) by Marine Mammal Hearing Group

Group	Non-impulsive TTS threshold SEL (weighted)	Non-impulsive INJ threshold SEL (weighted)	Impulsive TTS threshold SEL (weighted)	Impulsive TTS threshold peak SPL (unweighted)	Impulsive INJ threshold SEL (weighted)	Impulsive INJ threshold peak SPL (unweighted)
VLF	177	197	168	216	183	222
LF	177	197	168	216	183	222
HF	181	201	178	224	193	230
VHF	161	181	144	196	159	202
OCW	179	199	170	224	185	230
PCW	175	195	168	217	183	223
SI	180	200	171	219	186	225
OCA	157	177	148	171	163	177
PCA	134	154	125	156	140	162

Behavioral Response Criteria

For their Phase IV analysis the Navy updated the criteria and numeric thresholds for predicting marine mammal behavioral responses when exposed to non-impulsive sources and impulsive sources. Behavioral response functions were developed to estimate numbers of animals that could exhibit significant behavioral responses to HCTT activities. The term "significant behavioral response" is used by the Action Proponents to describe behavioral observations that may rise to the level of "harassment" under the MMPA. Under the MMPA, for military readiness activities such as Navy and Coast Guard training and testing, behavioral harassment is defined as "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are

abandoned or significantly altered" (Section 315(f) of Public Law 107–314; 16 U.S.C. 703 note)[26].

Responses reported from all marine mammal behavioral response and controlled exposure studies were included in the analysis. Responses were scored and response significance was determined based on a behavioral response severity scale (i.e., low, moderate, or high) derived from Southall et al. (2021) as follows:

- Low severity responses are 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.
- Moderate severity responses could become significant if sustained over a longer duration. For the derivation of the Navy's behavioral criteria, a long duration was defined as a response that lasted for the duration of exposure or longer, regardless of how long that exposure lasted.
- High severity responses include those with immediate consequences (e.g., stranding) and those affecting animals in vulnerable life stages (i.e., calf, pup, or cub), and are therefore always considered to be a significant behavioral reaction.

Marine mammals were divided into four groups for this analysis, three of which include species considered in this opinion (i.e. mysticetes, odontocetes, and pinnipeds). For each group, to estimate the impacts from non-impulsive sources (including HCTT sonar), a biphasic behavioral response function was developed using best available data and Bayesian dose response models (Figure 70 through Figure 72). Compared to Phase III, the odontocete behavioral response function is less sensitive in Phase IV. Similarly, the mysticete curve in Phase IV is less sensitive at higher received levels than the general mysticete curve from Phase III.

To estimate the significant behavioral response impacts from explosives, if more than one explosive event occurs within any given 24-hour period within a training or testing activity, criteria are applied to predict the number of animals that may have a behavioral reaction. For events with multiple explosions, the behavioral threshold used in this analysis is 5 dB less than the TTS onset threshold. This value is derived from observed onsets of behavioral response by test subjects (bottlenose dolphins) during non-impulse TTS testing (Schlundt et al. 2000). For single explosions at received sound levels below hearing loss thresholds, the modeled assumption is that significant behavioral reactions would not be expected to occur (i.e., either low severity responses or short-term moderate severity responses would be expected).

[26] By comparison, NMFS's interim guidance on the term "harass" under the ESA defines 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" NOAA. 2016. Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Silver Spring, Maryland..

There is empirical evidence to suggest animals are not likely to exhibit behavioral response when exposed to low levels of sound from a source that is far away (Navy 2025c). To account for this in Phase IV, NAEMO has implemented behavioral cutoffs that consider both received sound level and distance from the source. These cutoffs are unique to each behavioral hearing group. For mysticetes the modeled cutoff was 10 km with a received level of 185 dB RMS; i.e. animals that are beyond 10 km from the source and below 185 dB re: 1 µPa RMS are unlikely to incur significant behavioral reactions. For odontocetes the modeled cutoff was 15 km with a received level of 168 dB RMS; i.e., animals that are beyond 15 km from the source and below 168 dB re: 1 µPa RMS are unlikely to incur significant behavioral reactions. For both mysticetes and odontocetes, regardless of received level, the model considered animals beyond 100 km from the source as unlikely to incur significant behavioral responses.

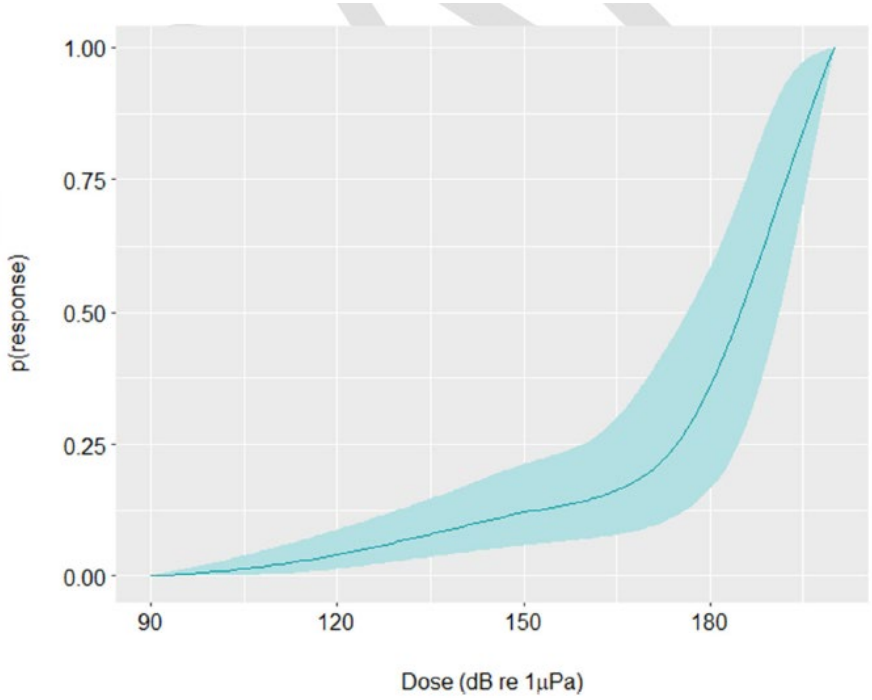


Figure 70. Phase IV Mysticete Biphasic Behavioral Response Function (50% credible interval shaded in teal)

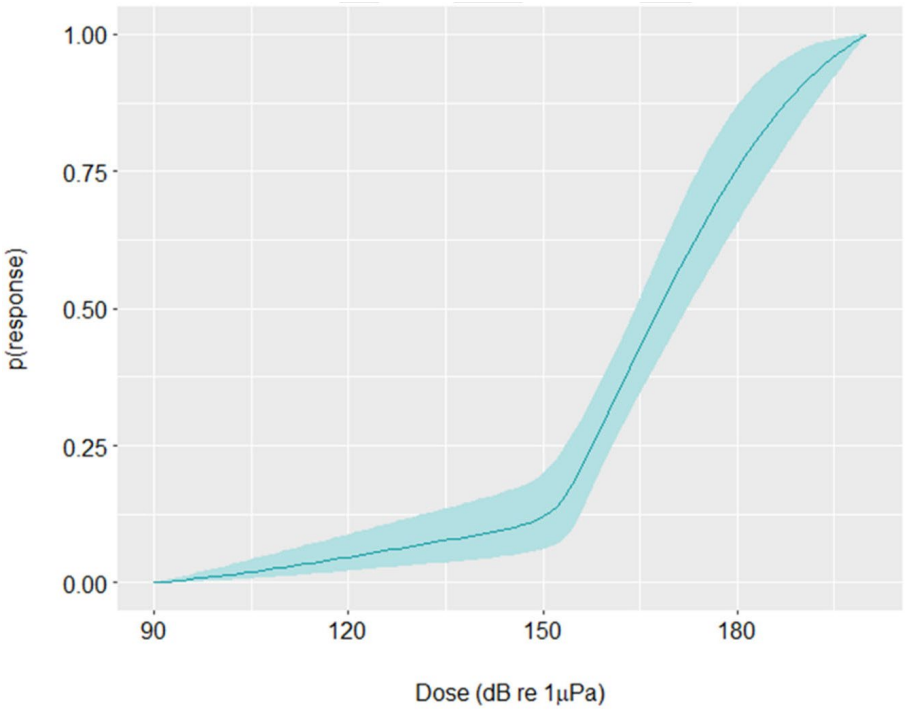


Figure 71. Phase IV Odontocete Biphasic Behavioral Response Function (50% credible interval shaded in teal)

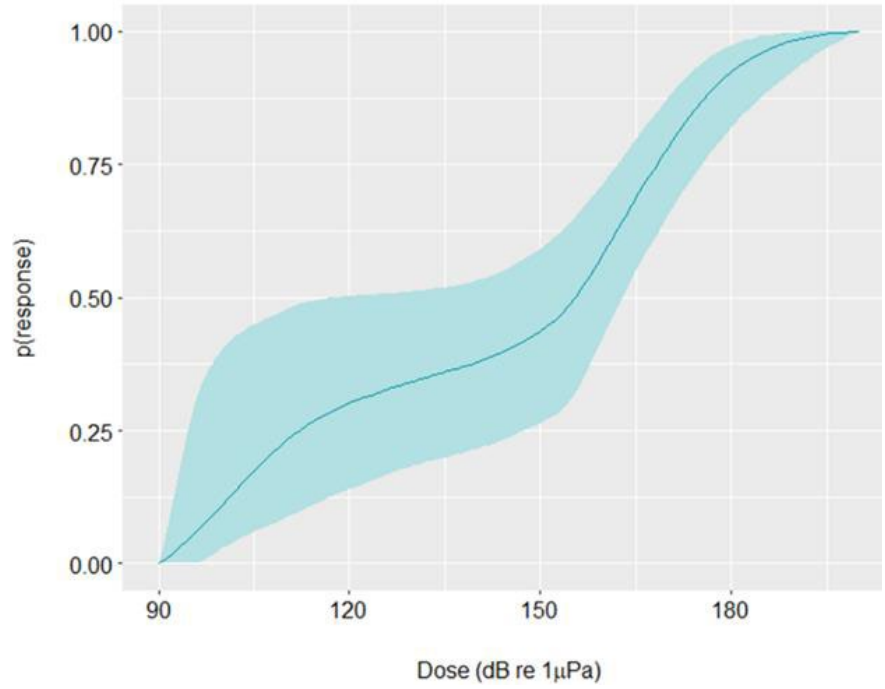


Figure 72. Phase IV Pinniped (in Water) Biphasic Behavioral Response Function (50% credible interval shaded in teal)

Ranges to Effects

A "range to effect" is the distance from where a source is used to the furthest distance at which an effect would be expected to occur. Ranges to effects are used to help predict impacts from acoustic and explosive sources and assess the benefit of mitigation zones (Navy 2024d). Ranges are determined by modeling the distance that noise from a source will need to propagate to reach exposure level thresholds specific to a hearing group that will cause a significant behavioral response, TTS, AINJ, non-auditory injury, or mortality. This value depends on several factors including: source bin parameters (frequency, sound level, pulse length, pulse interval); source exposure parameters (duration for sonar, cluster size for explosives); source origin (latitude, longitude, depth, time of year); source directionality (bearing); species dive profile; and species effect criteria. New for Phase IV, the Navy used species dive profiles when computing ranges-to-effects. In Phase III only the maximum dive depth of the species was used and an even dive profile (constant for all depths) was assumed. The ranges shown below do not account for an animal avoiding a source nor for the movement of the platform, both of which would influence the actual range to onset of auditory effects during an actual exposure (Navy 2024d).

Ranges to effects to marine mammals from exposure to sonar and other transducers, airguns, and explosives are provided in the Navy's *Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area* (Navy 2025f), and are incorporated into this opinion by reference (Section 2.5.1, pp. 2-91 to 2-139). For sonar and other transducers, ranges to TTS and AINJ are shown by functional hearing group (i.e., VHF, HF, LF, VLF) for four different exposure durations (i.e., 1, 30, 60, and 120 seconds) for the following six representative sonar systems: hull-mounted surface ship sonar (MF1); hull-mounted surface ship sonar (MF1K – Kingfisher mode); hull-mounted surface ship sonar (MF1C – duty cycle > 80%); helicopter dipping sonar; sonobuoy sonar; and towed mine-hunting sonar. For each sonar system, behavioral response curves are provided showing the probability of a behavioral response (i.e., mean, 5th, and 95th percentile) as a function of range (km) by behavioral response group: sensitive species, mysticete, odontocete, and pinniped (note: only mysticete and odontocete groups relevant for this consultation).

For explosives, ranges are provided for a representative cluster size for each bin. Ranges for behavioral response are only provided if more than one explosive cluster occurs. Single explosions at received sound levels below TTS and AINJ thresholds are most likely to result in a brief alerting or orienting response. Due to the lack of subsequent explosions, a significant behavioral response is not expected for a single explosive cluster (Navy 2024d). 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 AINJ 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. The explosive ranges to effects for TTS and AINJ that are in the

tables are based on the metric (i.e., SEL or SPL) that produced longer ranges (Navy 2024d). For non-auditory injury in the tables, the larger of the range to slight lung injury or gastrointestinal tract injury was used as a conservative estimate (Navy 2024d). Since the non-auditory metric is SPL-based, ranges are only available for a cluster size of one. Animals within water volumes encompassing the estimated range to non-auditory injury 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 (Navy 2024d).

Exposure Analysis

Above we presented information on the criteria and thresholds used to estimate impacts to marine mammals from acoustic and explosive stressors. Additional information on these criteria is described in the Navy's criteria and thresholds technical report (Navy 2025c). This section presents information on the estimated number of exposures of ESA-listed marine mammals to acoustic (i.e., sonar and other transducers, and air guns) and explosive stressors that are expected to result in a significant behavioral response, TTS, AINJ, non-auditory injury, or mortality. Exposure estimates were produced using the Navy's NAEMO approach as described above, and in more detail in the Navy's technical report (Navy 2024h).

Exposure impacts are assigned to the highest order threshold exceeded at the animat (i.e., non-auditory injuries are assumed to outrank auditory effects, and auditory effects are assumed to outrank significant behavioral responses) (Navy 2025c). In all instances any auditory impact or injury are assumed to represent a concurrent significant behavioral response. For example, if a significant behavioral response and TTS are predicted for the same animat in a modeled event, the effect is counted as a TTS in the table.

The summation of instances of effect includes all fractional values caused by averaging multiple modeled iterations of individual events. Impacts are only rounded to whole numbers at the level of substressor and type of activities. Rounding follows standard rounding rules, in which values less than 0.5 round down to the lower whole number, and values equal to or greater than 0.5 round up to the higher whole number (Navy 2025c). A value of one shown in parentheses [i.e., (1)] indicates that although the maximum annual estimate was less than 0.5, one or more impacts are predicted over every seven years of the proposed action.

The impacts tables below for each ESA-listed marine mammal include five sections, each representing a different way to analyze impacts.

- The first section shows the number of instances of each effect type that could occur due to each substressor (sonar, air guns, or explosives) and type of activity (Navy training [including U.S. Marine Corps], Coast Guard training, Navy testing activities, or Army training) over a maximum year of activity. No in-water explosives or acoustic stressors would result from Air Force activities.

- The second section estimates the average number of times an individual in the stock would be affected. The average annual impacts per individual is the sum of all instances of effect divided by the population abundance estimate. As noted previously, the number of instances of effect is not the same as the number of individuals that could be affected, as some individuals in a stock could be affected multiple times, whereas others may not be affected at all. The population abundance estimate used to calculate this average was derived as the greater value of the following: 1) the best population estimate from the most recent NMFS stock assessment report (SAR) for that species or stock (Carretta et al. 2024; Carretta et al. 2025), or 2) the average abundance predicted by the NMSDD within the U.S. EEZ. These values are shown in Table 34 for ESA-listed species/stocks with modeled impacts in the action area.

Table 34. Estimated Abundances of ESA-Listed Marine Mammal Stocks Present in the HCTT Action Area Used in NAEMO (Navy 2025f)

Species	*Stock*	*SAR*[1]	*NMSDD in EEZ*[2]
Blue whale	Eastern North Pacific	1,898	3,233
Blue whale	Central North Pacific	133	170
Grey whale	Western North Pacific	290	110
Fin whale	Hawaii	203	226
Fin whale	California/Oregon/Washington	11,065	12,304
Sei whale	Hawaii	391	452
Sei whale	Eastern North Pacific	864	155
Humpback whale	Mainland Mexico - California/Oregon/Washington	3,477	3,741
Humpback whale	Central America/Southern Mexico - California/Oregon/Washington	1,496	1,603
Sperm whale	Hawaii	5,387	6,062
Sperm whale	California/Oregon/Washington	2,606	4,549
False killer whale	Main Hawaiian Islands Insular	138	98
Guadalupe fur seal	Mexico	63,850 [3]	96,468 [4]
Hawaiian monk seal	Hawaii	1,605 [3]	967

[1] Best abundance estimate from the 2023 U.S. Pacific Marine Mammal Stock Assessment Report(Hayes et al. 2023; Carretta et al. 2024; Carretta et al. 2025) used unless otherwise noted.
[2] See the Navy *Density Technical Report* (Navy 2025n) for additional information.
[3] Best abundance estimate for this species from the draft 2024 U.S. Pacific Marine Mammal Stock Assessment Report (Carretta et al. 2025).
[4] Based on emergent unpublished data indicating a higher abundance for Guadalupe fur seals than previously estimated in the NMSDD or the draft 2024 SAR (Navy 2025f).

- The third section shows the percent of total impacts that would occur within seasons and the following general geographic areas: HRC; SOCAL RC; PMSR; NOCAL RC; and High Seas (transit lanes between the California and Hawaii portions of the action area).
- The fourth section shows which military readiness activities are most impactful to a species. Activities that cause 5% or more of total impacts to a stock are shown.

- The fifth section shows the number of impacts within critical habitat (if designated for the species) and within identified BIAs, which represent areas and times where marine mammal species are known to concentrate for activities related to reproduction, feeding, and migration, as well as the known ranges of small and resident populations. Impacts are categorized by type of BIA as follows: S - Small/Resident population, M - Migratory, F - Feeding, R - Reproductive, P - Parent, and C - Child/Core.

Blue Whale

Estimated exposures to blue whales from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 35 and Table 36. The large majority of impacts to blue whales will occur to individuals from the Eastern North Pacific stock within the SOCAL RC. Anti-Submarine Warfare activities account for a large proportion of the acoustic impacts from sonar, whereas most blue whale impacts due to explosives are from Mine Warfare activities, with some impacts attributable to Small Ship Shock Trials (Navy 2025f). Because blue whales densities are much lower in the Central North Pacific region, there are comparatively fewer impacts to this stock as compared to the Eastern North Pacific stock.

Table 35. Estimated Effects to the Eastern North Pacific Stock of Blue Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	0	-	-	-	-
Explosive	Navy Training	65	81	1	-	-
Explosive	Navy Testing	21	25	2	-	-
Explosive	USCG Training	(1)	-	-	-	-
Sonar	Navy Training	646	1,924	16	-	-
Sonar	Navy Testing	696	1,094	8	-	-
Sonar	USCG Training	18	-	-	-	-
Maximum Annual Total		**1,447**	**3,124**	**27**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
3,233	1.42	0.01

	Percent of Total Effects		
Season	SOCAL	PMSR	NOCAL
Warm	44%	7%	5%
Cold	43%	1%	1%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	17%
Acoustic and Oceanographic Research (ONR)	Navy Testing	11%
Medium Coordinated Anti-Submarine Warfare	Navy Training	10%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	8%
Small Joint Coordinated Anti-Submarine Warfare	Navy Training	7%
At-Sea Sonar Testing	Navy Testing	6%
Composite Training Unit Exercise (Strike Group)	Navy Training	6%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
F-BIA-C	West Coast (6,7,8,9,10,11)	37	60	1	-	-
F-BIA-P	West Coast (6,7,8,9,10,11)	461	645	3	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Table 36. Estimated Effects to the Central North Pacific Stock of Blue Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	(1)	-	-	-	-
Sonar	Navy Training	10	56	0	-	-
Sonar	Navy Testing	5	19	(1)	-	-
Sonar	USCG Training	(1)	-	-	-	-
Maximum Annual Total		**17**	**75**	**1**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
170	0.55	0.01

	Percent of Total Effects	
Season	HRC	High Seas
Warm	29%	1%
Cold	66%	4%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	36%
Acoustic and Oceanographic Research (ONR)	Navy Testing	14%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	9%
Submarine Sonar Maintenance and Systems Checks	Navy Training	9%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Fin Whale

Estimated exposures to fin whales from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 37 and Table 38. The large majority of impacts to fin whales will occur to individuals from the California, Oregon, and Washington stock within the SOCAL RC. Some whales from this stock may be present throughout the year in southern and central California, as the Southern California Bight is likely home to a small year-round resident population (Navy 2025f). On average, individuals from the California, Oregon, and Washington stock will be impacted about once a year, although we would expect higher than average exposures to acoustic impacts for resident fin whales that occur within SOCAL RC throughout most or all of the year. Anti-Submarine Warfare activities account for a large proportion of the acoustic impacts from sonar, whereas fin whale impacts due to explosives are from a variety of activities including Ship Shock Trials, Explosive Ordnance Disposal (EOD) Mine Neutralization, and Amphibious Breaching activities (Navy 2025f). Because fin whale densities are much lower in the Hawaii Range Complex, there are comparatively fewer impacts to this stock as compared to the California, Oregon, and Washington stock.

Table 37. Estimated Effects to the California, Oregon, and Washington Stock of Fin Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	0	0	-	-	-
Explosive	Navy Training	98	114	5	(1)	-
Explosive	Navy Testing	76	69	6	0	-
Explosive	USCG Training	0	0	0	-	-
Sonar	Navy Training	1,727	5,470	22	-	-
Sonar	Navy Testing	1,741	4,144	21	-	-
Sonar	USCG Training	62	-	-	-	-
Maximum Annual Total		**3,704**	**9,797**	**54**	**1**	**-**

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
12,304	1.10	0.00

	Percent of Total Effects		
Season	SOCAL	PMSR	NOCAL
Warm	28%	19%	23%
Cold	23%	4%	2%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	21%
Acoustic and Oceanographic Research (ONR)	Navy Testing	17%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	11%
At-Sea Sonar Testing	Navy Testing	10%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	6%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	6%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
F-BIA-C	West Coast (6,7,8,9,10,11)	1,405	3,974	19	-	-
F-BIA-P	West Coast (6,7,8,9,10,11)	1,977	5,653	28	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Table 38. Estimated Effects to the Hawaii Stock of Fin Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	(1)	0	0	-	-
Explosive	Navy Testing	(1)	0	-	-	-
Sonar	Navy Training	12	46	0	-	-
Sonar	Navy Testing	5	19	(1)	-	-
Sonar	USCG Training	2	-	-	-	-
Maximum Annual Total		**21**	**65**	**1**	**-**	**-**

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
226	0.38	0.00

	Percent of Total Effects	
Season	HRC	High Seas
Warm	24%	1%
Cold	73%	2%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	30%
Acoustic and Oceanographic Research (ONR)	Navy Testing	16%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	13%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Humpback Whale - Mexico DPS

Estimated exposures to the humpback whale Mexico DPS from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 39. The highest abundance of this DPS in the action area is during the cold season, when humpbacks are most likely to be impacted by sonar training and testing activities during Anti-Submarine Warfare activities, Intelligence, Surveillance, Reconnaissance testing activities, Acoustic and Oceanographic Research, and At-Sea Sonar Testing. About one-third of the total impacts to this DPS would occur in humpback whale critical habitat off the coast of California (Navy 2025f). On average, individuals from the Mexico DPS could be impacted about once a year, and a small number of non-auditory injuries (i.e., less than one per year) are predicted from acoustics impacts to this DPS.

Table 39. Estimated Effects to the Humpback Whale Mexico DPS over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	0	0	-	-	-
Explosive	Navy Training	35	85	3	-	-
Explosive	Navy Testing	31	29	1	(1)	-
Explosive	USCG Training	(1)	0	-	-	-
Sonar	Navy Training	375	1,906	31	-	-
Sonar	Navy Testing	818	1,155	8	-	-
Sonar	USCG Training	14	-	-	-	-
Maximum Annual Total		**1,274**	**3,175**	**43**	**1**	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
3,741	1.20	0.01

	Percent of Total Effects			
Season	SOCAL	PMSR	NOCAL	High Seas
Warm	6%	6%	17%	0%
Cold	52%	12%	6%	1%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	19%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	12%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	11%
Acoustic and Oceanographic Research (ONR)	Navy Testing	10%
At-Sea Sonar Testing	Navy Testing	7%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	6%
Undersea Warfare Testing	Navy Testing	5%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat	CA Central Coast (All)	54	222	4	-	-
Critical Habitat	CA North Coast (All)	0	3	0	-	-
Critical Habitat	Channel Islands Area (All)	71	307	4	-	-
Critical Habitat	San Francisco Monterey Bay Area (All)	64	680	10	-	-
F-BIA-C	West Coast (3,4,5,6,7,8,9,10,11)	17	72	1	-	-
F-BIA-P	West Coast (3,4,5,6,7,8,9,10,11)	94	495	8	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Humpback Whale - Central America DPS

Estimated exposures to the humpback whale Central America DPS from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 40. This DPS of humpback whales migrates through California with peak abundance December through June ("cold season"), when humpbacks are most likely to be impacted by sonar training and testing during Anti-Submarine Warfare activities, Intelligence, Surveillance, Reconnaissance testing

activities, Acoustic and Oceanographic Research, and At-Sea Sonar Testing. About one-third of the total impacts to this DPS would occur in humpback whale critical habitat off the coast of California (Navy 2025f). On average, individuals from the Central America DPS could be impacted about once a year, and no non-auditory injury or mortality is predicted from acoustics impacts to this DPS.

Table 40. Estimated Effects to the Humpback Whale Central America DPS over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	0	-	-	-	-
Explosive	Navy Training	18	27	(1)	-	-
Explosive	Navy Testing	13	11	1	-	-
Explosive	USCG Training	0	0	-	-	-
Sonar	Navy Training	166	831	13	-	-
Sonar	Navy Testing	343	472	4	-	-
Sonar	USCG Training	7	-	-	-	-
Maximum Annual Total		**547**	**1,341**	**19**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
1,603	1.19	0.01

	Percent of Total Effects		
Season	SOCAL	PMSR	NOCAL
Warm	5%	6%	17%
Cold	51%	14%	6%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	20%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	12%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	11%
Acoustic and Oceanographic Research (ONR)	Navy Testing	10%
At-Sea Sonar Testing	Navy Testing	7%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	6%
Unmanned Underwater Vehicle Testing	Navy Testing	5%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat	CA Central Coast (All)	25	111	1	-	-
Critical Habitat	CA North Coast (All)	0	1	-	-	-
Critical Habitat	Channel Islands Area (All)	30	141	2	-	-
Critical Habitat	San Francisco Monterey Bay Area (All)	28	295	4	-	-
F-BIA-C	West Coast (3,4,5,6,7,8,9,10,11)	7	28	1	-	-
F-BIA-P	West Coast (3,4,5,6,7,8,9,10,11)	40	214	3	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Gray Whale - Western North Pacific DPS

There are two populations of gray whales that occur in the California portion of the action area: the ESA-listed Western North Pacific DPS and the non-listed Eastern North Pacific DPS. Both stocks are migratory and most likely to be in the California portion of the action area during their migrations from winter to spring within 10 km (5.4 nmi) of the coast. Some gray whales transit further offshore in Southern California when making straight line transits south of Point Conception, California to and from Mexico.

Since these DPSs are not separated geographically estimated impacts had to be proportioned between the two populations. Some gray whales that feed off Sakhalin Island in summer migrate east across the Pacific to the west coast of North America in winter, while others migrate south

to waters off Japan and China. However, the proportion of gray whales off California from each DPS at any given time is largely unknown, and likely varies over time. For example, Urbán R.J. et al. (2019) reported the number of whales documented moving between the western and eastern populations represents 14% of gray whales identified off Sakhalin Island and Kamchatka, while Cooke et al. (2019) estimated the fraction of the Western North Pacific DPS that migrates to the Eastern North Pacific DPS is between 45-80%. Given the estimated 290 whales in the Western North Pacific DPS and 26,960 in the Eastern North Pacific DPS (Carretta 2021b), at most (assuming 100% migration to the east) about 1% of the Western North Pacific DPS would mix with the Eastern North Pacific DPS. Thus, for NAEMO purposes, it was assumed that 1% of all gray whale impacts from HCTT sonar and explosives within SOCAL RC/NOCAL RC are to whales from the ESA-listed Western North Pacific DPS (Chip Johnson, Navy US Pacific Fleet, pers. comm. to Ron Salz, NOAA Fisheries, Office of Protected Resources, December 16, 2024).

Estimated exposures to the gray whale western North Pacific DPS from the use of sonar and explosives as part of the proposed action are shown in Table 41. On average, individuals in the Western North Pacific DPS would be impacted by HCTT acoustic stressors less than once per year, with the large majority of exposures resulting in TTS or a behavioral response (Navy 2025f). A very small number of auditory injuries (i.e., less than one per year) are also predicted for this DPS from acoustics impacts. These impacts are most likely to occur in the cold season when gray whales would be seasonally in the area during migration (Navy 2025f).

Table 41. Estimated Effects to the Gray Whale Western North Pacific DPS over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	(1)	(1)	0	-	-
Explosive	Navy Testing	2	(1)	0	-	-
Sonar	Navy Training	18	28	(1)	-	-
Sonar	Navy Testing	50	67	1	-	-
Sonar	USCG Training	(1)	-	-	-	-
Maximum Annual Total		**72**	**97**	**2**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
290	0.59	0.01

	Percent of Total Effects	
Season	SOCAL	PMSR
Warm	0%	0%
Cold	97%	3%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
At-Sea Sonar Testing	Navy Testing	21%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	19%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	19%
Undersea Warfare Testing	Navy Testing	15%
Unmanned Underwater Vehicle Testing	Navy Testing	10%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	5%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Sei Whale

Estimated sei whale exposures from the use of sonar and explosives as part of the proposed action are shown in Table 42 and Table 43 for the Eastern North Pacific and Hawaii stocks,

respectively. On average, sei whales from either stock would be impacted by HCTT acoustic stressors less than once per year, with the large majority of exposures resulting in TTS or a behavioral response (Navy 2025f). A very small number of auditory injuries (i.e., less than two per year) are also predicted for each stock from acoustics impacts. The majority of impacts are from sonar exposures including Anti-Submarine Warfare training activities and Acoustic and Oceanographic Research testing activities.

Table 42. Estimated Effects to the Eastern North Pacific Stock of Sei Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	5	1	0	-	-
Explosive	Navy Testing	2	2	(1)	-	-
Sonar	Navy Training	38	151	1	-	-
Sonar	Navy Testing	37	65	(1)	-	-
Sonar	USCG Training	1	-	-	-	-
Maximum Annual Total		**83**	**219**	**3**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
864	0.35	0.00

	Percent of Total Effects			
Season	SOCAL	PMSR	NOCAL	High Seas
Warm	30%	5%	5%	2%
Cold	42%	8%	7%	1%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	17%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	13%
Acoustic and Oceanographic Research (ONR)	Navy Testing	12%
Small Joint Coordinated Anti-Submarine Warfare	Navy Training	9%
Composite Training Unit Exercise (Strike Group)	Navy Training	7%
Intelligence, Surveillance, Reconnaissance (NAVWAR)	Navy Testing	6%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Table 43. Estimated Effects to the Hawaii Stock of Sei Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	1	(1)	0	-	-
Explosive	Navy Testing	0	0	-	-	-
Explosive	USCG Training	-	0	-	-	-
Sonar	Navy Training	25	173	(1)	-	-
Sonar	Navy Testing	11	41	(1)	-	-
Sonar	USCG Training	1	-	-	-	-
Maximum Annual Total		**38**	**215**	**2**	**-**	**-**

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
452	0.56	0.00

	Percent of Total Effects	
Season	HRC	High Seas
Warm	30%	1%
Cold	65%	4%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	37%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	18%
Acoustic and Oceanographic Research (ONR)	Navy Testing	9%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Sperm Whale

Estimated sperm whale exposures from the use of sonar and explosives as part of the proposed action are shown in Table 44 and Table 45 and for the California, Oregon, and Washington stock and the Hawaii stock, respectively. On average, sperm whales from either stock would be impacted by HCTT acoustic stressors less than once per year, with the large majority of exposures resulting in TTS or a behavioral response (Navy 2025f). A very small number of auditory injuries (i.e., less than one or two per year) are also predicted for each stock from acoustics impacts. The majority of impacts to sperm whales are from sonar exposures during Anti-Submarine Warfare training activities.

Table 44. Estimated Effects to the California, Oregon, and Washington Stock of Sperm Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	2	4	(1)	-	-
Explosive	Navy Testing	2	1	(1)	-	-
Explosive	USCG Training	0	-	-	-	-
Sonar	Navy Training	2,133	758	(1)	-	-
Sonar	Navy Testing	834	129	-	-	-
Sonar	USCG Training	28	-	-	-	-
Maximum Annual Total		**2,999**	**892**	**3**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
4,549	0.86	0.00

	Percent of Total Effects			
Season	SOCAL	PMSR	NOCAL	High Seas
Warm	32%	6%	5%	2%
Cold	38%	9%	5%	2%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	20%
Medium Coordinated Anti-Submarine Warfare	Navy Training	18%
Small Joint Coordinated Anti-Submarine Warfare	Navy Training	7%
Composite Training Unit Exercise (Strike Group)	Navy Training	6%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Table 45. Estimated Effects to the Hawaii Stock of Sperm Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	(1)	-	-	-	-
Explosive	Navy Training	2	1	(1)	-	-
Explosive	Navy Testing	0	(1)	-	-	-
Sonar	Navy Training	939	354	0	-	-
Sonar	Navy Testing	288	56	0	-	-
Sonar	USCG Training	7	-	-	-	-
Maximum Annual Total		**1,237**	**412**	**1**	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
6,062	0.27	0.00

	Percent of Total Effects	
Season	HRC	High Seas
Warm	43%	2%
Cold	51%	4%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	29%
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	10%
Submarine Sonar Maintenance and Systems Checks	Navy Training	9%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

False Killer Whale – Main Hawaiian Islands Insular DPS

Estimated exposures to the false killer whale Main Hawaii Islands Insular DPS from the use of sonar and explosives as part of the proposed action are shown in Table 46. On average, individuals from this DPS would be impacted by HCTT acoustic stressors less than once per year, with all exposures resulting in either TTS or a behavioral response (Navy 2025f). An estimated 19% of all impacts are predicted to occur within false killer whale critical habitat.

There are no auditory or non-auditory injuries or mortalities predicted for this DPS from HCTT acoustics impacts. The majority of impacts to this DPS are from sonar exposures during Anti-Submarine Warfare training activities, and very few (i.e., less than one per year) impacts are predicted from HCTT explosives (Navy 2025f).

Table 46. Estimated Effects to the Main Hawaiian Islands Insular Stock DPS of False Killer Whales over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	-	0	-	-	-
Explosive	Navy Testing	(1)	(1)	-	-	-
Sonar	Navy Training	68	54	-	-	-
Sonar	Navy Testing	32	9	-	-	-
Sonar	USCG Training	4	-	-	-	-
Maximum Annual Total		**105**	**64**	-	-	-

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
138	1.22	0.00

	Percent of Total Effects
Season	HRC
Warm	53%
Cold	47%

Activities Causing 5 Percent or More of Total Effects	Category	Percent of Total Effects
Medium Coordinated Anti-Submarine Warfare	Navy Training	32%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	7%
Anti-Submarine Warfare Torpedo Exercise - Submarine	Navy Training	6%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat	Critical Habitat (All)	31	1	-	-	-
S-BIA-C	Main Hawaiian Islands (All)	8	-	-	-	-
S-BIA-P	Main Hawaiian Islands (All)	54	12	-	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Hawaiian Monk Seal

Estimated Hawaiian monk seal exposures from the use of sonar and explosives as part of the proposed action are shown in Table 47. On average, Hawaiian monk seals would be impacted by HCTT acoustic stressors less than once per year, with the large majority of exposures resulting a behavioral response (77%) or TTS (22%) (Navy 2025f). A small number of auditory injuries (i.e., less than four per year) are also predicted for this species from acoustics impacts. The majority of these impacts are from sonar exposures during Anti-Submarine Warfare, Surface Ship Object Detection, and Submarine Navigation training activities. One non-auditory injury due to explosives during Mine Warfare and Expeditionary Warfare conducted at Puuloa Underwater Range is also predicted every seven years of the proposed action. The pre-event activity-based mitigation (Section 2.4.2) prescribed for these activities may further reduce the potential for injurious impacts (Navy 2025f).

Table 47. Estimated Effects to Hawaiian Monk Seals over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	14	21	3	(1)	0
Explosive	Navy Testing	10	11	1	-	-
Explosive	Army Training	(1)	-	-	-	-
Sonar	Navy Training	590	123	0	-	-
Sonar	Navy Testing	75	43	(1)	-	-
Sonar	USCG Training	1	-	-	-	-
Maximum Annual Total		**691**	**198**	**5**	**1**	**0**

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
1,605	0.56	0.00

	Percent of Total Effects	
Season	**HRC**	**High Seas**
Warm	45%	1%
Cold	54%	1%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Surface Ship Object Detection	Navy Training	28%
Anti-Submarine Warfare Torpedo Exercise - Ship	Navy Training	13%
Anti-Submarine Warfare Torpedo Exercise - Submarine	Navy Training	9%
Submarine Navigation	Navy Training	8%
Mine Countermeasures - Ship Sonar	Navy Training	5%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat	Critical Habitat (All)	460	69	2	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Guadalupe Fur Seal

Estimated Guadalupe fur seal exposures from the use of sonar and explosives as part of the proposed action are shown in Table 48. The large majority of impacts will occur within the SOCAL RC during Anti-Submarine Warfare training activities. Most impacts would be either behavioral responses or TTS due to sonar used in testing activities, with a small number of auditory (i.e., 25 annually), and non-auditory (i.e., two annually) injuries also anticipated in a maximum year of training and testing. NAEMO also predicts one Guadalupe fur seal mortality from explosives training activities over every seven years of the proposed action. On average, individual Guadalupe fur seals would be impacted by HCTT acoustic stressors about three to four times per year (Navy 2025f).

Table 48. Estimated Effects to Guadalupe Fur Seals over a Maximum Year of Proposed Activities (Navy 2025f)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	1	-	-	-	-
Explosive	Navy Training	50	60	4	1	(1)
Explosive	Navy Testing	73	90	12	2	-
Explosive	USCG Training	(1)	-	-	-	-
Sonar	Navy Training	217,808	77,386	32	-	-
Sonar	Navy Testing	44,414	3,814	3	-	-
Sonar	USCG Training	3,852	4	-	-	-
Maximum Annual Total		**266,199**	**81,354**	**51**	**3**	**1**

Population Abundance Estimate	Annual Effects per Individual	Annual Injurious Effects per Individual
96,468	3.60	0.00

	Percent of Total Effects		
Season	SOCAL RC	PMSR	NOCAL RC
Warm	40%	7%	1%
Cold	42%	7%	1%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Anti-Submarine Warfare Tracking Exercise - Ship	Navy Training	31%
Medium Coordinated Anti-Submarine Warfare	Navy Training	13%
Small Joint Coordinated Anti-Submarine Warfare	Navy Training	8%
Submarine Sonar Maintenance and Systems Checks	Navy Training	7%
Composite Training Unit Exercise (Strike Group)	Navy Training	7%
Surface Ship Sonar Maintenance and Systems Checks	Navy Training	6%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Response Analysis

Given the above estimated exposures of ESA-listed marine mammals to acoustic and explosives stressors associated with the proposed action, in this section we describe the range of responses to those exposures. This includes significant behavioral responses, sound-induced hearing loss (i.e., TTS and AINJ), non-auditory injury, as well as other possible responses (e.g., physiological stress) that marine mammals may exhibit when exposed to sound fields from these sources. Masking can also occur when the perception of a biologically-important sound is interfered with by a second sound (e.g., noise from training and testing).

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 marine mammals. 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. 2007; Southall et al. 2021). 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 source of the sound. Studies on captive odontocetes suggest some species may learn to reduce their hearing sensitivity (presumably to protect their hearing) when warned of an impending intense sound exposure (Nachtigall and Supin 2013; Nachtigall et al. 2016; Finneran 2018). It is unknown if marine mammals in the wild would also suppress their hearing if they could anticipate an impending, intense sound, or during a prolonged exposure (even if unanticipated).

Our aim with this response analysis is to assess the potential responses that might reduce the overall fitness of individual ESA-listed marine mammals. Such responses would likely rise to the level of take under the ESA. 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" (16 U.S.C. §1532(19)). NMFS has a definition for the term harass, which is: *to create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering* (NMFS Policy Directive 02-110-19). Harm is defined as an act that actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering (50 CFR §222.102).

Non-Auditory Injury

From our exposure analysis above, proposed HCTT activities involving explosives will result in a small number (i.e., less than one per year) of non-auditory injuries to fin whales, humpback whales Mexico DPS, Hawaiian monk seals, and up to two non-auditory injuries per year to Guadalupe fur seals. Non-auditory injuries are not anticipated to occur in any other ESA-listed marine mammal species as a result of the proposed action.

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; Richmond et al. 1973; General 1991). 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. 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.

Relatively little is known about auditory system trauma (i.e., tissue damage) in marine mammals resulting from explosive exposure, although it is assumed that auditory structures would be vulnerable to blast injuries. Here, we consider primary blast injury to auditory tissues, which is distinct from TTS or AINJ discussed below. 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.

For the purposes of this analysis, we assume that marine mammals experiencing a non-auditory injury would be temporarily impaired as a result. While the animal is recovering from its injury, though there is some uncertainty, we assume the animal's ability to conduct important life functions (e.g., breeding, feeding) would be diminished, but that the animal would survive. We do not have information available to determine how long an injured marine mammal would remain impaired. For a slight lung injury, which defines the threshold for onset of non-auditory injury, it is reasonable to assume the animal would recover within several months. A more severe injury could take longer until full recovery, which could result in a decrease in the animal's reproductive potential, particularly if it occurs to a mature female. For marine mammals with long life spans (e.g., large whales), it is unlikely that a single injury would result in a significant impact on the exposed individual's life-time reproductive potential.

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 level, temporal pattern, and duration. Higher-level sound exposures are more likely to cause PTS or other auditory injury. 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. For marine mammals, auditory injury (AINJ) is considered to be possible when sound exposures are sufficient to produce 40 dB of TTS measured approximately four minutes after exposure (Navy 2024h). 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, depending on the ability of the individual to hear at the affected frequencies.

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.

Also important to consider is the potential for repeat instances of TTS due to exposure to acoustic stressors. 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. It is possible that repeat instances of TTS could result in PTS. This has been shown in terrestrial animals (e.g., Kujawa and Liberman 2009; Lin et al. 2011), and in one case, marine mammals as well (Kastak et al. 2008).

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. Small amounts of TTS (a few dB) typically begin to recover immediately after the sound exposure and may fully recover in minutes, while larger amounts of TTS take longer to recover. Studies have also found substantial individual variation both in the amount of TTS produced by similar SELs (Kastelein et al. 2012; Popov et al. 2013), and in recovery from similar TTS (Finneran 2015). Unlike TTS, AINJ (including PTS) has the potential to affect aspects of an animal's life functions, including those that do not overlap in time and space with the proposed action.

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 Action Proponents use sonars operating at a wide range of frequencies (i.e., from low frequency sources to extremely high frequency

sources). Marine mammals that experience PTS or 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. Sounds at frequencies well below the region of best sensitivity are generally less hazardous than those near the region of best sensitivity (Finneran and Schlundt 2013; Kastelein et al. 2020; Gransier and Kastelein 2024). In some cases, sound exposures of a narrow frequency range can produce TTS over a large frequency range (Finneran et al. 2007; Kastelein et al. 2020). 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. The ability to detect anthropogenic sounds may be important to provide information on the location and direction of human activities, and may provide a warning regarding nearby activities (e.g., vessels) that may be hazardous. The ability to detect conspecifics is also important for mating and mother-calf communication. Depending on the severity and frequency range affected, PTS and TTS may also impact foraging success for sperm whales and other echolocating odontocetes that rely on sound to find prey. 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.

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 little to no effect on an animal's fitness. Because narrow-band active sonar is typically transmitted at a specified frequency, animals experiencing TTS or AINJ from narrow-band sonar would likely only experience threshold shifts around that particular frequency. By contrast, for animals experiencing TTS or AINJ from broadband sonar sources or explosives, which are considered broadband, a greater frequency band would likely be affected. Since a greater frequency band could be affected due to broadband sound sources, there is an increased chance that the resulting hearing impairment will affect frequencies utilized by animals for acoustic cues.

Based on the best available information, we expect instances of TTS from HCTT sonar, air guns, and explosives to be short-term and of relatively low severity because of animal avoidance and the transient nature of most sonar, air guns and explosive sources. This conclusion is supported by studies indicating that even following relatively prolonged periods of sound exposure resulting in TTS, recovery occurs quickly (Finneran 2015). Additionally, we do not anticipate that marine mammals will experience long duration or repeat exposures within a short period of time due to the species wide ranging life histories and the fact that long duration (i.e., more than one day) HCTT activities also occur over large geographic areas (i.e., both the marine mammals and the activity are moving within the action area, most likely not in the same direction). This decreases the likelihood that animals and HCTT activities will co-occur for extended periods of

time or repetitively over the duration of an activity. From our exposure analysis above, the average number of annual TTS impacts per individual was less than one for all marine mammals species (or DPSs). Although some individual marine mammals may experience TTS more often than the average, the number of impacts estimated per population suggests that the maximum number of TTS exposures for any given individual within a population will be relatively small (i.e., likely less than five per year). One exception to this may be acoustic exposures to the resident population of fin whales that occur within SOCAL RC throughout most or all of the year. We do not have any additional information to determine how (or if) acoustic impacts to this resident population may differ from impacts to migratory fin whale populations in the action area. It is possible that resident fin whales with years of experience with military sonar have become acclimated to military sonar or have learned behavioral responses to avoid or minimize acoustic impacts. In general, any adverse effects resulting from TTS are expected to be relatively minor and short-term. For these reasons, we do not anticipate that instances of TTS from HCTT activities will result in long-term fitness consequences affecting the reproduction or survival of ESA-listed marine mammals.

Unlike TTS, AINJ could be permanent (e.g., PTS) meaning the effects of the injury last well beyond the duration of the proposed action and outside of the action area as animals migrate. While hearing loss in marine mammals resulting from exposure to PTS-causing sound levels is not expected to deafen the animals, we expect it would have some effect on the hearing ability of the whales in the frequencies of the sound that caused the damage. For the purposes of this assessment, we assume that the frequencies affected overlap with those utilized by animals for acoustic cues. Therefore, PTS may interfere with a marine mammals' ability to hear sounds produced by ships, construction activities, seismic surveys, or communication signals of conspecifics. The ability to detect anthropogenic sounds may be important to provide information on the location and direction of human activities, and may provide a warning regarding nearby activities that may be hazardous. The ability to detect conspecifics is important for socialization, mating and mother-calf communication. Depending on the severity, permanent hearing impairment has the potential to affect individual marine mammal survival and reproduction, although the available information does not allow us to quantify the proportion of AINJ that will result in mild, moderate or high severity PTS, nor to evaluate how such threshold shifts directly relate to individual marine mammal fitness.

We anticipate that a large majority of PTS exposures resulting from the proposed action would be of mild severity due to conservative assumptions incorporated into the NAEMO modeling of quantitative impacts (Navy 2024h) and the proposed mitigation measures described in Section 2.4.2. Individual marine mammals experiencing a minor degree of PTS could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and should still be able to detect threats with enough time to avoid injury. Similarly, while odontocetes (e.g., sperm whales and false killer whales) with hearing loss could also be less efficient at finding prey using echolocation (Kloepper et al. 2012), we expect individuals with a minor degree of

PTS will still be able to forage successfully. The majority of the effects would likely be in the mid-frequency range, below the lower limit of false killer whale echolocation signals but within the range of sperm whale echolocation signals. Our conclusion that odontocetes with mild severity PTS will still be able to perform these important life functions assumes that these individuals are reasonably healthy at the time of exposure, and are not compromised or impacted by other stressors such as entanglement, vessel strike, or nutritional deficiency.

Although we anticipate that a large majority of PTS exposures resulting from HCTT activities will be of mild severity, it remains possible that some exposures could result in more severe incidents of PTS, with greater fitness consequences for those individual whales exposed. As discussed above, higher severity instances of PTS that can lead to fitness consequences are more likely to occur from exposure to broadband sound sources (e.g., explosives), since there is an increased chance that the resulting hearing impairment will affect frequencies utilized by animals for acoustic cues. From our exposure analysis above, fin whales are expected to be exposed to 54 AINJ annually from all sources during a maximum year of HCTT activities, with 11 of these exposures resulting from explosive sound sources. Guadalupe fur seals are expected to be exposed to 54 AINJ annually from all sources during a maximum year of HCTT activities, with 16 of these exposures resulting from explosive sound sources. None of the other marine mammals analyzed in this opinion are expected to be exposed to more than four AINJ from explosives annually in a maximum year (Mexico DPS humpback whale 4; Hawaiian monk seals 4; blue whale 3; all others 2 or less).

Therefore, while moderate to high severity instances of PTS are expected to be rare, in general, the likelihood of this occurring is greater for fin whales and Guadalupe fur seals given the relatively large number of AINJ impacts estimated for this species. To a lesser extent, a small number of Mexico DPS humpback whales, Hawaiian monk seals and blue whales could also experience moderate to high severity PTS. For the other marine mammals, all instances of PTS resulting from the proposed action are anticipated to be of mild severity, given the relatively small number of estimated AINJ impacts to these species (and specifically impacts from explosives), the conservative assumptions incorporated into the NAEMO modeling of quantitative impacts (Navy 2024h), and the proposed mitigation measures.

Masking

Masking occurs when one sound, distinguished as the "noise," interferes with the detection or recognition of another sound. 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). 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, changing frequency, or cessation of vocalization) and behavioral changes (e.g., cessation of foraging,

leaving an area) on the part of both signalers and receivers, in an attempt to compensate for noise levels (Erbe et al. 2015).

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). Vocal changes can be temporary or permanent (Tennessen and Parks 2016). In some cases, 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). There is also evidence that odontocetes shift the main energy of their echolocation signals outside of the band of masking noise (Thomas and Turl 1990; Romanenko and Kitain 1992; Erbe et al. 2015). 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).

Masking could occur as a result of sonar and other transducers. Low-frequency active sonar could result in masking if there is overlap between the sound source frequencies and mysticete vocalizations (e.g., humpback whales). In the presence of low-frequency active sonar, humpback whales have been observed to increase the length of their songs (Miller et al. 2000; Fristrup et al. 2003), possibly due to the overlap in frequencies between the whale song and the low-frequency active sonar. Odontocetes that use echolocation to hunt may experience masking of the echoes needed to find prey when exposed to low or mid-frequency sonar. High-frequency sonar sources have a smaller zone of potential masking effects because sounds at higher frequencies attenuate more rapidly in the water compared to low or mid-frequency sources (Navy 2025f). While high-frequency (10-100 kHz) sonar has the potential to mask marine mammal vocalizations under certain conditions, reduction in available communication space or ability to locate prey is less likely because of the smaller zone of effect and lower sources levels, as compared to mid-frequency anti-submarine warfare sonar (Navy 2025f).

For military sonar systems with low duty cycles, relatively short duration, and narrow bandwidth that do not overlap with vocalizations for most marine mammal species, masking would be limited as compared to continuous sources (e.g., vessel noise). Masking is less likely or is expected to be less impactful when the noise is intermittent, such as low-duty cycle sonar systems or impulsive noise, compared to when the noise is continuous, such as vessel noise, high-duty cycle sonar, or continuous active sonar. This is because for intermittent noise, the signal of interest can be detected during the quiet periods (e.g., "gap" listening) between noise events (Navy 2025f).

High duty cycle or continuous active sonar systems have more potential to mask vocalizations, particularly for mid or high-frequency cetaceans (e.g., ESA-listed sperm whales and false killer whales) given the overlap in vocalization frequency with these sonar systems. These sonar systems transmit more frequently (greater than 80% duty cycle) than traditional sonars, but at a substantially lower source level. 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. von Benda-Beckmann et al. (2021) modeled the effect of pulsed and continuous 1-2 kHz active sonar on sperm whale echolocation clicks, and found that the presence of upper harmonics in the sonar signal increased masking of clicks produced in the search phase of foraging compared to buzz clicks produced during prey capture. Different levels of sonar caused intermittent to continuous masking (120-160 dB re: 1 μPa2, respectively), but varied based on click level, whale orientation, and prey target strength. Continuous active sonar resulted in a greater percentage of time that echolocation clicks were masked compared to pulsed active sonar.

Continuous noise at the same frequency of communicative vocalizations may cause disruptions to communication, social interactions, acoustically mediated cooperative behaviors (Sørensen et al. 2023) such as foraging and mating, and can also trigger changes to vocalization amplitude and frequency, reduced communication space (Brumm and Slabbekoorn 2005; Hotchkin and Parks 2013). 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 recruitment if masking interferes with reproductive activities or mother-calf communication (Gordon et al. 2003), and a potential decrease in survivorship if predator vocalizations (e.g., killer whales) are masked (Brumm and Slabbekoorn 2005).

Results of the recent studies are variable in terms of finding differences between continuous and non-continuous active sonar. von Benda-Beckmann et al. (2021) provides initial support for the hypothesis that continuous active sonars have more potential to mask vocalizations, particularly for sperm whales, as compared to pulsed sources. However, at this time, information regarding the behavioral effects resulting from continuous active sonar, and the consequences of those effects on individuals and populations, is lacking.

The proposed action includes training and testing activities involving mid-frequency continuous active sonar. Navy sonar bin MF1C (previously MF11) includes hull-mounted surface ship sonars with active duty cycle greater than 80%. HCTT proposed activity levels for this continuous source of active sonar are shown above in Section 2.5.1, Table 20. Compared to HSTT Phase III, the proposed action includes approximately 50% more hours of high duty cycle (MF1C) hull-mounted sonar in the California portion of the action area and greater than 60% more hours in the Hawaii portion of the action area in a maximum year of training and testing activities (Navy 2024d).

Although NAEMO does not estimate impacts by sonar bin, it does provide a breakdown of impacts by training or testing activity. The primary mission area most closely associated with the use of MF1C sonar is anti-submarine warfare. Thus, species with a large number of predicted impacts from anti-submarine warfare activities likely have greater potential to be affected by masking compared to species with a relatively small number of impacts from these activities. Blue, fin, and humpback whales have a relatively large number and proportion (i.e., between 30-40%) of estimated impacts from anti-submarine activities. However, as low-frequency cetaceans, these species are less susceptible to masking effects from mid-frequency sonar given the minimal overlap in frequencies of sounds that may be important for their survival.

As noted above, odontocete species are more susceptible to masking vocalization given the overlap in vocalization frequency with these sonar systems. For sperm whales impacted by HCTT sonar, between 39% (Hawaii stock) and 45% (California, Oregon, and Washington Stock) of the estimated impacts are from a combination of the following anti-submarine activities: Tracking Exercise – Ship, Medium Coordinated Event, and Small Join Coordinated Event. Masking can impact the ability of sperm whales to locate prey since their echolocation clicks, which range from 10 Hz–30 kHz, overlap with the frequencies of MF1C sonar.

Nearly one-half of the estimated sonar impacts to MHI IFKWs are from a combination of the following anti-submarine activities: Medium Coordinated Event, Torpedo Exercise – Ship, and Torpedo Exercise – Submarine. In addition to anti-submarine warfare activities, other activities that could overlap with MHI IFKW critical habitat and include high or continuous duty cycle sonars are Multi-Domain Unmanned Autonomous Systems Training, Vehicle Testing, Long Range Acoustic Communications Testing, and Submarine Sea Trials - Weapons System Testing. The range of MHI IFKWs whistle frequencies, which have been recorded between 4-10 kHz, falls just outside of the narrow band frequency of Navy surface ship hull-mounted mid-frequency sonar (i.e., MF1 low duty cycle; MF1C high duty cycle), which is centered around or about 3 kHz. Thus, the potential for masking of MHI IFKW whistle sounds used for communication from HCTT sonar would likely be limited to other (i.e. not hull-mounted) Navy mid-frequency or broadband sonar sources within the 4-10 kHz range. False killer whale echolocation clicks have been recorded between 17-104 kHz (Au et al. 1995; Oswald et al. 2003; Madsen et al. 2004; Thode et al. 2016), which is completely outside of the Navy mid-frequency sonar range, but falls within the high-frequency sonar range. As discussed above, masking effects from high-

frequency sonar, including high-duty cycle sources, would likely be minimal given the lower source levels and smaller zone of effects (due to higher rates of attenuation), as compared to mid-frequency anti-submarine warfare sonar. Higher frequency sounds in the range of false killer whale echolocation clicks would be expected to undergo significant absorption loss as they propagate through seawater, corresponding to a rapidly diminished acoustic signal intensity over distance.

Unlike continuous or high-duty cycle systems, impulsive sound sources, including explosions, are intense and typically short in duration. Since impulsive noise is intermittent, the length of the gap between sounds (duty-cycle) and received level are pertinent when considering the potential for masking. Impulsive sounds with lower duty cycles or lower received levels are less likely to result in masking than higher duty cycles or received levels. There are no direct observations of masking in marine mammals due to exposure to explosive sources. Potential masking from explosive sounds or weapon noise is likely similar to masking studied for other impulsive sounds, such as air guns or pile-driving.

Masking of mysticete calls could occur due to the overlap between their low-frequency vocalizations and the dominant frequencies of impulsive sources (Castellote et al. 2012; Nieukirk et al. 2012). For example, blue whale feeding/social calls increased when seismic exploration was underway (Di Lorio and Clark 2010), indicative of a possible compensatory response to masking effects of the increased noise level. However, mysticetes that call at higher rates are less likely to be masked by impulsive noise with lower duty cycles (Clark et al. 2009) because of the decreased likelihood that the noise would overlap with the calls, and because of dip listening. Field observations of masking effects such as vocal modifications are difficult to interpret because when recordings indicate that call rates decline, this could be caused by (1) animals calling less frequently (actual noise-induced vocal modifications), (2) the calls being masked from the recording hydrophone due to the noise (e.g., animals are not calling less frequently but are being detected less frequently), or (3) the animals moving away from the noise, or any combination of these causes (Blackwell et al. 2013; Cerchio et al. 2014).

Behavioral and Physiological Stress Responses

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 depends 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).

Behavioral Responses to Sonar and Other Transducers

Sonar and other transducers can range in frequency (from less than 1 kHz to over 200 kHz) 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 be 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.

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. 2007; 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 (Southall et al. 2007; Southall et al. 2016). More minor reactions have also been observed including alerting to the sound source and startle responses. Based on their review, Southall et al. (2016) found that many, but not all responses of cetaceans to sonar reported in the literature have been relatively mild and/or brief.

It should be noted that the responses of animals to real world military 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. This is in contrast to what would be expected during some military activities (e.g., MTEs) involving sonar where multiple vessels are operating concurrently in close proximity, during an exercise that lasts for an extended period of time (i.e., multiple days to weeks). 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. Additionally, while the experimental 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 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. In addition, Forney et al. (2017) noted that species that respond to noise from military sonars by avoiding an area are unlikely to be observed in these studies based on traditional methods (e.g., Lookouts or passive acoustic monitoring) because animals often react at distances far greater than the detection range of these methods.

Passive acoustic monitoring and visual observational behavioral response studies have also 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 (Mccarthy et al. 2011; Tyack et al. 2011b; Moretti et al. 2014; Deakos and Richlen 2015; Martin et al. 2015b;

Mobley and Deakos 2015; Henderson et al. 2016; Manzano-Roth et al. 2016). However, passive acoustic studies are limited to observations of vocally-active marine mammals and visual studies are limited to what can be observed at the surface.

The responses of large whales 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. 2015b; 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 (Goldbogen et al. 2013; DeRuiter et al. 2017). These findings indicate that the behavioral state of the animal plays a role in the type and severity of a behavioral response. 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 showed that while blue and fin whale lunge rates remained similar to baseline regardless of exposure levels, humpback whales – which were exposed the highest sonar sound levels – showed a greater degree of feeding disruption during and up to 15 minutes after sonar exposure. In a Norwegian behavioral response study, humpback whales responded more severely to killer whale vocalization playbacks than they did to the sonar playbacks, and may have been habituated slightly between the first and second sonar exposures (Sivle et al. 2015). Several humpback whales have been observed during aerial or visual surveys during Navy training events involving sonar (Sivle et al. 2015; Wensveen et al. 2017; Harris et al. 2019; Henderson et al. 2019). No avoidance or other behavioral responses were ever noted, even when the whales were observed within five kilometers 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., Smultea et al. 2009; Mobley 2011; Mobley and Pacini 2012). Studies from 1997 to 1998 conducted under the Navy's Low-Frequency Sound Scientific Research Program 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.

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). However, they were unable to determine if suppression of calls reflected a change in their feeding performance or abandonment of foraging behavior.

Harris et al. (2019) used data from passive acoustic monitoring of Navy training ranges to test minke whale responsiveness to naval sonar on the Navy's PMRF located off Kauai, Hawaii. They found that the distribution of calling whales changed in both years surveyed in the vicinity of the center of ship activity during naval warfare training activities, as compared to before such activities started. The avoidance behavior, as evidenced by the redistribution of calls, extended for approximately five days after the end of the training activity in both years. As the data are from passive acoustic monitoring, it is unknown whether this change in the probability of acoustic presence across the range is a result of whales moving away from the range or ceasing calling, or perhaps a combination of both. Henderson et al. (2019) studied humpback whale behavior in response to mid-frequency active sonar on the PMRF. While some behavioral response may have occurred in individuals exposed to mid-frequency active sonar, the authors noted that this response was limited, with the whales focused on breeding behavior. Durbach et al. (2021) analyzed acoustic tracks from minke whales detected on the PMRF in Hawaii in three years before, during, and after major Navy training exercises. Results of this study showed that minke whales moved faster and movements were more directed during periods of active sonar. Minke whales also avoided the locations of the ships producing the sonar, and were more likely to cease calling during sonar.

Southall et al. (2019a) measured the degree of behavioral responses in blue whales to controlled exposure to mid-frequency military sonar off the southern California coast. More than 50% of blue whales in deep-feeding states responded to exposure experiments, whereas no changes in behavior were identified in shallow-feeding blue whales. Responses were generally brief, of low to moderate severity, and highly dependent on exposure context such as behavioral state, source-to-whale horizontal range and prey availability (Southall et al. 2019a). Using a similar experimental design, Southall et al. (2023) studied the behavioral responses of fin whales to military sonar and other stimuli. Behavioral changes were detected in 5 of 15 whales during controlled exposure experiments using mid-frequency active sonar or pseudo-random noise of similar frequency, duration and source and received level. No changes were detected during six control (no noise) sequences. Overall responses were more limited in occurrence, severity and duration than in blue whales and were less dependent upon contextual aspects of exposure and more contingent upon exposure received level (Southall et al. 2023). Result from Stanistreet et al. (2022) suggest that beaked and sperm whales ceased foraging in the vicinity of a multinational naval exercise conducted off eastern Canada.

A growing number of behavioral response studies have been conducted on odontocete responses to active sonar transmissions or controlled exposure playback of simulated sonar on various

military ranges (Claridge and Durban 2009; Moretti et al. 2009; Mccarthy et al. 2011; Southall et al. 2012; Southall et al. 2013; Henderson et al. 2015; Wensveen et al. 2019; Isojunno et al. 2020). While these studies have focused primarily on non ESA-listed beaked whales (or other smaller species), results suggest that sperm whales are not as sensitive to anthropogenic sound sources as these smaller odontocetes (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 (Miller et al. 2009; Miller et al. 2012; Sivle et al. 2012; Isojunno et al. 2016). Sperm whales may be more sensitive to larger amounts of cumulative sound energy they receive, rather than higher sound amplitudes during sonar exercises. 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. 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).

Curé et al. (2025) examined the effects of MFAS received level and source distance on the behavioral responses of 14 tagged male sperm whales off northern Norway. Behavioral responses were scored using the severity scale from Southall et al. (2021), with probability and severity of behavioral responses (e.g., changes in vocal and dive behaviors, avoidance, cessation of feeding or resting, locomotion or orientation changes) increasing with higher received levels (maximum sound exposure level) and closer source proximities. From observations, modeling indicates that beyond 14 km (7.6 nmi) no significant behavioral responses are predicted regardless of received level. Using the same tagged whales as Curé et al. (2025), Wensveen et al. (2025) found that sperm whales increased their time in a non-foraging behavioral state or produced a decrease in buzzes (indicative of reduced prey capture) when foraging with MFAS exposure.

Baird (2013), Baird et al. (2014), and Baird et al. (2017a) 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 from 3.2 to 94.4 km. 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 m to 268 m) 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 pelagic population, leading Baird et al. (2016) 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.

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, and 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.

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). One study found that captive hooded seals reacted to 1–7 kHz 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. 2010); 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 kHz 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. 2015). 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. 2013). 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 2 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. 2015). 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 kHz tone and 1-4 kHz sweep at source levels of 165 dB re: 1 µPa caused the sea lions to haul out (Akamatsu et al. 1996). D'Amico (2013) studied Hawaiian monk seals responses to Navy mid-frequency active sonar by comparing animal movements during periods with and without sonar exercises. The majority of the time, the four monk seals were exposed to mid-frequency sonar transmissions while in the 15-36 km range bin, receiving a maximum receive level of 150 dB re: 1 µPa. Study results suggested that there were no significant impacts from mid-frequency active sonar on the Hawaiian monk seals tagged in the HRC during the 2010–2011 time period that the study was conducted (D'Amico 2013). Similar to the other taxonomic groups assessed, pinniped behavioral responses to sonar and other active acoustic sources 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.

Behavioral Responses to Explosives and Air Guns

As shown in our exposure analysis above, NAEMO predicts significant behavioral responses from explosives used in HCTT for all of the marine mammals considered in this opinion. Significant behavioral responses from Navy air guns are only predicted for Guadalupe fur seals, and at a very low level of occurrence (i.e., less than one exposure per year).

Impulsive signals, 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. 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-airgun 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 HCTT impulsive sources such as explosives. HCTT 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). The NAEMO quantitative effects analysis approach assumes that significant behavioral responses to solitary explosions would not occur due to the short duration of acoustic exposure from such explosions. The mitigation measures that would be implemented (such as exclusion zones) may further reduce the potential for significant behavioral responses to occur from exposure to solitary explosions.

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 (Richardson et al. 1985; McCauley et al. 2000; Gordon et al. 2003; Southall et al. 2007). 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. Depending on numerous factors (e.g., proximity, attentional focus, charge weight of blast, and experience of the animal) the behavioral responses of individuals may vary, and we would assume some animals would exhibit more of a reaction than others.

Migratory behavior seems to lead to a higher likelihood of response, with some species demonstrating more sensitivity than others. 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 in Australia showed avoidance behavior at ranges between three and 8 km from a seismic array during observational studies and controlled exposure experiments (McCauley et al. 1998; Dunlop et al. 2017), decreased their dive times and reduced their swimming speeds in another study (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 airgun 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 airguns 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. (1995) 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).

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 from a sparker at average received peak-to-peak measurements 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 airgun 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. 2013). In other studies, noise from seismic surveys or explosions was not found to result in any behavioral response from mysticetes (Gailey et al. 2007; Yazvenko et al. 2007; Vilela et al. 2016).

Madsen et al. (2006) and Miller et al. (2009) tagged and monitored eight sperm whales in the Gulf of America exposed to seismic airgun surveys. Sound sources were from approximately 2 to 7 nm 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 airguns 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 airgun 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 airgun impulsive sounds observed.

Physiological Stress Responses

Marine mammals naturally experience stress 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, and social interactions with conspecifics are also stressors, although they are natural components of an animal's life history. Anthropogenic activities, including acoustic and explosive sound sources, have the potential to provide additional physiological stress beyond those that occur naturally (Rolland et al. 2012; Fair et al. 2014; Meissner et al. 2015).

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.

The stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor (Moberg 2000). 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). Chronic elevations of glucocorticoids (i.e., weeks/months) may result in decreased growth, depressed immune system function, and suppression of reproduction (e.g., Sapolsky et al. 2000; Romero and Wikelski 2001). Yang et al. (2021) measured cortisol concentrations in blood samples of two captive bottlenose dolphins and found significantly higher levels after exposure to high sound level (140 dB re: 1 µPa) impulsive noise playbacks, compared to control and low sound levels (0 and 120 dB re: 1 µPa, respectively). Results suggest that repeated exposures or sustained stress response to impulsive sounds reduce an individual's fitness.

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; St Aubin and Dierauf 2001; Finneran and Branstetter 2013). 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.

What is known about the function of the various stress hormones is based largely upon observations of the stress response in terrestrial mammals. However, 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). Relatively little information exists on the linkage between anthropogenic sound exposure and stress in large whales, and even less information exists on the ultimate consequences of sound-induced stress responses (either acute or chronic).

Summary of Behavioral and Physiological Responses

In general, large whale behavioral responses to sonar appear to be fairly low to moderate severity across all received levels, with very few strong responses reported. Large whale 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. 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 cessation of sound

exposure. 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. 2007). As noted in Southall et al. (2007), 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. Other studies support the conclusion that behavioral responses to 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 to an animal (Southall et al. 2007; New et al. 2014; King et al. 2015; Villegas-Amtmann et al. 2015; Harris et al. 2017; U.S. Department of the Navy 2017; Farmer et al. 2018).

There is no HCTT activity involving sonar, air guns or explosives that is both long in duration (more than a day) and concentrated in the same location (e.g., within a few square miles). Given the dispersed nature of the proposed HCTT sonar use in space and time, we would expect that in most instances individual whales could 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. Similarly, long-term or chronic physiological stress responses that could result in fitness consequences are not anticipated given the types of acoustic sources and their spatial and temporal distribution throughout the action area.

Energetic costs to exposed individuals may be greater during MTEs and other large-scale military events that occur over multiple days (to weeks), over large ocean areas and involve multiple sonar platforms. During such events, cessation of foraging may last longer and whales may need to travel farther distances to resume foraging behaviors. However, there is a low likelihood that large whales and military readiness activities would co-occur for extended periods of time or repetitively over the duration of an activity or event. Proposed mitigation measures may further reduce the likelihood of this occurring.

As discussed above, large whales may be more susceptible to masking from continuous or high-duty cycle active sonar sources noise, which may cause disruptions to communication that can, in turn, result in a behavioral response. 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. In summary, the available information indicates a range of behavioral responses to sonar may occur, but most responses are expected to be brief, with the animal returning to baseline behavior shortly after the exposure is over.

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 impulsive sounds such as explosives and air guns. Significant behavioral responses are not anticipated from solitary explosions due to the extremely short duration of acoustic exposure

from such explosions. Military readiness activities involving multiple explosions over a short time could result in significant behavioral responses. Significant behavioral responses from air guns would be very rare (i.e., only predicted for Guadalupe fur seals, and at a very low level of occurrence) due to the characteristics and limited use of this acoustic stressor as part of the proposed action.

Mysticetes seem to be the most sensitive taxonomic group of marine mammals 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. 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 HCTT impulsive sound sources. However, the proposed impulsive sources would largely be stationary and short-term (i.e., instantaneous for explosives) as compared to sources in experimental studies, and so responses to military impulsive sounds would likely occur in closer proximity or not at all. The activity-based mitigation measures and mitigation areas that would be implemented for HCTT Phase IV may further reduce the potential for significant behavioral responses to occur from exposure to explosions and air guns.

Bioenergetic Models of Marine Mammal Responses to Acoustic Stressors

The long-term consequences due to individual behavioral reactions, instances of physiological stress, or auditory impacts can be difficult to predict because individual experience over time can create complex contingencies, especially for long-lived animals like marine mammals. To evaluate the potential consequences of behavioral responses and stress to affected animals, one should 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.

In recent years, population models have been developed, within the ONR Population Consequences of Disturbance (PCoD) framework (Pirotta et al. 2018), to link short-term effects to individuals due to anthropogenic stressors with long-term consequences to both the exposed individuals and the marine mammal populations they represent. PCoD models have been used to forecast the possible consequences of a range of disturbance scenarios (Keen et al. 2021). 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. Pirotta et al. (2021) developed a bioenergetic model to predict effects from military active sonar on a blue whale's daily energy intake. Activity budgets and movements were highly variable in space and time and among individuals, resulting in large variability in predicted energetic intake and costs. A portion (11%) of those simulated whales had negative net energy even after brief (e.g., 6-30 min) or weak (e.g., 160-180 dB re: 1 µPa source level) events, which

indicated that they would not be able to cover that day's energetic cost. This study suggests that context can be an important predictor of the energetic costs of sonar disturbance. Czapanskiy et al. (2021) modeled energetic costs associated with behavioral response to mid-frequency active sonar using datasets from 11 cetaceans' feeding rates, prey characteristics, avoidance behavior, and metabolic rates. They found that the short-term energetic cost was influenced more by lost foraging opportunities than increased locomotor effort during avoidance, and that mysticetes incurred more energetic costs than odontocetes, even during mild behavioral responses to sonar. Pirotta et al. (2022) integrated results from a decade of multidisciplinary research to quantify the effects of military sonar activities and potential future environmental changes on the vital rates of blue whales. Their model predicted that environmental changes would severely affect vital rates, while the current regime of sonar activities would not, despite individuals changing their behavior in response to sonar exposure (Southall et al. 2019b). Migrating humpback whale mother-calf pairs' responses to seismic surveys were modeled by Dunlop et al. (2021). The results demonstrated that migrating whales would not likely experience enough of a delay as a result of disturbance to result in population consequences, but whales disturbed in breeding or resting areas would be more vulnerable to consequences of disturbance.

Keen et al. (2021) summarized information from marine mammal PCoD models and highlighted essential intrinsic and extrinsic factors to consider when assessing likely consequences to individuals and marine mammal populations from anthropogenic stressors, including acoustic impacts (Figure 73). Here, we consider how these factors relate to the marine mammal species affected by the proposed action in terms of potential long-term consequences from acoustic and explosive impacts.

Marine mammal movement patterns broadly fall into three categories: resident, nomadic, and migratory. Species that exhibit migratory movement patterns transit annually or seasonally between sites within their range; species that exhibit resident movement patterns occupy small home ranges relative to the population's overall range; and nomadic species move over much of the population's range without spatial or temporal consistency. Whereas most large whales (including those considered in this opinion) exhibit migratory movement patterns, there is a resident population of fin whales off the California coast. The MHI IFKW is a non-migratory marine mammal that is considered a resident species of the Main Hawaiian Islands. Guadalupe fur seals spend a portion of the year living pelagically, and have been known to travel great distances, with sightings occurring thousands of kilometers away from the main breeding colonies (Aurioles-Gamboa et al. 1999). This species would not be susceptible to HCTT acoustic impacts when on their breeding colonies, which are all located outside of the action area. Hawaiian monk seals are generally considered non-migratory, typically remaining near the atoll or island where they were born, but some individuals do travel long distances.

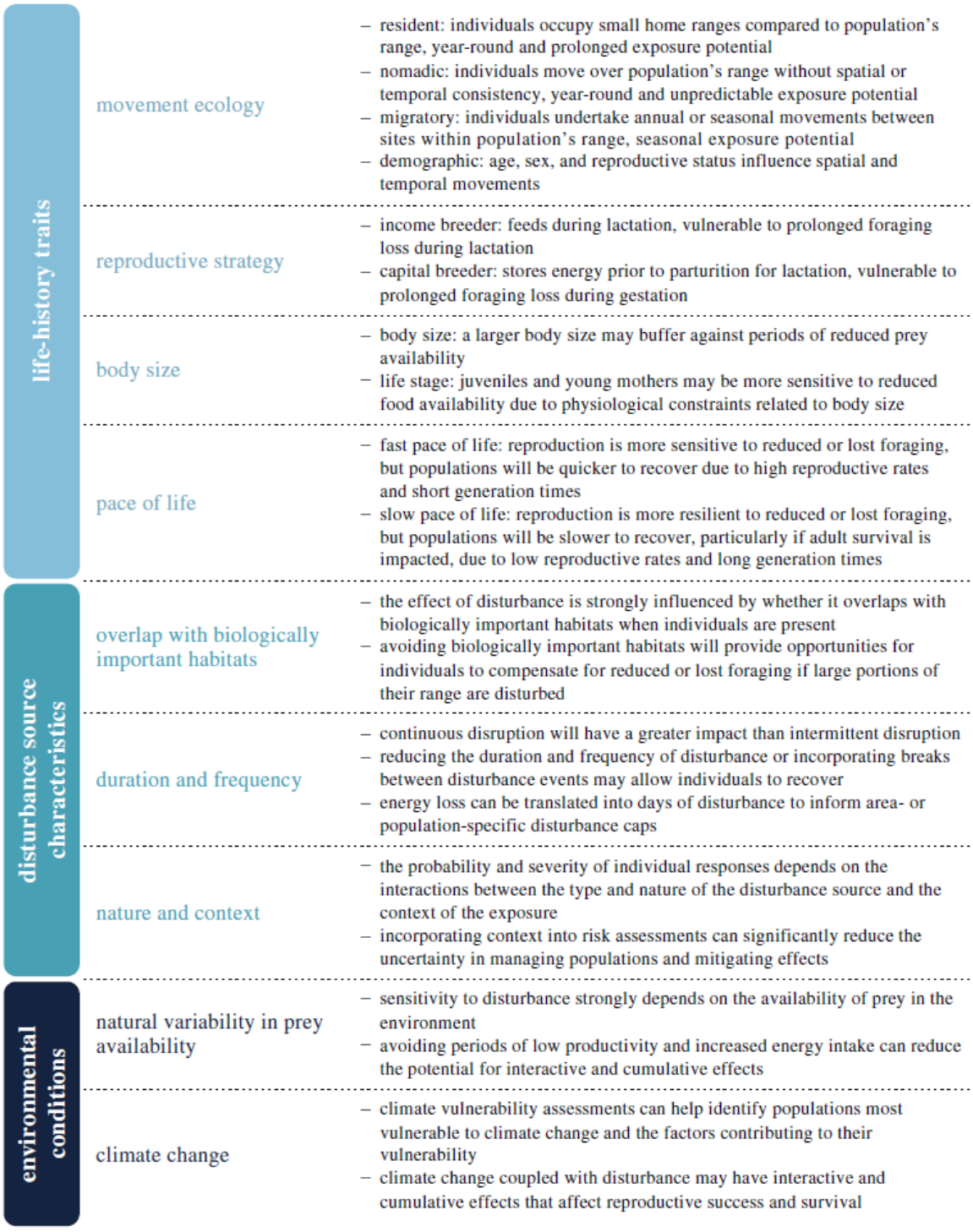


Figure 73. Intrinsic and Extrinsic Factors to Consider when Assessing Likely Consequences to Individuals and Marine Mammal Populations from Anthropogenic Stressors (Keen et al. 2021).

The exposure potential for resident or nomadic populations is year-round and the risk of repeated impacts on individuals is likely similar within the population as animals move throughout their

range (Keen et al. 2021). Because foraging grounds and reproductive areas spatially overlap for nomadic and resident populations, the behaviors potentially disrupted will depend on the timing of the disturbance event (Gallagher et al. 2021).

The other large whale species affected by the proposed action are migratory: they winter at low latitudes, where they calve and nurse, and summer at high latitudes, where they feed. In general, the exposure potential for migratory species is seasonal as this movement pattern reduces the potential for year-round, prolonged exposure from a particular disturbance-inducing activity in a given area. While the proportion of the population exposed will be high during synchronous migrations, individual exposure levels will be relatively low because individuals generally transit quickly.

Other life history traits considered in bioenergetic models include reproductive strategy, body size, and pace of life (Figure 73). All of the ESA-listed mysticetes affected by the proposed action are considered *capital breeders* because they store energy prior to parturition and lactation. Capital breeders are less sensitive to short-term foraging losses, particularly during the lactation period, because they rely on energy that has already been stored (Pirotta et al. 2019). However, capital breeders may not be able to accumulate sufficient energy for successful reproduction if disturbance within important foraging areas is prolonged, or if the available foraging habitat is spatially limited (Costa et al. 2016). In addition to a capital breeder reproductive strategy, the large body size and slow pace of life of the mysticetes considered in this opinion may be valuable in buffering against periods of reduced food availability. Sperm whales, compared to other large whales, may be less resilient to short-term foraging losses due to their income-breeding strategy (Keen et al. 2021), coupled with the physiological properties of their blubber (i.e., the vast majority of their blubber lipids are stored as wax esters, a less accessible source of metabolizable energy, rather than as triacylglycerols) (Koopman 2007).

Female true seals (family Phocidae; including Hawaiian monk seals) tend to be relatively large, with mothers fasting throughout the weaning period and relying entirely on stored capital to provision their pups (Houston et al. 2007). Thus, female phocids are often cited as classic examples of capital breeding. By contrast, smaller Otariid (including Guadalupe fur seals) females are considered income breeders, employing a foraging cycles strategy in which mothers intersperse bouts of lactation with long foraging trips, thus provisioning pups from current income (Houston et al. 2007).

The pysiological constraints of body size may also affect how individuals in early life stages respond to disturbance. Juveniles and younger (i.e. smaller) mature females are considered the most sensitive groups within a population. Our exposure analysis did not quantify HCTT impacts across the different life stages that could potentially be exposed. However, considering the species' life histories and the degree of overlap between the proposed action and species' ranges, the species with early life stages that could potentially be most at risk are Hawaiian monk seals and MHI IFKWs.

An individual's propensity to respond and the severity of the response likely depend on additional intrinsic factors that can influence the effect of disturbance on individual health and vital rates. These may include species, sex, age class, body condition, and contextual factors such as current behavioral or energetic state. When considering the potential consequences of exposure and response to military sonar, 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 long-term or repeated disturbance than an animal in poor condition. Marine mammal behavioral research has indicated that many species including humpback whales (Silve 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).

For most of the marine mammal species that will be impacted by HCTT sonar and explosive activities in the action area, we do not have information to suggest affected animals are likely to be in a compromised state at the time of exposure. During the Guadalupe fur seal UME, many individuals from this population were likely already in a compromised state of health at the time of acoustic exposure, as detailed necropsies and epidemiological analysis found the three main causes of death were emaciation (44%), trauma (29%), and infectious disease (19%) (D'Agnese et al. 2020). The UME was attributed to malnutrition likely caused by suboptimal prey conditions driven by unusually warm water oceanographic conditions in the Pacific Ocean. Although the UME was declared over in September of 2021, the Guadalupe fur seal population remains susceptible to malnutrition from future changes in oceanographic conditions. Hawaiian monk seals exposed to HCTT acoustic stressors may already be in a state of compromised health due to non-lethal interactions with fishing gears, marine debris, or diseases (NMFS 2015a).

Also important to consider is an animal's prior experience with a sound source. Given their longevity, the majority of large whales exposed to sound from HCTT training and testing activities have likely been exposed to such sources previously as these activities have been occurring in the action area for decades. Harris et al. (2017) 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. (2017b) found no large-scale avoidance by false killer whales of areas with relatively high mid-frequency active sonar use on the PMRF. The authors suggested that since sonar had been used at the PMRF 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.

An individual's sensitivity to a disturbance-inducing activity is also affected by the characteristics of the disturbance source including the spatial and temporal features and nature of the disturbance source (e.g. type, intensity, frequency, moving versus stationary), which can interact with life-history traits and other contextual factors to influence the probability and severity of individual responses (Ellison et al. 2012; Keen et al. 2021). A population's sensitivity to disturbance will be strongly influenced by the importance of the disturbed area for foraging, reproduction and migration. 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. For example, if a behavioral response occurs in foraging critical habitat, the expectation is that feeding would be interrupted in some manner; if a behavioral response occurs in calving critical habitat, the expectation is that activities associated with calf rearing (e.g., nursing young, resting) would be disrupted.

In our exposure analysis above, quantitative impacts based on NAEMO are partitioned by area type for anticipated impacts in critical habitat or BIAs. None of the migratory large whale species in the action area would be exposed to acoustic stressors in known reproductive areas. MHI IFKWs and Hawaiian monk seals could be exposed to HCTT acoustic stressors within reproductive areas used by these species. A large number of acoustic impacts along the California coast are predicted to occur in large whale foraging critical habitat (humpback whale Mexico and Central America DPSs) or foraging BIAs (blue whale, fin whale, humpback whale. A majority (59%) of the estimated impacts to Hawaiian monk seals would occur within designated critical habitat areas that are important for foraging.

The temporal features of disturbance also greatly influence the extent to which vital rates are impacted by acoustic stressors (Keen et al. 2021). Reducing the duration and frequency of disturbances can help mitigate any energetic consequences by allowing individuals to regain energy reserves. As discussed above, most HCTT activities involving sonar or explosives are short in duration (i.e., less than one day) and do not occur with any regularity in a given location. For longer duration events (e.g., MTEs), acoustic and explosives stressors would generally be dispersed over large ocean areas, thus reducing the likelihood of prolonged or persistent exposure to individual whales. Farmer et al. (2018) found that less frequent disturbance provided opportunities for sperm whales to compensate for missed foraging, with the time to terminal starvation being roughly inversely proportional to the frequency of the disturbance event. Our exposure analysis above shows the estimated average annual number of exposures (i.e., behavioral responses, TTS and AINJ combined) per individual based on the population abundance estimate and the NAEMO exposure estimates. For all marine mammals, except Guadalupe fur seals, the estimated average annual number of exposures per individual is less than two. For Guadalupe fur seals, the average annual number of exposures per individual is 3.4. Although some individual marine mammals will be exposed more often (and some will not be exposed at all) in a given year, these average annual exposure estimates provide some indication

of the frequency that most individuals would likely be exposed to acoustic and explosive stressors.

Environmental conditions can influence female body condition and impact reproductive success via changes in prey, from distribution and abundance to composition and caloric value (Keen et al. 2021). Changes in prey availability can occur across a range of spatial and temporal scales, from hour-long tidally driven hotspots and mesoscale features that can persist for months (e.g. eddies and fronts) to ocean basin-wide shifts resulting from within- (e.g. seasonal variability) and between-year (e.g. climatic oscillations) variability (Keen et al. 2021). Changing environmental conditions affect these natural oceanographic and atmospheric processes and, subsequently, the distribution and viability of marine mammal populations. Lettrich et al. (2023) conducted a trait-based assessment of vulnerability to changing environmental conditions using expert elicitation for marine mammal stocks and stock groups in the western North Atlantic, Gulf of Mexico, and Caribbean Sea. Based on this assessment, the overall vulnerability rank was considered "High" for all the large whale species considered in this opinion (fin, sei, blue, and sperm whales) that were part of this assessment (Lettrich et al. 2023). The effects of changing environmental conditions on ESA-listed species in the action area are discussed in more detail in our Environmental Baseline section above (Section 5.1)

For more detailed information on how PCoD Models are utilized, including data input needs, contextual considerations, and case studies from best available science, please refer to Section D.4.8 (*Population Consequences to Marine Mammals From Acoustic Stressors*) in Appendix D of the HCTT Final EIS/OEIS (Navy 2025d).

6.1.2 Weapons Firing Noise: In-Air Effects to Hauled Out Pinnipeds

In Section 4.1.1 we determined that the effects of HCTT weapons noise on ESA-listed marine mammals in the water are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated) and, therefore, are not likely to adversely affect these species. In this section we address the effects of weapons firing noise to ESA-listed pinnipeds when they are hauled out on land. Noises with sudden onset or high amplitude relative to the ambient noise level may elicit a behavioral response from pinnipeds resting on shore. Guadalupe fur seal are not known to haul out on land in any areas where HCTT weapons noise would occur. Therefore, this section focuses on the following acoustic stressors that may result in effects to Hawaiian monk seals: noise associated with missile and aerial target launches occurring on land-based sites at PMRF; and noise associated with artillery firing at PMRF from land-based sites (Navy 2025g). The Navy has proposed to conduct up to three artillery events (with each event lasting up to four days), and 20 missile launch events (with each event lasting up to three days) annually at PMRF.

Exposure Analysis

For the non-impulsive missile or air vehicle launch noise analysis, SPL and SEL were measured from a single Medium-Range Ballistic Missile (T3C2 missile) Target Vehicle launch event at four locations on PMRF (Kim and Norman 2023). The approach used is described in the Navy

technical memorandum *Acoustic Measurements of Pacific Missile Range Facility (PMRF) Missile Launch: March 2023* (NMFS 2023j). The unweighted SPL of the non-impulsive event at the closest of the four locations to the launch site was 129.1 dB re: 20 µPa (peak received level of 143.2 dB re: 20 µPa). The spherical spreading model was used to estimate distances to threshold levels based on the recorded acoustic data. For each doubling of distance from a point source, the sound level attenuates (or drops off) at a rate of 6 dB (Navy 2025g). It is important to note that the spherical spreading model used in the analysis in this paper does not account for attenuation due to meteorological conditions, physical barriers in the environment, and variations in the type or density of vegetation impeding sound propagation, all of which would affect how far the sound propagates from the source and the ranges to auditory effects (Navy 2025g).

To estimate distances to weighted SEL thresholds for Hawaiian monk seals, the time waveform from the site closest to the launch location was filtered using the audiometric weighting function for phocids in air, and a SEL was calculated using the estimated 4.7 second duration of the event (Kim and Norman 2023). This analysis applied auditory impact thresholds for TTS and AINJ developed for phocids in air. The TTS weighted non-impulsive SEL threshold for phocids in air is 134 dB re: (20 µPa)^{2}s, and the AINJ weighted SEL non-impulsive threshold for phocids in air is 154 dB re: (20 µPa)^{2}s. The development of the auditory criteria is described in detail in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV)* (Navy 2025c).

To estimate ranges to effects, the time waveform was scaled until the calculated SEL matched first the AINJ threshold and then the TTS threshold. The unweighted peak SPL of the waveform was then noted, and utilizing the spherical spreading model as described above, the approximate distance to the peak SPL threshold was calculated (Navy 2025g). Using this method, the distances to the weighted SEL thresholds were estimated as follows: Weighted Non-Impulsive AINJ Threshold (SEL) = 6 ft; Weighted Non-Impulsive TTS Threshold (SEL) = 620 ft (Navy 2025g).

The Navy has indicated that the T3C2 missile launch used to calculate these ranges to effects represents the maximum non-impulsive sound source that would be used at PMRF. A smaller rocket fired from the High Mobility Artillery Rocket System (HIMARS) would be used more regularly at PMRF. Based on a similar approach to that described for the T3C2 missile, the largest estimated ranges to effects from the HIMARS, using the dBA at the source (i.e., even smaller ranges were calculated using dBA at 250 m correction for spherical spreading loss), were as follows: TTS = 232 ft; AINJ 8 ft.

Artillery firing activities would take place in the same locations at PMRF where the missile and launch vehicle activities would be conducted, and noise from those events has the potential to reach Hawaiian monk seal haulout locations (Navy 2025g). No acoustic measurements of artillery firing noise have been recorded at PMRF, so an estimate of ranges to auditory effects was based on published measurements. This analysis is based on overpressure measurements reported in a study from firing a 155mm M777 Howitzer (Wiri et al. 2023). Median and

maximum peak overpressure were measured at 178.6 dB re: 20 µPa and 186.8 dB re: 20 µPa, respectively (Wiri et al. 2023). The average distance of the sensors from the firing position in this study was approximately six feet. Using the spherical spreading model and the maximum overpressure recorded, the distance to an approximate received level of 162 dB re: 20 µPa (the Phase 4 impulsive peak SPL unweighted AINJ threshold for phocids in air) would be approximately 95 ft from the source. The impulsive unweighted peak SPL threshold for TTS is 156 dB, which would be approximately 190 ft from the source (Navy 2025g).

Of the three sound sources analyzed (i.e., T3C2 ballistic missile, HIMARS rocket, and M777 Howitzer), the largest estimated range to TTS or AINJ effects was 620 ft. As shown in Figure 74, Hawaiian monk seals hauled out on beaches near PMRF would not be located within this estimated range to effects. Therefore, based on the analysis above, Hawaiian monk seals would not be exposed to weapons firing noise from PMRF at levels that would exceed either the TTS or AINJ thresholds (Navy 2025g).



Figure 74. Distances to Non-Impulsive Phocid Thresholds from Launch of T3C2 Missile at PMRF (Navy 2025g). Green circle represents a 620 ft. range to effect.

The in-air onset of behavioral disturbance acoustic threshold of 100 dB re: 20 µPa (Southall et al. 2007) was used to estimate Hawaiian monk seal significant behavioral responses to weapons firing noise at PMRF. The range to behavioral effects using the general in-air 100 dB re: 20 µPa rms SPL threshold would likely cover most of the beach of PMRF Barking Sands and adjacent nearshore waters for worst case launches (Navy 2025j).

The Navy has indicated that there are 23 planned launch events (3 artillery; 20 missile launches) at PMRF on an annual basis (Navy 2025l). Artillery events can last up to four days each and missile launch events can last up to three days each. Multiplying the maximum number of planned events by the maximum number of days per event (i.e., [3*4] + [20*3]) yields 72 as the maximum number of days per year that Hawaiian monk seals could be exposed to in-air acoustic effects when hauled out on beaches near PMRF.

The next step is to determine the average number of monk seals hauled out per day at beaches within range to behavioral effects from PMRF sound. From 2020 to 2023, an annual average of 215 monk seals were counted hauled out on beaches near PMRF (Navy 2025j). This annual estimate is based on the count data derived from survey protocols that include both daily and bi-weekly counts conducted by both Natural Resources and Security personnel at PMRF (Navy 2025j). However, this annual average likely represents a minimum estimate, since surveys are somewhat opportunistic and are not conducted every day or at every haul-out location within the range to acoustic behavioral effects. Rather than rely on the minimum annual average (i.e., 215) based on incomplete survey data to estimate the average number of monk seals exposed per event day, we instead use available information on the maximum number of monk seals observed during a single day at the surveyed sites. Based on survey data from 2020 through 2023, the maximum number of seals within the range to behavioral effects observed during a single day was five (Navy 2025j). Since the maximum number of monk seals will not likely be hauled out every day of artillery and missile launch events, this likely represents a conservative estimate of the average number of monk seals exposed per day. However, due to gaps in the monk seal survey effort, including haul-out sites not routinely surveyed and/or days not surveyed, we consider the maximum number of seals observed during a single day (i.e., five) to represent the best available information for conservatively estimating the average number of monk seals that will be exposed to behavioral disturbance effects per day during PMRF artillery and missile launch events.

Therefore, we estimate that the maximum number of Hawaiian monk seal exposures annually to PMRF weapons firing noise at levels resulting in a significant behavioral response is 360 (i.e., 5 monk seals per day * 72 days of launch events per year). This exposure estimate assumes that, on average, five Hawaiian monk seals would be within the range to behavioral effects during each launch day, and all exposed animals would have a significant behavioral response.

Response Analysis

To document the responses of pinnipeds to Navy launches of missiles and similar aerial vehicles, Holst et al. (2011) observed three species of pinnipeds during 77 launches from San Nicolas

Island off California from 2001 to 2008. The majority of observed California sea lions startled and showed increased vigilance up to two minutes after each launch. Responses often included movement on the beach or into the water and were significantly related to received sound level and distance from the vehicle's closest point of approach. Most observed northern elephant seals showed little reaction to launches and merely raised their heads briefly. Harbor seals were the most responsive species, with about two-thirds of animals within approximately 4 km of the launch trajectory leaving their haul-out site by entering the water. Hawaiian monk seal in-air hearing is less sensitive than hearing in other phocid seals (Ruscher et al. 2021; Ruscher et al. 2025), suggesting that Hawaiian monk seals, and other species in the Monachinae subfamily, may be even less likely to respond to in-air noise as compared to harbor seals or other highly sensitive Phocinae seals.

6.1.3 Vessel Strike Analysis

In this section, we analyze the effects of vessel strike to marine mammals as a result of HCTT activities. Our focus of this analysis is on large whales because, based on historical records of military vessel strikes in the action area, these species are the most susceptible marine mammals in terms of HCTT vessel strike risk. In our 2024 HSTT reinitiated biological opinion (NMFS 2024f), we determined that the risk of a Navy vessel strike of a Guadalupe fur seal, Hawaiian monk seal, or MHI IFKW was so low as to be considered discountable (i.e., extremely unlikely to occur). We found no new information to indicate that this determination should be revised. While these species could potentially be at risk of a vessel strike, there have not been any documented cases of a Navy or Coast Guard vessel striking a pinniped (e.g., Guadalupe fur seal or Hawaiian monk seal) and only one instance of a Navy vessel striking a small odontocete (a non-listed dolphin), and this occurred outside of the HCTT action area[27]. Other than this one reported strike of a dolphin, the Permits Division has never received any reports from other LOA or Incidental Harassment Authorization holders indicating that these smaller marine mammal species have been struck by military vessels. Their much smaller size and greater maneuverability make these species less susceptible to a vessel strike in general, as compared to large whales. This is supported by worldwide vessel strike records that show strikes of smaller, more agile marine mammals are extremely rare. Due to these species' greater maneuverability, the relative lack of known susceptibility to vessel strike, the lack of documented incidences where these species have been struck, and the Action Proponents' proposed mitigation to avoid striking marine mammals, the risk of a vessel strike of a Hawaiian monk seal, Guadalupe fur seal, or MHI IFKW as a result of the action is extremely low. As such, we find that stressors resulting from HCTT vessel strike may affect but are not likely to adversely affect the Hawaiian monk seal, Guadalupe fur seal, or MHI IFKW as these effects are discountable.

[27] A dolphin was accidentally struck by a Navy small boat in the fall of 2021 in Saint Andrew's Pass, Florida.

Exposure Analysis

We start our vessel strike exposure analysis by reviewing the recent history of military vessel strikes involving large whales in the action area, the particular circumstances surrounding those strikes, and how information on past vessel strikes can inform our analysis for this consultation. Next, we consider the risk factors that affect the probability of a large whale vessel strike, and how they relate to the proposed action. We then describe the proposed mitigation measures for minimizing the risk to large whales from vessel strike, and discuss the anticipated effectiveness of those measures. We base our analytical vessel strike exposure analysis on the quantitative approach developed for the reinitiated Navy HSTT Phase III biological opinion (NMFS 2024f), updating this analysis based on new vessel activity and historical strike information, and including Coast Guard vessel activities and strike history in the HCTT action area. Our quantitative analysis considers vessel strike of large whales comprehensively, as a result of all Action Proponent vessel movement throughout the action area, as opposed to in the context of specific training or testing exercises. Finally, we consider application of our quantitative analysis to our overall exposure analysis in light of the recent history of military vessel strikes, context-specific risk factors, and proposed mitigation measures.

Recent History of Military Vessel Strikes of Large Whales in the HCTT Action Area

In the Hawaii portion of the action area, the Navy struck a total of five large whales in the 18-year period from 1991 through 2008. All five Navy vessel strikes over this period in Hawaii were reported as injuries. The vessel struck species include: one humpback whale in 1998, one unknown species and one humpback whale in 2003, one sperm whale in 2007, and one unknown species in 2008. There have not been any documented vessel strikes associated with training and testing in the Hawaii portion of action area since 2008.

In the SOCAL RC portion of the action area, the Navy struck a total of 16 large whales in the 20-year period from 1991 through 2010 (i.e., an average of about one per year). Of the 16 Navy vessel strikes over the 20-year period in SOCAL RC, there were seven mortalities and nine injuries reported. The vessel struck species include: two mortalities and eight injuries of unknown species, three mortalities of gray whales (one in 1993 and two in 1998), one mortality of a blue whale in 2004, and one morality and one injury of fin whales in 2009. The Navy vessels involved in the two separate fin whale strikes in 2009 were a destroyer and a cruiser. The locations of the two strikes in 2009 were both south and east of San Clemente Island (Figure 75). From 2010 through 2020, there were no documented vessel strikes associated with Navy training and testing activities in the SOCAL RC portion of action area.

On May 7, 2021, the Royal Australian Navy HMAS Sydney, a 147.5 meter Hobart Class Destroyer, struck and killed two fin whales (a mother and her calf) while operating within the SOCAL RC portion of the action area. The carcasses were first sighted under the bow of the vessel while it was approaching the Naval Base in San Diego. The whales had been pinned to a sonar dome in the front of the vessel due to the force of water as the ship was underway. The

morning of the strike the HMAS Sydney was getting into position to participate in a U.S. Navy-led exercise later that day. However, because the vessel was not actively participating in a U.S. Navy-led exercise at the presumed time of the strike, the actions of the HMAS Sydney resulting in this strike are not considered a consequence of the Navy's action (NMFS 2022b). Although the HMAS Sydney vessel strike of two fin whales is not considered a consequence and, therefore, not an "effect," of the action, we still consider it relevant new information as an indicator of the potential risk of large whale vessel strike from the action because it involved a foreign navy vessel that was transiting to participate in a U.S. Navy-led exercise later that day.

In addition to the Australian Navy strike, there have been five reported U.S. Navy vessel strikes to unknown species of large whales within the HSTT Southern California action area since the 2018 HSTT biological opinion was issued. These five vessel strikes fall within the scope of the Navy's action for the 2018 HSTT Phase III consultation. On June 29, 2021, a Navy cruiser struck an unknown whale species approximately 95 nm west of San Diego (Figure 75). On July 11, 2021, a Navy cruiser struck an unknown whale species approximately 90 nm south-south west of San Diego (Figure 75). The ship had been participating in a Major Training Exercise (Large Integrated Anti-Submarine Warfare - Composite Unit Training Exercise) within the SOCAL RC portion of the action area (Navy 2021). On May 20, 2023, a Navy aircraft carrier presumably struck a whale approximately 70 nm west of San Diego. Navy personnel captured video and photos of the carcass, and based on those images, NMFS and the Navy determined this whale was either a fin whale or sei whale; the two species are very similar morphologically and are difficult to distinguish from one another at sea. On July 15, 2025, a Navy destroyer struck an unknown species of large whale approximately 57 nmi (105.6 km) west/southwest of San Clemente Island (Figure 75). The whale was initially lodged to the bow and the ship took immediate action to dislodge the whale. Navy personnel estimated that the whale was approximately 20 to 25 ft long, but low light prevented observation of other identifying features and the carcass quickly sank after dislodging. On August 10, 2025, an amphibious transport dock ship struck a large whale approximately 13 nmi (24 km) west/southwest of San Clemente Island (Figure 75). Vessel crew noted that the struck whale was alive and continued to transit in a northeast direction following the strike, surfacing approximately every eight minutes. Although the species of whale was not identified at the time of the strike, four days later (on August 14) Navy biologists found a dead blue whale washed up on San Clemente. While the stranding location and timing are close to the location and timing of the Navy strike, with a significant large whale population off Southern California, based on the available information at this time, we cannot say conclusively whether the blue whale carcass found on August 14 was the same whale struck by a Navy vessel on August 10.

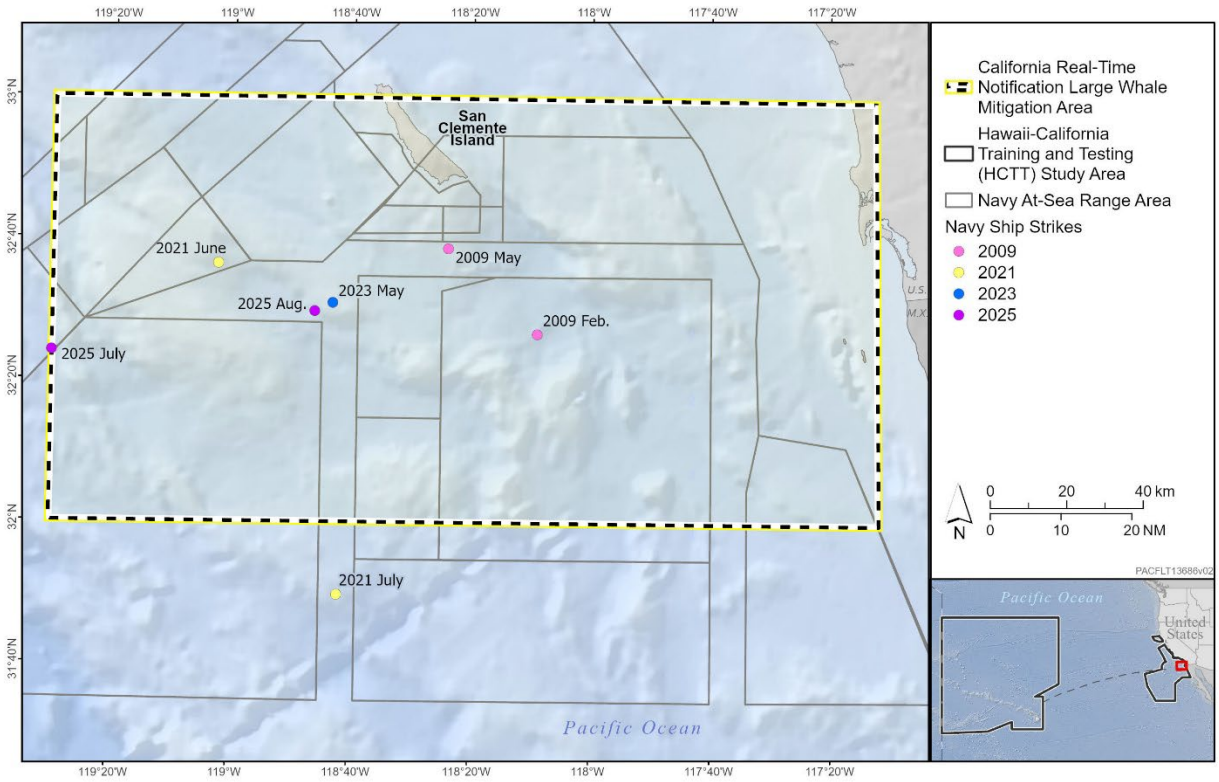


Figure 75. Locations of U.S. Navy Vessel Strikes of Large Whales from 2009 through August 2025 within the SOCAL RC Portion of the Action Area (Navy 2023d)

The Coast Guard reported striking just one large whale within the HCTT action area since 2009. This strike of a non-ESA-listed humpback whale by a 45-foot Response Boat-Medium vessel occurred on February 14, 2020 in Papawai Point, Maui, Hawaii. Based on the report details from this incident this was likely a non-lethal vessel strike. In addition to this one strike within the action area, the Coast Guard reported striking two marine mammals just outside of the action area since 2009. On June 6, 2016, an 87-foot Coast Guard Cutter reported striking a humpback whale near the Golden Gate Bridge in San Francisco Bay, California, just east of the NOCAL RC. The whale was seen swimming away with no apparent injuries. On March 15, 2024, an 87-foot Coast Guard Cutter reported striking an unknown marine mammal in Half Moon Bay, California, in between the two portions of the NOCAL RC (Figure 9). Although the 2016 and 2024 Coast Guard strikes occurred outside of the HCTT action area, and are thus not included in our quantitative analysis below, given their proximity to the NOCAL RC we still consider them as relevant qualitative information for evaluating the potential risk of large whale vessel strike from the proposed action.

For purposes of our analysis, we assume that the large whale vessel strikes described above represent a complete record of recent Navy and Coast Guard vessel strikes within the action area. Whereas reported or documented whale strike incidents involving commercial vessels likely represent a small fraction of actual commercial vessel strikes since they are not required to report

them (NMFS 2011; Carretta et al. 2021), the Action Proponents remain confident that their ability to detect whale strikes as they happen is effective, and reporting to NMFS is an accurate record of past Navy and Coast Guard strikes. This is based on the higher manning requirements on military ships (bridge watch team including dedicated lookouts, deck personnel, etc.), standard operation procedures, mitigation measures, reporting requirements, and marine species awareness training (Navy 2021).

In some instances, a struck whale can become attached or impinged to a feature on the vessel's hull and may not be detected when the strike occurs. While this is more common with large commercial vessels, for the Australian Navy military vessel strike described above, the two whales were lodged on the sonar dome in the bow of the vessel when it came into port. The Navy has indicated that it would be implausible for a marine mammal to become lodged on the sonar dome of a U.S. Navy ship and remain undetected because of a technological standard operating procedure (Navy 2021). Sonar domes on U.S. Navy ships have a pressurized rubber window that maintains 150 PSI through the ships fire main. If anything affects the pressure, an alarm sounds alerting personnel that something had hit the sonar dome. The shape, hydrodynamic design, construction using a non-abrasive material, and regular hull cleaning procedures to remove barnacles and other growth on U.S. Navy ships are also reasons that it is highly unlikely a whale could become lodged on a U.S. Navy ship's sonar dome (or other part of the bow) and remain undetected (Navy 2021).

Large Whale Vessel Strike Risk Factors

The relative risk of a large whale vessel strike within a particular area is primarily a function of animal density and the magnitude of vessel traffic (e.g., Fonnesbeck et al. 2008; Vanderlaan et al. 2008). The relative density of large whales in areas with concentrated vessel activity can vary significantly by species, time of year, and over time with changing environmental conditions (e.g., temperature, prey availability). Blondin et al. (2020) combined predicted daily whale distributions with continuous vessel movement data to estimate blue whale ship strike risk in the Southern California Bight. Their results suggest that vessel strike risk intensity is typically greater and extends over a longer portion of the annual cycle during an El Niño year than during a "normal" climatic year. Szesciorka et al. (2020) found that average blue whale residency time at the feeding grounds off Southern California increased by about a month from 2008-2017 (Szesciorka et al. 2020). They concluded that blue whale arrival at their feeding grounds and start of reproductive calling behavior appears to be driven by an interaction with temperature and prey (Szesciorka et al. 2020). With projected increases in the frequency of extreme El Niño events (Cai et al. 2014; Wang et al. 2017) and in vessel traffic to 2050 (Sardain et al. 2019), blue whale vessel strike risk in the Southern California Bight could continue to increase into the near future.

Other factors, such as vessel speed, size, and maneuverability can also influence both the probability of a vessel strike occurring and the outcome (i.e., minor injury, serious injury,

mortality) when a strike occurs. In this section, we focus on the factors affecting vessel strike risk as they relate to the proposed action. We will consider the factors that could affect the outcome of a vessel strike that could result from the action in our response analysis below. For a more general discussion of vessel strike risk from past and present activities in the HCTT action area, including military and non-military vessels, we refer the reader to Section 5.14 of the Environmental Baseline.

Most military readiness activities involve vessel movement. Vessel strikes to large whales are not associated with any specific training or testing activity but rather a limited, sporadic, and accidental result of vessel movement in the action area. Vessel movement can be widely dispersed throughout the action area, occurring in both offshore and inshore water areas. HCTT vessel traffic is concentrated around the main Hawaiian Islands, particularly Pearl Harbor and off the coast of Southern California. During RIMPAC years, foreign military vessels can add a significant amount of vessel activity with the HCTT action area (see Section 2.3 for details). For more information on HCTT vessel activity see Section 2.5.1.

While large whales overlap spatially with HCTT vessel activity throughout the offshore portion of the HRC, we would not expect concentrations of large whales in the nearshore areas of high intensity vessel traffic around the main Hawaiian Islands. While Action Proponent vessels have struck large whales in the HRC in the past, as noted above, only one military vessel strike (i.e., Coast Guard strike of a non-ESA-listed humpback whale in 2020), has been recorded in this portion of the action area since 2008. The California coast, including portions of the action area, represents an area of significant overlap between important feeding grounds and migratory routes for large whale species and heavy vessel activity, primarily commercial vessels operating in shipping lanes to and from major ports. The spatial overlap between high whale densities and heavy vessel traffic has resulted in a relatively large number of vessel strikes in this area. While the large majority of whale vessel strikes recorded off of California are from commercial vessels, military vessel strikes have been reported within the SOCAL RC portion of the HCTT action area, with the most recent strikes occurring in 2021 (four whales) and 2023 (one whale).

Vessel size is also an important predictor of large whale vessel strike risk. Vessel strikes of large whales by military vessels typically involve very large vessels such as cruisers, destroyers, and aircraft carriers. In general, smaller vessels have greater maneuverability, which allows them to change course more quickly to avoid a collision once a whale has been seen. Lookouts may also be more effective at spotting whales within the full 360 degree radius mitigation zone on smaller vessels as compared to larger vessels (although the vessel configuration and location and number of Lookouts would also be factors). However, as noted above, Coast Guard strikes involving vessels ranging from 45-87 ft have been reported in recent years, both within and outside of the action area. The use of small crafts associated with HCTT activities within inshore waters would occur on a more regular basis than offshore large vessel use, and typically involve high speed (greater than 10 knots) vessel movements. However, inshore waters are generally more confined waterways where ESA-listed large whale species considered in this opinion either do not

typically occur or occur in very low densities. Vessel size also plays a role in the anticipated outcome when a large whale vessel strike occurs. Kelley et al. (2021) found that while vessels of all sizes can cause lethal injuries to whales, large vessels (> 20 m) produce much larger stresses even when travelling at reduced speeds (i.e., 10 knots).

The operation and configuration of large military vessels (greater than 18 m [59 ft] in length) within the action area differ from commercial merchant vessels in ways that may reduce the risk of striking a large whale. Many Action Proponent ships have their bridges positioned closer to the bow, offering better visibility ahead of the ship (compared to a commercial merchant vessel). The crew sizes on large military vessels are generally larger than merchant ships, allowing for stationing more trained lookouts on the bridge and in other locations. There are often aircraft associated with HCTT activities, which can serve as additional lookouts for detecting cetaceans in the vicinity of a vessel or ahead of a vessel's present course before crew on the vessel would be able to detect them. Military vessels would also likely be more maneuverable in avoiding large whales spotted in the path of the vessel than commercial merchant vessels due to their quicker rudder responses, resulting in a turning radius that is typically smaller compared to commercial ships.

In addition to the volume of vessel traffic and vessel size, a correlation between vessel speed and the risk of striking a whale has been shown in several studies. 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. Jensen and Silber (2004) analyzed 292 records of known or probable ship strikes (inclusive of military and non-military vessels) of all large whale species from 1975-2002. Of the cases where vessel speed at the time of collision was reported (n=58), about two-thirds resulted in serious injury (n=19) or death (n=20). While reported operating speeds at the time of strike ranged from 2-51 knots, the majority (79%) of vessels that struck various species of large whales reported speeds of 13 knots or greater. Rockwood et al. (2021) developed a ship strike model using whale density data and automatic identification system (AIS) vessel data to estimate mortality under different management scenarios in the Santa Barbara Channel, California. Their results showed that if 95% cooperation with the voluntary vessel speed reduction recommendation occurred in the shipping lanes, whale deaths there would decrease by 22–26%. Similarly, results from a study by Redfern et al. (2019) suggest that a large decrease in ship-strike risk can be achieved by speed reductions throughout the Southern California Bight. Rockwood et al. (2020) studied the effectiveness of speed reduction measures for decreasing whale-strike mortality in shipping lanes off San Francisco, California. They predicted that if 95% of mariners adhered to the recommended 10 knot limit in the shipping lanes, twice as many blue whale and 3 times as many humpback whale deaths would be avoided relative to current adherence to this voluntary limit.

Crum et al. (2019) modeled vessel strike risk to North Atlantic right whales before and after implementation of time-area vessel speed restrictions. They found that the expected seasonal mortality rates of right whales decreased by 22% on average after the speed rule was

implemented, indicating that the rule is effective at reducing lethal collisions. By comparison, Conn and Silber (2013) estimated that vessel speed restrictions reduced North Atlantic right whale ship strike mortality risk levels by 80–90%. The primary difference between the two models is that Conn and Silber (2013) measured the effect of vessel speed on a process that combined the encounter rate and avoidance rate, whereas the Crum et al. (2019) model considered these as separate processes. Thus, we would expect the reduction in vessel strike mortality rates due to vessel speed restrictions to be greater if whales exhibiting avoidance behavior are more susceptible to collisions (i.e., reduced ability to avoid a nearby vessel) with faster moving vessels. In some scenarios, slower vessels may allow whales increased opportunities for avoidance due to a combination of maneuverability and detection (Gende et al. 2019). A faster vessel will travel a greater distance compared to a slower moving vessel over a set period of time, such as when whales are submerged between surfacing events, on deep dives, or during time lags related to decision-making and communications on the bridge following detection (Gende et al. 2019). These factors can all result in an increased susceptibility of large whales to vessel strike with increasing vessel speed.

Vessel speed can also influence whale avoidance by influencing detection probability. Case studies of large whale behavioral avoidance suggest that blue whales may respond to close encounters with large vessels by altering their diving behavior rather than moving horizontally at the surface (McKenna et al. 2015; Szesciorka et al. 2019). In one case study of a close vessel encounter with a blue whale, Szesciorka et al. (2019) noted that the ship's reduced speed (i.e., 11.3 knots) may have played a role by giving the whale enough time to respond to the nearby vessel. A behavioral dive response may not be as effective at avoiding vessels traveling at higher speeds (McKenna et al. 2015). Higher vessel speeds increase the risk that a whale could have been struck at the surface or get close enough to the ship's draft that the propeller suction effect created by the ship's hydrodynamic flow could pull the whale toward the hull (Szesciorka et al. 2019). In general, vessels transiting at high speeds will produce more noise than vessels transiting at low speeds, thus increasing the whale's ability to avoid an encounter. However, many military vessel are strategically designed to be quieter than similar sized commercial vessels, which may reduce the available reaction time for large whales to avoid a vessel interaction.

Military vessels generally transit at speeds that are optimal for fuel conservation or to meet operational requirements (Navy 2024d). Large Navy surface vessels (greater than 59 ft in length) generally operate at average speeds of between 10 and 15 knots. While these speeds are considered averages and representative of most events, some vessels need to operate outside of these parameters during certain situations. Large vessels may occasionally be required to operate at speeds that are higher than normal operating speeds, which may pose a greater strike risk to large whales because there would be less time for the vessel crew to detect a large whale and maneuver to avoid a strike, and there would be less time over a given distance for the animal to react and avoid the vessel. For example, to produce the required relative wind speed over the

flight deck for take-off and landings, an aircraft carrier vessel group engaged in flight operations must adjust its speed through the water accordingly.

Within the HRC core use areas, average Navy vessel speeds were more variable by area, and generally higher than average non-military vessel speeds, particularly in offshore areas. Navy vessel speeds averaged 10-15 knots or less within most areas in and around Pearl Harbor and the main Hawaiian Islands, but increased to 15-20 knots in more offshore areas. By comparison, non-military vessel speeds averaged 5-10 knots in most areas around the main Hawaiian Islands, increasing to 10-15 knots in more offshore areas (Starcovic and Mintz 2021). Within the SOCAL RC core use areas, the average speed of Navy vessels is lower than the average speed of commercial vessels. Typical average speeds of Navy vessels from 2014-2018 were consistently between 10 and 15 knots or lower throughout most of the SOCAL RC (Starcovic and Mintz 2021). By comparison, typical average speeds of non-military vessels from 2014-2018 were between 15 and 20 knots in many portions of the SOCAL RC, particularly areas farther from shore (Starcovic and Mintz 2021). While average Navy vessels speeds are typically lower than those of non-military vessels, they are still above the vessel strike mitigation speed of 10 knots that has been implemented (or recommended as voluntary) for commercial vessels in other parts of the country. Although, at present, there is no voluntary commercial vessel speed recommendation for commercial vessels operating within the HCTT action area. While average Navy vessel speeds in HSTT core high use areas are between 10-15 knots, there are times when operational, scheduling, logistics, training, or testing requirements mandate vessels travel at speeds in excess of 15 knots (Navy 2021). For example, for the 2 Navy vessel strikes in SOCAL RC in 2021, one vessel was traveling at 25 knots at the estimated time of impact, while the other vessel had reduced its speed from between 25-30 knots to 17 knots, which was the speed at the estimated time of impact. For the 2023 Navy vessel strike in SOCAL RC the ship speed at the time of impact remains unknown. Although the ship speed when the carcass was first sighted attached to the ship's bow was recorded as 5 knots, the ship was traveling at higher speeds, ranging from 5-17 knots, in the hour before the carcass was spotted.

There are a few specific offshore events, including high-speed tests of newly constructed vessels, where vessels would operate at higher speeds. Propulsion testing, which sometimes includes ships operating at speeds in excess of 30 knots, and use of large high-speed unmanned surface vessels may pose a higher strike risk because of the high speeds at which some vessels need to operate to complete the testing activity. Smaller military vessels (i.e., less than 50 ft in length), which are all support craft, typically have much more variable speeds (0 to 50+ knots, dependent on the mission) compared to large vessels. High speed movements of smaller vessels during inshore operations could occur more frequently. However, given the location (inshore or nearshore) and maneuverability of smaller vessels, the risk of a large whale vessel strike in such scenarios would be much lower than the strike risk from a large vessel operating in offshore portions of the action area at high speeds.

A whales' behavioral state has been shown to influence the context-dependent behavioral response to acoustic stressors. Feeding whales may be distracted and thus be less capable of detecting and avoiding approaching vessels (Szesciorka et al. 2019). Mother-calf pairs have greater energetic requirements than other age and social classes and, therefore, are likely affected by approaching vessels in ways that are not immediately apparent (Smultea et al. 2022). Smultea et al. (2022) observed a blue whale mother and calf's behavioral response to an approaching small recreational vessel in the Southern California Bight. The calf first approached the vessel while it was stationary, and then quickly swam away when the vessel started to move. Maximum spacing between the mother and calf increased from about 25-50 m when the vessel was present. The observed behavioral avoidance of the vessel by the calf once the vessel started moving, which was associated with an increase in speed at the water surface, matches other reports of blue whales exhibiting fleeing responses to vessel disturbance, which are similar to flight responses by blue whales during observed predator attacks (Ford and Reeves 2008).

Vessel Strike Mitigation Measures

Proposed mitigation measures to minimize the risk of vessel strike for HCTT Phase IV that were carried over from HSTT Phase III include: 1) required training for watchstanders and Lookouts to improve the effectiveness of visual observations for marine mammals and sea turtles; 2) issuance of area-specific seasonal awareness messages to alert Action Proponent vessels and aircraft operating in the area to the possible presence of concentrations of large whales; 3) activity-based mitigation involving the use of Lookouts to maintain minimum distances to marine mammals when a vessel is underway; 4) designated marine mammal mitigation areas that limit the use of HCTT sonar and/or explosives, either seasonally or year-round, in areas with high densities of marine mammals; and 5) a large whale aggregation real-time reporting requirement in portion of the SOCAL RC. These mitigation measures are all described in detail in in Section 2.4.2. Given the lack of data on their effectiveness, mitigation measures proposed by the Action Proponents to reduce the risk of vessel strike are not factored into our quantitative exposure analysis (see Vessel Strike Probability Calculation section below), but are considered qualitatively in our overall vessel strike effects analysis.

Action Proponent Lookouts and bridge watch teams are required to maintain a 500-yard mitigation zone around whales, to avoid approaching any whale head-on, and to maneuver the vessel to maintain distance (when safe to do so) if a whale is observed within the zone. The Navy updated their navigation instructions in 2018 to require three Lookouts on Navy cruisers and destroyers (Navy 2021). This change occurred for increased safety navigation purposes in general, and was implemented after the 2018 HSTT biological opinion was issued. The effectiveness of the vessel movement mitigation measure in terms of reducing vessel strike risk is largely based on the Lookouts' ability to effectively monitor the mitigation zone. As discussed above (Section 6.1.1), the results of Oedekoven and Thomas (2022) suggest that Navy Lookout Teams (including Lookouts and other crew members) fail to detect large whales within the designated mitigation zone a large proportion of the time (i.e., 85% undetected within 500 yards;

80% undetected within 200 yards). While the Navy's activity-based mitigation for vessel movement may provide some benefit in terms of reducing vessel strike risk when whales (or signs of whales) are detected, the anticipated high rate of undetected whales in close proximity to the vessel suggests that the overall effectiveness of this mitigation may be limited, particularly in certain situations.

In addition, maneuvering to avoid observed whales at the surface may inadvertently put the vessel on a collision course with undetected whales. Whales that spend a lot of time below the surface are particularly vulnerable to vessel strike because opportunities for Lookouts to detect them are more limited. This risk is compounded by high vessel speeds, which can affect both the ability of Lookouts to detect a whale, and the whale's ability to avoid being struck by the vessel. As described above, for the seven military vessel strikes (three U.S. Navy and two Royal Australian Navy) from 2021 through 2025, prior to being struck, the whale(s) was either not seen until the last minute or not seen at all by vessel crew. In a study quantifying behavior and collision risk of humpback whales surfacing near large ships in Glacier Bay Alaska, Helm et al. (2023) found that around half the time that whales ascended from depth and surfaced to breathe they went undetected in the 20-second period prior to the encounter period in which they were detected. Further, in a given 20-second period following initial detection, a whale was only visible to an observer stationed at the bow about one-third of the time, on average, that it was present at the surface (Helm et al. 2023). This study highlights the shortcomings of vessel strike mitigation that relies solely on maneuvering to avoid observed whales, because whales at the surface within the strike zone often go undetected.

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. However, the use of passive acoustic devices is not a required mitigation measure, and the majority of time Navy vessels are underway they are not actively using these devices. In addition, passive acoustic monitoring is only effective at detecting whales when they are actively vocalizing, and spatial precision based on a single receiving platform is low because whales typically have very large sound propagation ranges.

The Navy will issue real-time notifications of large whale aggregations (three or more whales) within 1 nm of a Navy vessel in a specified area of the SOCAL RC. These real-time notifications can complement the Navy's existing seasonal broadcast awareness messages to alert vessel captains and crews of the presence of large whale concentrations in specific areas, either seasonally or on a given day. However, the usefulness of either seasonal or real-time notifications in terms of reducing vessel strike risk is dependent on the proactive measures the vessel takes when transiting through these higher risk areas. This could include reducing vessel speed, adding Lookouts, changing course to avoid (or partially avoid) the area, or complementing visual observations with passive acoustic assets. To evaluate the effectiveness of

the Navy's large whale notifications in terms of reducing vessel strike risk we need information indicating that the Navy implements proactive measures to minimize the risk of vessel strike in scenarios when that risk is likely relatively high based on either historical information, modeling, past strikes, real-time observations or some combination of these.

The activity-based mitigation for vessel movement (Table 7) indicate that if a marine mammal or sea turtle is observed, in addition to maneuvering to maintain distance, vessel operators may reduce speed if the circumstances warrant. While this mitigation language highlights the importance of reducing speed as a vessel strike mitigation measure, it does not require Navy vessels to reduce speed. The Navy has indicated that any requirements to reduce vessel speeds restrict the options of vessel operators and have significant direct negative effects on HCTT mission effectiveness (Navy 2018). To evaluate the effectiveness of this mitigation measure at reducing vessel strike risk we would need more information regarding how often Navy vessels reduce speed after observing a large whale within the mitigation zone. In addition, the effectiveness of the vessel movement mitigation measure is directly linked to the effectiveness of Navy Lookouts at observing large whales within the mitigation zone. Based on the results of Oedekoven and Thomas (2022), and recent Navy whale strike history, in many instances large whales will not be observed by Lookouts in time to successfully implement the vessel movement mitigation measure.

The proposed HCTT geographic mitigation areas (Section 2.4.2) for marine mammals are primarily designed to avoid or reduce impacts from acoustic and explosive stressors from training and testing activities. The Navy used available information on identified marine mammal BIAs (Baird et al. 2015; Calambokidis et al. 2015) in the design of these mitigation areas. While the mitigation measures within these areas limit the level of activities involving surface ship hull-mounted mid-frequency active sonar (MF1) and explosive detonations, they would also likely reduce Navy vessel activity in these areas, which may, in turn, reduce large whale vessel strike risk within the mitigation area. For example, mitigation areas around the main Hawaiian Islands may limit Navy vessel activity in portions of the HRC identified as BIAs for breeding humpback whales from the non-listed Hawaii DPS from December through April. In California, HCTT mitigation areas have overlap with BIAs for blue whale feeding (i.e., Northern California Large Whale Mitigation Area, Central California Large Whale Mitigation Area, and Southern California Blue Whale Mitigation Area) and humpback whale feeding (i.e., Northern California Large Whale Mitigation Area, Central California Large Whale Mitigation Area). Limitations on the use of mid-frequency active sonar in these areas apply from June through October, when blue whales are most likely to be in these areas. These limitations may result in reduced Navy vessel activity in areas (and during times of year) where blue and humpback whales are likely to congregate off the California coast.

In general, the HCTT mitigation areas could potentially reduce the risk of Navy and Coast Guard vessel strike for particular species within certain areas and/or during certain times of year.

However, the effectiveness of the mitigation areas at reducing vessel strike risk for ESA-listed large whales within the HCTT action area overall may be somewhat limited for several reasons. First, while mitigation measures within these areas focus on activities involving sonar and explosives, other activities involving surface vessels (including transiting through the areas) would not be limited. As such, there are no specific measures (i.e., specific to these mitigation areas) restricting vessel activity, vessel speed, or vessel types used within these areas where aggregations of large whales are known to occur. Second, because the HCTT mitigation areas do not cover all areas or all times of year where aggregations of ESA-listed large whales could occur, the limitations within these areas may be shifting HCTT vessel activity to other areas where ESA-listed species would be at greater risk of vessel strike. In addition, the HCTT marine mammal mitigation areas do not overlap with any of the locations where recorded military vessel strikes of large whales since 2009 have occurred within SOCAL RC (i.e., south of San Clemente Island and Cortes Bank area).

The 2021 Australian Navy vessel strike raised questions about the mitigation requirements for foreign vessels if they are conducting activities with the U.S. Navy when operating in the HCTT action area. As pointed out by the Navy, as sovereign entities foreign vessels are not required to implement the same vessel strike mitigation measures as the U.S. Navy. Depending on the location of the foreign military vessel in relation to the U.S. territorial sea and its involvement in a U.S. Navy event, the Navy has indicated that it can either request (i.e., when within the territorial sea) or encourage (i.e., when beyond the territorial sea) the foreign vessel follow the U.S. Navy's mitigation measures when participating in a U.S. Navy-led exercise. While, in practice, many foreign vessels participating in U.S. Navy activities within HCTT would implement vessel strike mitigation measures that are similar to or the same as U.S. Navy measures, there is no guarantee that this would occur or that foreign mitigation measures would be as effective at minimizing vessel strike risk as U.S. Navy mitigation measures. The sovereign nature of foreign vessels limits U.S. government authority or control over those vessels under principles of international maritime law.

Vessel Strike Probability Calculation

Data from 2009 through August 2025 were used to calculate the most current probability of an Action Proponent vessel or USV striking a whale during HCTT activities. For our analysis, we calculated separate vessel or USV strike probabilities for the Navy and the Coast Guard. This time frame was selected because it represents a period of standardized mitigation, monitoring, and reporting requirements across Navy at-sea range complexes, and full implementation of improved marine species awareness training and mitigation measures specifically designed to minimize large whale interactions with Navy vessels. Thus, the period from 2009-2025 provides us with the most robust and representative time series available for estimating the future risk of vessel or USV strike. We recognize that additional mitigation measures, as well as changes in other vessel or USV strike risk factors (e.g., whale population abundances, distribution through space and time, oceanographic conditions, and ecological changes), could affect the reliability of

this time series for predicting future vessel strike risk. These potential changes are evaluated in our summary exposure analysis section below.

From 2009 through August 2025, a total of seven reported whale strikes have occurred from Navy training and testing activities in the action area, all within the SOCAL RC. These incidents were described in more detail above. The probability of a Navy vessel or USV strike to large whales is influenced by the amount of time at sea for Navy vessels within the action area. The estimated total number of at-sea days from 2009 through August 2025 for Navy manned vessels greater than 118 m (387 ft or Littoral Combat Ship size and above) is 36,306 days. The annual average number of Navy at-sea days over this time period is 2,178 days per year (Navy 2025e). This estimate is based on positional tracking data records from the Navy's Authoritative Maritime Services database for the years 2016-2023. Tracking data is only available for the larger Navy ship classes, which, as described above, represent the greatest vessel strike risk to large whales. The Navy used the average of the 2016-2023 annual values as a surrogate for annual at-sea days for each year from 2009 through 2015. The annual value for 2023 was used for 2024 and 2025, pro-rated through August 31 only (Navy 2025b).

The number of past Navy vessel strikes are combined with the at-sea vessel days to calculate a vessel strike rate based on the 2009 through August 2025 reporting period. Dividing the seven reported strikes by the total vessel at-sea days (i.e., 7/36,306) results in a strike rate of 0.000193 strikes per day.

Based on information provided by the Navy, we assume that the estimated average annual number of Navy HCTT at-sea vessel days over the 7-year period from December 2025 through December 2032 will be similar to the historical average from 2009-2025. However, in addition to the vessel activity during Phase III, for HCTT Phase IV USVs greater than 61 m (200 ft) will be added as a new category of surface vehicle for testing activities with an estimated annual average of 728 at-sea days per year. We add the large USV vessel activity (728 days per year) to the average annual days per year for other vessels (2,178) to get 2,906 at-sea days per year for manned Navy vessels greater than 127 m (418 ft) and large USVs. Therefore, the estimated total Navy vessel and USV at-sea days within the action area for HCTT Phase IV (i.e., from December 2025 through December 2032) is 20,345 days. The historic daily strike rate (0.000193 strikes per day) can be multiplied by the estimated Navy vessel and USV at-sea days for Phase IV (20,345) to estimate the number of large whale strikes from late 2025 through late 2032 as 3.92 whales.

From 2009-2024, the Coast Guard reported striking one large whale within the HCTT action area. The estimated total number of at-sea days from 2009 through August 2025 for Coast Guard vessels greater than 100 m (328 ft) is 4,351 days. The annual average number of Coast Guard at-sea days over this time period is 262 days per year (Navy 2025b). This estimate is based on positional data records from the Navy's Authoritative Maritime Services database using the same approach described above for Navy at-sea days calculations. Although the single reported Coast Guard large whale vessel strike was from a smaller craft (i.e., between 40-100 ft), tracking data

were not available for Coast Guard vessels less than 328 ft. Although the Coast Guard has a history of striking large whales with small to medium sized vessel, both within and outside the action area, we use the large vessel at-sea days estimates as the best available information for evaluating the risk of serious injury or mortality from vessel strike. As we discuss below in our response analysis, based on information provided by the Coast Guard, strikes of large whales by smaller vessel are often non-lethal.

The number of past Coast Guard vessel strikes are combined with the at-sea vessel days to calculate a vessel strike rate based on the 2009 through August 31, 2025 reporting period. Dividing the one reported strike by the total vessel at-sea days (i.e., 1 / 4,351) results in a strike rate of 0.00023 strikes per day.

Based on information provided by the Action Proponents, we assume that the estimated average annual number of Coast Guard HCTT at-sea vessel days over the 7-year period from December 2025 through December 2032 will be similar to the historical average from 2009-2025. However, to account for limitations in data availability, future new vessel or repositioning home port assignments, and the history of medium sized Coast Guard vessels striking large whales, the Coast Guard anticipates an additional 100 at-sea days per year during HCTT Phase IV activities (Navy 2025b). We add the 100 additional at-sea days to the average annual days per year for other vessels (262 days) to get 362 at-sea days per year for Coast Guard vessels greater than 100 m. Therefore, the estimated total Coast Guard at-sea days within the action area for HCTT Phase IV (i.e., from late 2025 through late 2032) is 2,534 days. The historic daily strike rate (0.00023 strikes per day) can be multiplied by the estimated Coast Guard vessel at-sea days for Phase IV (2,534) to estimate the number of large whale strikes at 0.58 whales from December 2025 through December 2032.

The probabilities of a specific number of strikes (n=0, 1, 2, etc.) over the 7-year period from December 2025 through December 2032 can be derived from a Poisson distribution. A Poisson distribution is often used to describe random occurrences when the probability of an occurrence is small, e.g., count data such as cetacean sighting data, or in this case strike data, often described as a Poisson or over-dispersed Poisson distribution. The formula for a Poisson distribution is:

$$P\left\langle n \middle| \mu \right\rangle = \frac{e^{-\mu} \bullet \mu^{n}}{n!}$$

P ($n|\mu$) is the probability of observing n events in some time interval, when the expected number of events in that time interval is μ. For this analysis, μ is the estimated strike rate of 3.92 vessel strikes for Navy vessels and USVs and 0.58 strikes for Coast Guard vessels for the period from December 2025 through December 2032.

From the 7-year Navy strike rate (3.92), the Poisson distribution can estimate the probability of n where n=0 (no strikes), 1 strike, 2 strikes, and 3 strikes over the 7-year period from December 2025 through December 2032 as follows:

- P(n =0) = 0.020 or a 2.0% chance of 0 strikes
- P(n > 0) = 0.980 or a 98.0% chance of more than 0 strikes

- P(n = 1) = 0.078 or a 7.8% chance of 1 strike
- P(n > 1) = 0.903 or a 90.3% chance of more than 1 strike
- P(n= 2) = 0.152 or a 15.2% chance of 2 strikes
- P(n > 2) = 0.750 or a 75.0% chance of more than 2 strikes
- P(n= 3) = 0.199 or a 19.9% chance of 3 strikes
- P(n > 3) = 0.551 or a 55.1% chance of more than 3 strikes
- P(n= 4) = 0.195 or a 19.5% chance of 4 strikes
- P(n > 4) = 0.356 or a 35.6% chance of more than 4 strikes
- P(n= 5) = 0.153 or a 15.3% chance of 5 strikes
- P(n > 5) = 0.203 or a 20.3% chance of more than 5 strikes
- P(n= 6) = 0.100 or a 10.0% chance of 6 strikes
- P(n > 6) = 0.103 or a 10.3% chance of more than 6 strikes
- P(n= 7) = 0.056 or a 5.6% chance of 7 strikes
- P(n > 7) = 0.047 or a 4.7% chance of more than 7 strikes

From the 7-year Coast Guard strike rate (0.58), the Poisson distribution can estimate the probability of *n* where n=0 (no strikes), 1 strike, and 2 strikes over the 7-year period from December 2025 through December 2032 as follows:

- P(n =0) = 0.559 or a 55.9% chance of 0 strikes
- P(n > 0) = 0.441 or a 44.1% chance of more than 0 strikes
- P(n = 1) = 0.325 or a 32.5% chance of 1 strike
- P(n > 1) = 0.116 or a 11.6% chance of more than 1 strike
- P(n= 2) = 0.095 or a 9.5% chance of 2 strikes
- P(n > 2) = 0.021 or a 2.1% chance of more than 2 strikes

Based on the resulting probabilities presented in the analysis above, there is an estimated 10.3% chance that the proposed action will result in more than six large whale vessel strikes by Navy vessels or USVs over the 7-year period from December 2025 through December 2032. Therefore, we anticipate that the Navy could strike up to seven large whales incidental to HCTT activities within the action area over the course of the HCTT Phase IV 7-year MMPA permit. The probability of striking more than seven large whales over this period is sufficiently low (i.e., less than 5%) and, therefore, considered unlikely to occur.

Based on the resulting probabilities presented in the analysis above, there is an estimated 11.6% chance that the proposed action will result in more than one large whale vessel strike by Coast Guard vessels over the 7-year period from December 2025 through December 2032. Therefore, we anticipate that the Coast Guard could strike up to two large whales incidental to HCTT activities within the action area over the course of the HCTT Phase IV 7-year MMPA permit. The probability of striking more than two large whales over this period is sufficiently low (i.e., less than 3%) and, therefore, considered unlikely to occur. These estimated strikes by Action

Proponent vessels or USVs could include a combination of ESA-listed and non-listed large whales. We discuss the probability of striking particular large whales species (or DPSs) below.

As previously noted, for our effects analysis we assume that HCTT activities by the Action Proponents during the Permits Division's proposed incidental take authorization pursuant to the MMPA will continue into the reasonably foreseeable future at levels similar to those assessed in this opinion. Therefore, in addition to the calculations above for the 7-year MMPA permit period, we anticipate that the proposed action will result in up to seven large whale Navy vessel or USV strikes and up to two Coast Guard vessel strike every seven years into the reasonably foreseeable future.

Next, we consider the likelihood of a Navy or Coast Guard vessel strike of each particular species (or DPS) of ESA-listed large whale within the HCTT action area. Many Navy and Coast Guard reported whale strikes are not identified to the species level, making it difficult to predict which species of large whales are most likely to be struck during future training and testing activities. In order to predict the likelihood of striking any particular species, we compiled information on detected annual rates of large whale mortality and serious injury (M/SI) from vessel strikes recorded in the Central North Pacific and Eastern North Pacific from recent years (i.e., 2017-2021, unless otherwise noted) based on information provided in the 2023 U.S. Pacific Marine Mammal Stock Assessment Report (Carretta et al. 2025). We recognize that not all ship strikes are documented or reported (e.g., See Rockwood et al. 2017), so not all instances of serious injury and mortality are represented in the annual rates reported in the SARs. However, the annual rates of large whale serious injury and mortality from vessel collisions in the SARs do provide a reasonable science-based representation of the relative susceptibility of different large whale species to vessel strike in the action area. To calculate the relative likelihood of striking each species, we summed the annual rates of mortality and serious injury, then divided each species' annual rate by this number (Table 49). Because the whales struck by the Action Proponents in previous years included non-ESA-listed large whales species (or DPSs) and future strikes could involve non-listed species, we include non-ESA-listed large whales in this calculation. The only non-ESA-listed large whale species in the North Pacific with an annual rate of serious injury and mortality from vessel strike greater than zero, based on the most recent 5-year data available for that species, were the Eastern North Pacific DPS gray whale and the Hawaii DPS humpback whale.

Based on the M/SI proportions in Table 49, if a large whale is struck within the HCTT action area the probability that it will be a particular species (or DPS) is highest for the non-listed Hawaii DPS humpback whale (33.3%), followed by the ESA-listed Central America and Mexico DPSs (26.3% combined), the non-listed Eastern North Pacific gray whale (18.2%), fin whale (16.2%), and blue whale (6.1%).

Except for the non-listed Hawaii DPS humpback whale, the vessel strikes of these species all occurred in the Eastern North Pacific. In general, with the exception of humpback whales, vessel

Table 49. Annual Rates of Mortality and Serious Injury (M/SI) for Large Whale Species (or DPSs) from Vessel Strikes in the Central and Eastern North Pacific from 2017-2021, and Relative Probability of Striking each Large Whale Species (or DPS)

Species / DPS	Annual rate of M/SI from vessel strike 2017-2021	Percent of total M/SI (all species)
Blue whale	0.6	6.1%
Fin whale	1.6	16.2%
Humpback whale (Central America and Mexico DPSs combined)	2.6[a]	26.3%
Humpback whale Hawaii DPS (non-ESA-listed)	3.3[a]	33.3%
Sei whale	0.0	0%
Gray whale Western North Pacific DPS	0.0	0%
Gray whale Eastern North Pacific DPS (non-ESA-listed)	1.8[b]	18.2%
Sperm whale	0.0	0%
Total for all species/DPSs combined	9.9	100%

[a] M/SI based on the years 2016-2020

[b] M/SI based on the years 2014-2018

strikes (both military and non-military) of other large whale species in the HRC are rare events (Carretta 2021b; Carretta 2022). Based on the recent history of both military and non-military vessel strikes within the HCTT action area, if the Action Proponents were to strike an ESA-listed large whale there is a higher probability that this would occur within the California portion of the action area as compared to the HRC. As noted above, nine of the last ten military vessel strikes involving large whales (dating back to 2009) within the HCTT action area have occurred within the SOCAL RC, and there has not been a documented Navy vessel strike associated with training and testing in the Hawaii portion of the action area since 2008. Although the last reported Coast Guard strike within the action area occurred within the HRC, the struck whale was identified as a non-listed Hawaii DPS humpback whale. Of the five large whales struck by Navy vessels in the

HRC from 1991-2008, only one (i.e., sperm whale struck in 2007) was confirmed to be an ESA-listed species (or DPS), while two were from the non-listed Hawaii humpback whale DPS, and two were unidentified. In addition, the vessel activity level for the larger Navy vessels that have historically been involved in large whale vessel strikes is about twice as great in SOCAL RC as compared to the HRC.

From the M/SI data in Table 49, there have been no reported strikes of either Western North Pacific DPS gray whales, sperm whales, or sei whales in the Central and Eastern North Pacific in recent years. In general, vessel strikes of these species within the HCTT action area are rare events (Carretta 2021b; Carretta 2022; Carretta et al. 2025). Given the very small overall population size, and low density of Western North Pacific DPS gray whales within the action area, we find it extremely unlikely (i.e., discountable) that an Action Proponent vessel or USV will strike a whale from this endangered population.

Four confirmed sperm whale vessel strikes have been recorded for the entire U.S. West coast and Alaska since 2007 (none off California[28]): two off Oregon, 2007 and 2023; two off Washington, 2007 and 2012. The last military vessel strike of a sperm whale within HCTT was a Navy vessel strike in 2007 in the HRC. Although historically the total strikes (i.e., military and non-military vessels) of sperm whales within the HCTT action area are low compared to other large whale species, since the Navy struck a sperm whale in the HRC within the past 20 years, we consider it reasonably likely that a Navy vessel or USV could strike a sperm whale over the HCTT Phase IV 7-year MMPA permit period.

From 2009-2024, the Coast Guard reported striking one sperm whale in the North Pacific but this strike occurred outside of the HCTT action area (i.e., 2017 strike in Samalga Pass, Alaska). The likelihood of a Coast Guard vessel striking a sperm whale within the HCTT action area is lower than that of a Navy vessel due to the following differences: 1) lower overall vessel traffic; 2) less vessel activity in offshore areas where sperm whale densities are highest; and 3) lower overall vessel strike risk based on the quantitative analysis above. For these reasons we find it extremely unlikely (i.e., discountable) that a Coast Guard vessel will strike a sperm whale.

Based on NMFS's stock assessment reports, the last recorded sei whale vessel strike within the HCTT action area was by a commercial vessel in 2015. However, there is one record in the NMFS West Coast Marine Mammal strandings database from 2019 of a sei whale stranded inside Anaheim Bay, California with possible signs of a propeller strike wound (NMFS 2024a). Although historically the total strikes of Eastern North Pacific sei whales are very low compared to other stocks, based on post-strike video, NMFS and the Navy think it is possible that the Navy strike in May 2023 was a sei whale. Therefore, we consider it reasonably likely that an Action

[28] The NMFS national strandings database shows one record of a sperm whale stranding off California in 2009 that could be considered an unconfirmed vessel strike. The remarks for this record note signs of blunt force trauma that could possibly indicate a vessel strike.

Proponent vessel or USV could strike a sei whale over the HCTT Phase IV 7-year MMPA permit period.

From our species probability analysis above, we determined that the large whale Navy vessel or USV strikes resulting from the proposed action (i.e., up to seven strikes over seven years) could include a combination of the following ESA-listed large whale species: blue, fin, sei, sperm, and humpback (Central America and Mexico DPSs). From our species probability analysis above, we determined that the large whale Coast Guard vessel strikes resulting from the proposed action (i.e., up to two strikes over seven years) could include a combination of the following ESA-listed large whale species: blue, fin, sei, and humpback (Central America and Mexico DPSs). The following non-ESA-listed species could also be struck by Navy and Coast Guard vessels or USVs: Eastern North Pacific DPS gray whale and Hawaii DPS humpback whale.

Because both Mexico DPS and Central America DPS humpback whales occur within the California portion of the action area, there is the potential that individual whales from either DPS could be struck as a result of the action. To determine which of these DPSs is most likely to be struck, and the relative probability of striking one DPS versus the other, we evaluated the relative abundance of each of these DPSs in California waters. Curtis et al. (2025) developed an approach to estimate proportions from different humpback whale populations in feeding areas off the U.S. West Coast based photo-identification matching rates to winter breeding areas. Within the latitude range from 31.7 N to 33.0 N, which encompasses the latitude range within SOCAL RC where all HCTT large whale vessel strikes have occurred since 2009, humpback whale DPSs are proportioned year-round as follows: 55% Mexico DPS; 44% Central America DPS, and less than 1% Hawaii DPS (A. Curtis, NOAA Fisheries, Southwest Fisheries Science Center personal communication to R. Salz, NOAA Fisheries, Office of Protected Resources, September 9, 2025). The estimated proportional distribution by DPS is reversed during the summer months (defined by Curtis et al. as June through October) with the Central America DPS accounting for 54% and the Mexico DPS accounting for 46% of humpback whales of the U.S. West Coast within this latitude range. Of the seven Navy large whale vessel strikes in this area since 2009, three occurred during these months. Based on the available information, we find that if a humpback whale is struck by a HCTT vessel in the California portion of the action area there is roughly an equal probability that it will be from either the Central America DPS or the Mexico DPS.

Although the Navy has not reported striking an Eastern North Pacific DPS gray whale during HCTT activities in the past, many Navy vessel strikes are reported as “unidentified species”, and Eastern North Pacific DPS gray whales are one of the most commonly struck species by commercial vessels on the U.S. West Coast. Similarly, the non-listed Hawaii DPS humpback whale is the most commonly struck large whale species (by all vessels) in the HRC, and Navy vessels have struck whales from this DPS in past years. From our M/SI analysis of species probabilities (Table 49), we find it reasonably likely that of the seven large whale strikes estimated every seven years from Navy vessels or USVs, at least one strike would be of either the non-listed Eastern North Pacific gray whale DPS or the non-listed Hawaii humpback whale

DPS. Therefore, after accounting for strikes of non-listed species (or DPSs), we estimate that up to six ESA-listed large whales will be struck by Navy vessels or USVs every seven years as a result of HCTT Phase IV activities.

The only reported Coast Guard large whale vessel strike since 2009 within the HCTT action area was of a non-listed Hawaii DPS humpback whale. The Coast Guard has also reported striking seven large whales along the U.S. West coast since 2009, all outside of the HCTT action area: two gray whales, two humpback whales, and three unidentified whales. Based on the recent history of Coast Guard vessel strikes, both within and outside the action area, we find it reasonably likely that of the two estimated Coast Guard strikes every seven years, at least one strike would be of a non-listed large whale species (either an Eastern North Pacific DPS gray whale or a Hawaii DPS humpback whale). Therefore, after accounting for strikes of non-listed species, we estimate that up to one ESA-listed large whale will be struck by a Coast Guard vessel every seven years as a result of HCTT Phase IV activities.

We next consider the likelihood that more than one whale from a particular ESA-listed species (or DPS) would be struck by a Navy vessel or USV over the 7-year MMPA permit period. From our M/SI species probability calculation for the SOCAL RC, we determined that fin whales had the second highest (after humpback whales, which combined Central America and Mexico DPSs) vessel strike probability (16.2%%) amongst ESA-listed species. The recent history of military vessel strikes within SOCAL RC suggests that our species-specific probability calculations may underestimate the chance of fin whales being struck by Navy vessels or USVs. The past four military vessel strikes of large whales in SOCAL RC (from 2009-2021) where the species could be identified have all involved fin whales. There were two documented instances of Navy vessel fin whale strikes in SOCAL RC in 2009. In 2021, an Australian Navy vessel, that was positioning to participate in a U.S. Navy-led exercise in SOCAL RC, struck two fin whales during a single incident. In addition, of the three unidentified large whales struck by Navy vessels from 2021 through mid-2023, all three could very well have been fin whales based on the offshore location (Calambokidis et al. 2015; Rockwood et al. 2017) and time of year (Candy et al. 2015) of these strikes, as well as video images from one of the strikes. This is further supported by model-predicted densities of large whales in the Southern California Bight from May through July 2021 (the time period when these unidentified whale strikes occurred) that estimated fin whale abundance as being nearly an order of magnitude higher than either blue or humpback whale abundance during this period (Becker et al. 2020; Zickel MJ et al. 2021). Ship-whale encounter models for the U.S. West Coast EEZ also indicated that vessel strike mortality estimates for fin whales were significantly higher than for blue whales and humpback whales (Rockwood et al. 2017). The comparatively higher modeled vessel strike rates for fin whales result from both the larger population and the more offshore distribution, which overlaps significantly with several major shipping routes for a much greater spatial extent (Rockwood et al. 2017).

Based on 1,243 visual boat-based sightings of 2,638 fin whales from 1991-2011, Calambokidis et al. (2015) found fin whale concentration areas included the San Clemente Basin, Tanner and Cortez Banks, and the shelf edge west of San Nicolas Island. These reported fin whale concentration areas overlap significantly with the areas where most of the recent Navy vessel strikes have occurred (Figure 75).

There are two different populations of fin whales that occur in the Southern California Bight: a seasonal population, and a localized population that occurs year-round with offshore/inshore movements (Campbell et al. 2015; Falcone et al. 2022). The presence of a local population likely makes fin whales more susceptible to vessel strike year-round, as compared to other large whale species that may only occur within the action area seasonally. This localized fin whale population could be at greater overall risk due to its smaller size, as compared to the migrating population, and its use of the nearshore waters of the Southern California Bight, where they are exposed to significantly elevated levels of anthropogenic activity (Falcone et al. 2022). Therefore, the recent history of military vessel strikes within HCTT, combined with other available information, suggests a higher likelihood of a Navy fin whale strike as compared to other species. Our evaluation of past military vessel strikes in HCTT, and other species specific risk information, indicates a relatively high likelihood of multiple fin whale strikes over the 7-year MMPA permit period. Because military vessel strikes of fin whales in HCTT have been clustered in past years (i.e., two in 2009; two to four in 2021), and other information suggests a higher probability of a fin whale strike by military vessels or USVs compared to other species (Becker et al. 2020; Zickel MJ et al. 2021), we find it reasonable to assume that up to five fin whale strikes by Navy vessels or USVs could occur every seven years of the proposed action.

Since 1991, there is only one confirmed incident of a blue whale vessel strike (2004 mortality in SOCAL RC) and one confirmed incident of a sperm whale vessel strike (2007 in the HRC) by Navy vessels within the HCTT action area. However, as discussed above, the August 2025 Navy strike could possibly have been the same blue whale that washed up on San Clemente Island four days later, based on the stranding location and timing in relation to the strike location and timing. While recent commercial vessel strikes of blue whales off California are more common (i.e., four recorded from 2015-2021), from 2007-2021 there are no recorded incidents of a blue whale vessel strike off San Diego County, the area within SOCAL RC where Navy vessel traffic will be most concentrated. Sperm whale vessel strikes (by all vessels) are generally rare events throughout the action area. Therefore, based on these factors we consider it unlikely that more than one blue whale or sperm whale strike by a Navy vessel or USV will occur over the 7-year MMPA permit period. Similarly, sei whale strikes in SOCAL RC are very rare historically, with only one such strike (from any type of vessel) recorded since 2010. As such, we consider it unlikely that more than one sei whale strike by a Navy vessel or USV will occur over the 7-year MMPA permit period.

Although humpback whales are more commonly struck by vessels along the US. West Coast as compared to either sei or blue whales, all recorded incidents of Navy vessel humpback whale

strikes within the HCTT action area occurred in Hawaii, with the last one occurring in 2003. Although some of the unidentified whales struck by Navy vessels in SOCAL RC may have been humpbacks, we have no recorded incidents of a Navy vessel strike of an ESA-listed DPS humpback whale in the action area. As discussed above, if a humpback whale is struck in the California portion of the action area we expect that there is roughly an equal probability that it will be from either the Central America DPS or the Mexico DPS. Based on the recent history of Navy vessel strikes, we find it unlikely that Navy vessels or USVs will strike two humpback whales from the same ESA-listed DPS over the 7-year MMPA period.

Our vessel strike probability calculations above are based on recorded Navy and Coast Guard vessel strikes within the HCTT action area over the past 16 plus years (2009 through August 2025). While information on past military vessel strikes can serve as a reasonable indicator of future vessel strike risk, we also need to consider how future conditions may differ from the past in ways that could influence the likelihood of a large whale vessel strike occurring as a result of the action. In general, the magnitude of vessel strike risk may be increasing over time as many whale populations are gradually recovering from centuries of commercial whaling (Redfern et al. 2020). The years 2018, 2019, and 2021 were the worst years on record for vessel strikes in general off the coast of California. Increased vessel strike risk off California in recent decades has been associated with increases in the abundance of fin and humpback whale populations in the North Pacific (Redfern et al. 2020). A continuing trend of increased fin whale abundance along the U.S. West Coast between 1991 and 2018 was documented in the U.S. Pacific Marine Mammal Stock Assessment Report for 2021 (Carretta 2022). It has also been suggested that the blue whale population in the Eastern North Pacific, inclusive of the SOCAL RC portion of the action area, is at carrying capacity and recovered to pre-whaling levels (Monnahan et al. 2014). In addition, the magnitude of risk may also be affected by shifts in whale distributions over time in response to environmental factors including changing environmental conditions, marine heatwaves, and associated changes in prey distribution.

Historically, military vessel strikes of large whales within the HCTT action area have been rare events, with only 10 such strikes (seven Navy, one Coast Guard, and two Royal Australian Navy) occurring over the past 17 years. However, the fact that two of these strikes occurred within a 2-month period (July-August) in 2025, four of these strikes occurred within a 3-month period (May-July) in 2021, and two occurred within a 4-month period (February-May) in 2009, suggests that military vessel strikes in California can be both highly episodic and clustered.

Results from a survey of whale watching vessel operators and crew in Southern California, combined with remote sensing data in the area, suggest that the number of large whales may have been greater in May through July of 2021 compared with previous years in certain high military vessel traffic and "core" use HCTT areas off Southern California, particularly farther offshore, as well as closer to shore off San Diego Bay (Zickel MJ et al. 2021). The higher density of large whales in the SOCAL RC portion of HCTT in 2021 is further supported by data provided by the Navy on the number of times Navy vessels participating in a MTE had to

maneuver to maintain the specified mitigation zone distance (i.e., 500 yards) from large whales (Navy 2023). The number of vessel maneuvers in 2021 (82) to avoid large whales during MTEs in SOCAL RC was the highest in the time series from 2009-2025, and was nearly four times the average (20.4) for that 17-year period. This suggests that whale densities in the late spring and early summer of 2021, which coincided with the four military whale strikes in the SOCAL RC, may have been considerably higher that year as compared to other years in the time series.

Overall, the significant increase in the military vessel strike rate over the past five years (2021-2025), as compared to the earlier years in our time series (i.e., 2009-2020), combined with the other strike risk factors discussed above, suggests an increasing trend in military vessel strike risk within SOCAL RC.

Exposure Analysis Summary

Our vessel strike exposure analysis was based on the results of our vessel strike probability calculations in combination with supplemental information including an evaluation of vessel strike risk factors, recent military and non-military vessel strike history in the action area, and species specific biological and life history information. In summary, we anticipate the following number of ESA-listed large whale vessel strikes as a result of the proposed action:

- Up to six large whale strikes by Navy vessels or USVs over the 7-year MMPA permit period, and over every seven years of the proposed action into the foreseeable future. This 7-year total can include any combination of up to five fin whales, up to one sei whale, up to one blue whale, up to one sperm whale, up to one Mexico DPS humpback whale, and up to one Central America DPS humpback whale.
- Up to one large whale strike by Coast Guard vessels over the 7-year MMPA permit period, and over every seven years of the proposed action into the foreseeable future. The one large whale strike over every seven years can be from any of the following species: fin whale, sei whale, blue whale, or humpback whale (either Mexico DPS or Central America DPS).

The available information does not allow for a detailed analysis of the sexes or life stages of large whales that we would expect to be struck by vessels as a result of the action. Based on strandings data reports from California, we assume that for all affected ESA-listed species (i.e., fin whales, sei whales, blue whales, and humpback whales) vessel strikes could include a combination of both males and females, and any of the following life stages: adults, subadults, yearlings, and calves.

The Action Proponents have proposed mitigation measures that could potentially reduce the risk of vessel strike in the HCTT action area (see Section 2.4.2 for details). As discussed above, the effectiveness of these new measures is largely dependent on the proactive actions of Navy and Coast Guard vessel operators in response to additional information indicating increased strike risk, including reducing vessel speed and/or changing course to avoid areas with concentrations of large whales. Since we do not have enough information to determine if these mitigation

measures will appreciably reduce vessel strike risk below that which was estimated in our exposure analysis, we did not adjust our quantitative estimate of vessel strike risk to account for mitigation measures. However, as discussed above, these mitigation measures are considered qualitatively in our overall large whale vessel strike effects analysis.

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 (Knowlton and Kraus 2001; Laist et al. 2001; Vanderlaan and Taggart 2007; Conn and Silber 2013). Higher speeds during collisions result in greater force of impact and also appear to increase the chance of severe injuries or death (Silber et al. 2010; Gende et al. 2011).

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). Laist et al. (2001) found that most large whale deaths from vessel strike occurred when a vessel was traveling in excess of 13 knots. Jensen and Silber (2004) 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%) 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%) 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% as vessel speed increased from 10 to 14 knots, and exceeded 90% at 17 knots. While modeling studies have suggested that hydrodynamic forces pulling whales toward the vessel hull increase with increasing speed (Knowlton et al. 1995; Clyne et al. 1999), 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 another 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% at 15 knots to approximately 20% at 8.6 knots. At speeds below 11.8 knots, the chances of lethal injury drop below 50%, while the probability asymptotically increases toward 100% above 15 knots.

Vessel size also plays a role in the anticipated outcome when a large whale vessel strike occurs. Kelley et al. (2021) found that while vessels of all sizes can cause lethal injuries to whales, large vessels (> 65 ft) produce much larger stresses even when travelling at reduced speeds (i.e., 10 knots). Based on the recent history of past vessel strikes, we anticipate that Navy vessel or USV strikes within the HCTT action area will likely involve very large (i.e., > 300 ft) vessels or USVs, while Coast Guard vessel strikes will likely involve a range of vessel sizes from small to medium vessels (e.g., 45 ft Response Boat, 154 ft Cutter) to very large vessels (e.g., 418 ft Cutter).

Our exposure analysis considered vessel strike of large whales comprehensively, as a result of all Navy and Coast Guard 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 or Coast Guard vessels or USVs that are expected to result in ship strikes of large whales. Because of these unknowns, we assume that all incidents of ESA-listed large whale vessel or USV strike associated with HCTT activities in the action area will result in mortality or serious injury leading to mortality.

6.1.4 Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat

In this section we analyze the effects of the proposed action on MHI IFKW critical habitat. As discussed in Section 4.1.3, we determined that the proposed action was not likely to adversely affect critical habitat of other marine mammals in the action area. In Section 4.2.7 above, we describe the designated critical habitat for the MHI IFKW, including a discussion of the characteristics of the essential PBF. The MHI IFKW designated critical habitat includes one PBF essential for conservation of the species, with the following four characteristics:

1. Adequate space for movement and use within shelf and slope habitat;
2. Prey species of sufficient quantity, quality, and availability to support individual growth; reproduction, and development, as well as overall population growth;
3. Waters free of pollutants of a type and amount harmful to MHI IFKWs;
4. Sound levels that would not significantly impair false killer whales' use or occupancy.

Given the stressors associated with the proposed action (Section 2.5), we find no pathway for the action to affect IFKW adequate space for movement and use of critical habitat (characteristic #1 above). In Section 4.1.1 we analyzed the indirect effects, including water quality, of stressors from HCTT activities to ESA-listed species habitat. We concluded that any effects from explosives, explosive byproducts and unexploded munitions, metals, or chemicals to water quality would be insignificant (i.e., so minor as to not be measureable). For the same reasons as described in Section 4.1.1, we find that an effects of the proposed action on the water quality characteristic (#3 above) of critical habitat to be insignificant. Therefore, this section focuses on PBF characteristics #2 (i.e., prey species) and #4 (i.e., sound levels) above. In this section we analyze the potential effects of the following stressors on these characteristics of the PBF of MHI IFKW critical habitat: vessel noise, physical disturbance and strike, explosives, and sonar and other transducers.

Vessel Noise

Additional discussion on vessel noise as a potential stressor is included in Section 2.5.2 and Section 4.1.1. Action Proponent vessels (including large ships and small craft) produce low-frequency, broadband underwater sound, though the exact level of noise produced varies by vessel type. HCTT 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. Activities involving vessel movements occur intermittently and are variable in duration, ranging from a few hours up to two weeks. HCTT vessel traffic could occur throughout designated critical habitat for MHI IFKWs and would also occur in other portions of the action area.

The analysis below describes the potential effects of HCTT vessel noise on the following characteristics of the PBF of MHI IFKW designated critical habitat: 1) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and 2) sound levels that would not significantly impair false killer whales' use or occupancy.

All fish species can detect vessel noise due to its low-frequency content and their hearing capabilities. Similarly, available information indicates that aquatic invertebrates, such as squid, are primarily sensitive to low frequency sounds. Most cephalopods (e.g., octopus and squid) likely sense low-frequency sound below one kHz, with best sensitivities at lower frequencies (Packard et al. 1990; Budelmann 1992b; Mooney et al. 2010). Given this, false killer whale fish and cephalopod prey species in designated critical habitat are likely to be exposed to and detect sounds emitted from Action Proponent vessels. Due to the characteristics of vessel noise, sound produced from military vessels is unlikely to result in direct injury, hearing impairment, or other trauma to fishes or cephalopods. While behavioral and/or physiological responses could occur, impacts from 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 or cephalopods exposed. Instead, the only impacts expected from exposure to Action Proponent vessel noise for fishes and cephalopods may include temporary auditory masking, short-term physiological stress, or minor changes in behavior. For these reasons, exposure to vessel noise is not expected to result in detectable impacts to the quantity, quality, or availability of false killer whale prey. Therefore, the effects of vessel noise on the prey species characteristic of the PBF for MHI IFKW designated critical habitat are considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Regarding sound levels that would not significantly impair false killer whales' use or occupancy, the final rule designating critical habitat (83Fed. Reg. 35062) explains: "*scientific information also indicates that the introduction of a permanent or chronic noise source can degrade the value of habitat by interfering with the sound-reliant animal's ability to gain benefits from that habitat, impeding reproduction, foraging, or communication (i.e., altering the conservation value of the habitat)… chronic exposure to noise as well as persistent noise may impede the population's ability to use the habitat for foraging, navigating, and communicating, and may*

deter MHI IFKWs from using the habitat entirely." Thus, our analysis on whether Action Proponent vessel noise would significantly impair MHI IFKW use or occupancy considers how the introduction of noise caused by HCTT vessels may or may not impede the population's use of designated critical habitat for important biological functions to determine whether the introduction of noise from vessels alters the conservation value of the habitat. Our analysis of individual animal responses to HCTT vessels can be used to indicate whether exposure to vessel noise could impair use or occupancy of the habitat at a population level.

A discussion on anticipated effects of noise from Action Proponent vessels on marine mammals, including IFKWs, is included in Section 4.1.1. In that section, we summarize that ESA-listed marine mammals in the action area (inclusive of marine mammals in MHI IFKW critical habitat) are either not likely to respond to HCTT 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, in that section we summarize that the effects of any temporary masking specifically from HCTT vessels on marine mammals, including IFKWs, is insignificant given the background noise levels in the action area independent of HCTT vessels and the small percentage of vessel traffic HCTT vessels represent in the action area (and in designated critical habitat). We do not anticipate Action Proponent vessel noise would result in disruptions of normal behavior patterns or masking that could be meaningfully evaluated. We do not anticipate vessel noise to generate sounds that would significantly impair false killer whales' use or occupancy by inhibiting MHI IFKW's ability to receive and interpret sound for the purposes of navigation, communication, and detection of predators and prey. Therefore, effects of vessel noise on the sound level characteristic of MHI IFKW designated critical habitat are so minor that the effect cannot be meaningfully evaluated, and we therefore consider them to be insignificant. Thus, we conclude that the effects of vessel noise resulting from the proposed action may affect, but are not likely to adversely affect MHI IFKW designated critical habitat.

Aircraft Noise

The analysis below describes the potential effects of HCTT aircraft noise on the following characteristics of the PBF of MHI IFKW designated critical habitat: 1) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and 2) sound levels that would not significantly impair false killer whales' use or occupancy. Additional discussion on vessel noise as a potential stressor is included in Section 2.5.1.

Direct injury and hearing impairment in pelagic fishes and cephalopods (e.g., squid) 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 under water. 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. Thus, we conclude that the effects of aircraft overflight noise and disturbance resulting from the proposed action are considered insignificant, and are not likely to cause

adverse effects to the prey species characteristic of the PBF of MHI IFKW designated critical habitat.

Similar to our analysis of vessel noise above, we do not anticipate Action Proponent aircraft noise would result in disruptions of normal MHI IFKW behavior patterns or masking that could be meaningfully evaluated (see Section 4.1.1.1.2 for details). We do not anticipate aircraft noise to generate sounds that would significantly impair false killer whales' use or occupancy by inhibiting their ability to receive and interpret sound for the purposes of navigation, communication, and detection of predators and prey. Therefore, effects of aircraft noise on the sound level characteristic of MHI IFKW designated critical habitat are so minor that the effect cannot be meaningfully evaluated and we therefore consider them to be insignificant. Thus, we conclude that the effects of aircraft noise resulting from the proposed action may affect, but is not likely to adversely affect MHI IFKW designated critical habitat.

Weapons Noise

The analysis below describes the potential effects of HCTT weapons noise on the following characteristics of the PBF of MHI IFKW designated critical habitat: 1) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and 2) sound levels that would not significantly impair false killer whales' use or occupancy. Additional discussion on vessel noise as a potential stressor is included in Section 2.5.1.

All of the noise produced during the use of the weapons described here is expected to be brief (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. Gunfire activities could produce multiple shots fired over a brief period of time.

Weapons noise could affect MHI IFKW fish and invertebrate (i.e., squid) prey located at or near the surface of the water, albeit in a narrow footprint under a weapons trajectory, as described previously. For exposed prey species, 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. HCTT gunfire could elicit a brief behavioral reaction such as startle reactions or avoidance and could expose fish and squid to multiple shots within a few seconds. 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 fish and squid under water. These are launched from aircraft which would produce minimal sound in the water due to the altitude of the aircraft when these are fired.

Additionally, because these activities are brief in duration and anticipated firing of weapons within MHI IFKW critical habitat will be limited, accumulation of levels high enough to cause TTS or masking of biologically relevant sound for fish and squid is also extremely unlikely. Exposure of prey species to the sound produced from weapons would only be expected to cause brief behavioral or stress responses should they detect the noise. Fish and squid 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, exposed fish and squid 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.

In summary, any effects on IFKW fish or invertebrate prey from weapons noise are anticipated to be minor, temporary, and are not expected to lead to a significant disruption of normal behavioral patterns. As such, the effects from weapons noise on the prey characteristic of the PBF of MHI IFKW critical habitat are considered insignificant.

Given the brief, transient, and limited nature of weapons firing noise within MHI IFKW critical habitat, we do not anticipate Action Proponent weapons noise would result in disruptions of normal MHI IFKW behavior patterns or masking that could be meaningfully evaluated (see Section 4.1.1 for details). In addition, we do not anticipate weapons noise to generate sounds that would significantly impair false killer whales' use or occupancy by inhibiting their ability to receive and interpret sound for the purposes of navigation, communication, and detection of predators and prey. Therefore, effects of weapons noise on the sound level characteristic of MHI IFKW designated critical habitat are so minor that the effect cannot be meaningfully evaluated, and we therefore consider them to be insignificant.

Thus, we conclude that the effects of weapons noise resulting from the proposed action may affect, but is not likely to adversely affect MHI IFKW designated critical habitat.

Physical Disturbance and Strike

False killer whale prey species have the potential to be susceptible to physical disturbance and strike from vessels and in-water devices. Consistent with the analysis for the ESA-listed fishes presented in Section 4.1.1 and Section 6.3.1, we anticipate that MHI IFKW prey species 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). Further, prey species of MHI IFKWs (e.g., tuna, marlin, and mahi mahi) are not generally considered vulnerable to vessel strike when compared with some other large, slow-moving species that occur regularly at the surface such as ocean sunfish. For these reasons, it is extremely unlikely that an Action Proponent vessel or in-water device associated with HCTT activities would strike MHI IFKW prey. The effects of strike on the prey species characteristic of MHI IFKW designated critical habitat is thus discountable.

Prey species may exhibit a temporary behavioral response to oncoming vessels or other in-water devices and 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. Potential implications of fish behavioral avoidance response to vessels was addressed above in Section 4.1.1. Consistent with that analysis, behavioral avoidance and associated stress responses from detection of vessels or other in-water devices is not expected to result in impacts to the quantity, quality, or availability of MHI IFKW prey (i.e., fish and cephalopods). Therefore, the effects of such responses on the prey species

characteristic of MHI IFKW designated critical habitat are considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Explosives and Air Guns

Below we summarize the analysis on the potential effects of HCTT explosive and air gun use on the following characteristics of the PBF of MHI IFKW designated critical habitat: 1) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and 2) sound levels that would not significantly impair false killer whales' use or occupancy.

As described previously in Section 2.5.2, 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 Action Proponents were not able to provide specific estimates of the number and size of explosives or air gun shots that will be used within MHI IFKW critical habitat versus other portions of the action area. Within MHI IFKW critical habitat, we anticipate that explosives would be used infrequently, during unit level training or testing activities, and would primarily involve ordnance in lower net explosive weight bins. Although approximately 6,000 bin-counts of explosives are proposed in the HRC, critical habitat overlaps only 6% of the entire area (Navy 2025a). Thus, the large majority of HCTT explosive use in the HRC occurs in designated areas outside of critical habitat designated for MHI IFKWs. As discussed in Section 2.4.2, the Action Proponents will not detonate in-water explosives (including underwater explosives and explosives deployed against surface targets) in either the Hawaii Island Marine Mammal Mitigation Area or the Hawaii 4-Islands Region Mitigation Area (Figure 76). These areas encompass approximately 46% (i.e., 21,379 km^2) of designated critical habitat for MHI IFKWs (Navy 2025l). Activities involving the highest net explosive weight detonations (e.g., Ship Shock Trials, Underwater Demolitions, and Sinking Exercises) are not planned within MHI IFKW critical habitat. Higher explosive weight bins (≥ E8 [i.e., ≥ 60 lb. NEW]) comprise less than 5% of the explosives in the HRC and would typically be used in scheduled offshore subareas outside of critical habitat (Navy 2025l). Just under half of the explosives used in the area would have very low explosive weight bins (E1-E2 [i.e., < 0.5 lb. NEW]) and over 90% of the explosives used in the area would be in explosive weight bin E5 or lower (i.e., < 10 lb. NEW) (Navy 2025l).

From our species acoustic exposure analysis, we anticipate a very small number (i.e., less than one significant behavioral response and one TTS per year) of MHI IFKWs will be exposed to explosives at levels resulting in an adverse effect (see Table 46). Although limited use of air guns could occur in MHI IFKW critical habitat (Navy 2025l), Based on NAEMO results, we do

not anticipate any impacts to MHI IFKWs from this sound source, either within or outside of critical habitat.

Prey Resources

Fishes can experience a range of effects from exposure to impulsive sound such as those produced by explosives, including mortality, auditory injury, barotrauma, and behavioral changes. Explosives and other impulsive sources generally have an increased capacity to induce physical injuries in fishes, especially those with swim bladders (Casper et al. 2012; Casper et al. 2013; Popper et al. 2014a) when compared with other sound sources. Fishes can survive and recover from some injuries, but in other cases, death can be instantaneous, occur within minutes after exposure, or occur several hours to days later. Other MHI IFKW prey items, such as squid, are also susceptible to effects from explosives. Similar to fishes, these animals may be injured or killed, or experience behavioral changes (e.g., McCauley et al. 2000).

The factors described above regarding Navy explosive use in the HRC indicate that in-water explosive use within designated critical habitat for MHI IFKWs would not be common, and when it does occur, the most likely explosives that will be used will be of small net explosive weight. These smaller bins produce relatively smaller ranges to effects such as mortality or injury compared to larger bin sizes, thus reducing the potential that IFKW prey would incur impacts leading to fitness consequences from most explosives. Only MHI IFKW prey within these ranges at the instant an explosion occurred would experience injury or mortality.

False killer whale prey is widely distributed within and outside of critical habitat. The final critical habitat biological report was unable to determine where prey resources of higher value exist for MHI IFKWs within or outside designated critical habitat (NMFS 2018c). While some instances of MHI IFKW prey injury or mortality could occur within designated critical habitat, the vast majority of prey within critical habitat will not be exposed to explosives due to the: 1) large area that has been designated critical habitat; 2) limited use of explosives in critical habitat; 3) mitigation areas overlapping with critical habitat where in-water explosives would not be used; and 4) fact that most explosives used in critical habitat will be from the smaller explosive bins with relatively shorter ranges to adverse effects.

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. Considering the mitigation areas and the limited overlap with locations in the HRC where explosives could be used, effects on critical habitat are unlikely to affect prey of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth. 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 an IFKW. 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). For all MHI IFKW prey species, 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 shortly after each explosion.

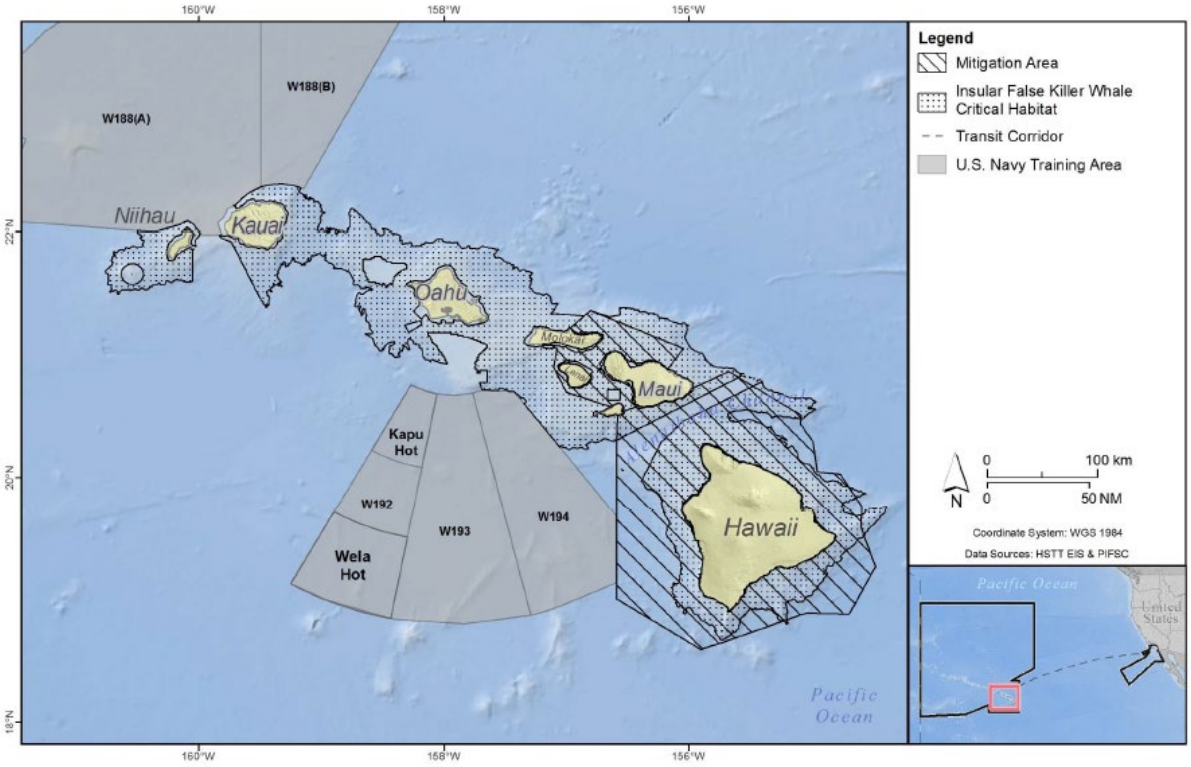


Figure 76. MHI IFKW Designated Critical Habitat and Hawaii Range Complex Sub-areas where Most Explosives are Used During Training and Testing

Air guns may affect prey species very close to the source, although the air guns used during HCTT activities are less powerful than those used in seismic surveys (Navy 2025l). Any impacts to prey are expected to be minimal, localized, and would not appreciably reduce overall aggregations of prey species available to MHI IFKWs.

For the reasons above, the effects of explosives and air guns on the prey species characteristic of MHI IFKW designated critical habitat are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Sound Levels

Explosions and air gun shots in the ocean or near the water surface can introduce loud, impulsive, broadband sounds into the marine environment. In contrast to acoustic sources such as sonar or low-frequency sound from vessels, sound from HCTT explosives and air guns lasts a very brief period of time (i.e., microseconds). Sounds from explosives and air guns are transient,

and do not persist in the marine environment. Further, HCTT activities involving impulsive sounds sources typically consist of a single or multiple explosions or air gun shots occurring over a short period of time in a relatively small area. Additionally, as described above, in-water explosive use within designated critical habitat for MHI IFKWs would not be common, and when it does occur the most likely explosives used will be of small net explosive weight, resulting in relatively small zones of impact or sound propagation for each detonation. Air guns use within MHI IFKW critical habitat is anticipated to be very limited (i.e., no more than two events every seven years) (Navy 2025k). In addition, the types of air guns used for HCTT are relatively low power compared to those used for seismic surveys (see Section 2.5.1 for details). These factors greatly limit both the frequency and duration of sound exposure from explosives and air guns to the water column within MHI IFKW critical habitat.

In Section 6.1.1 we discuss the range of behavioral responses of marine mammals when exposed to impulsive sound sources. This information is pertinent to evaluating how an activity, such as Navy training and testing using explosives and air guns, may affect MHI IFKW critical habitat because the characteristic of the PBF under consideration is how sound levels may impact the animals themselves within the habitat. That is, the characteristic is defined as "…*sound levels that would not significantly impair false killer whales' use or occupancy*" of the habitat. The final rule designating critical habitat explains: "*scientific information also indicates that the introduction of a permanent or chronic noise source can degrade the value of habitat by interfering with the sound-reliant animals ability to gain benefits from that habitat, impeding reproduction, foraging, or communication (i.e., altering the conservation value of the habitat)… chronic exposure to noise as well as persistent noise may impede the population's ability to use the habitat for foraging, navigating, and communicating, and may deter MHI IFKWs from using the habitat entirely.*" Thus, our analysis considers how the introduction of noise caused by HCTT explosives and air guns may or may not impede the population's use or occupancy of designated critical habitat for important biological functions to determine whether the introduction of noise alters the conservation value of the habitat. Our analysis of individual animal response to HCTT explosive use can be used to indicate whether these responses could impair use or occupancy of the habitat at a population level.

Available information on the effects of explosives and air guns on marine mammal behavior 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 sound producing activity ends. As described in Section 6.1.1, to estimate impacts from impulsive sources associated with the proposed training and testing activities, the Action Proponents performed a quantitative analysis to estimate the number of instances that could result in effects to ESA-listed marine mammals (including MHI IFKWs) and the magnitude of those effects (e.g., injury, hearing loss, behavioral response). The quantitative analysis utilizes NAEMO and takes into account criteria and thresholds used to predict impacts in conjunction with spatial densities of species within the action area. A detailed explanation of this analysis is in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training and Testing* (Navy 2024h).

As shown in Table 46, based on NAEMO, on average, we anticipated less than two instances annually where explosives would likely result in harassment (i.e., significant behavioral response or TTS) of an IFKW. It should be noted that the modeled group size for this DPS (i.e., 19 animals) is much higher than this annual average. Thus, in those rare years when a group of MHI IFKWs is exposed to sound from Navy explosives at levels resulting in an adverse effect, we would expect the number of animals affected to be somewhat greater than the annual average predicted by NAEMO. These exposures could occur inside or outside of designated critical habitat. No impacts to IFKWs were predicted by NAEMO from HCTT air gun use.

Additionally, based on best available information, we do not anticipate explosive or air gun use will lead to long duration abandonment or avoidance of any particular area within critical habitat, or of critical habitat as a whole. When marine mammals have been observed to leave a certain area following acoustic exposure, they are likely to return after the period of exposure ends (e.g., Tyack et al. 2011a; Baird et al. 2016). Even in intensively used training areas, photo identification results have indicated long-term residency of odontocete species (i.e., Cuvier's beaked whales) considered particularly sensitive to disturbance from acoustic exposure (Falcone et al. 2009; Falcone and Schorr 2014). IFKWs are not known to be as sensitive to acoustic disturbance as beaked whales. Even with decades of military readiness activities occurring in areas recently designated as critical habitat, as well as monitoring of IFKW use of this habitat, we do not have information to suggest that IFKWs are avoiding or abandoning any portions of critical habitat where impulsive sound sources are used, or critical habitat as a whole, due to explosive or air gun use. Given this information and the anticipated frequency, duration, and sporadic nature of explosive and air gun use in designated critical habitat described above, we find it extremely unlikely (i.e., discountable) that areas are being avoided to such an extent that it is affecting IFKW use or occupancy of critical habitat.

In summary, significant effects to the use or occupancy of MHI IFKW critical habitat due to noise generated by explosives and air guns are not anticipated due to the following factors: 1) the short duration of sounds emitted by these impulsive sources, 2) the infrequent use of explosives and air guns by Action Proponents in MHI IFKW designated critical habitat, 3) the widely dispersed nature of explosive and air gun use in the HRC, 4) the mitigation requirements in place restricting in-water explosive use throughout a large portion of critical habitat, and 5) the quantitative analysis estimate that less than two IFKWs will experience harassment from explosives annually (and zero impacts from air gun use), and that disruption would be for a short duration. Thus, effects of explosives and air guns on the sound level characteristic of MHI IFKW designated critical habitat are considered insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Sonar and Other Transducers

As described previously in Section 2.5.1, sonar and other transducers include 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 and high-

frequency small object detection sonars used to detect mines. The analysis below describes the potential effects of Action Proponent sonar and other transducers use on the following characteristics of the PBF of MHI IFKW designated critical habitat: 1) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and 2) sound levels that would not significantly impair false killer whales' use or occupancy.

Prey Resources

As previously discussed, false killer whale prey may include various tuna species, marlin species, jack species, mahi mahi, wahoo, moonfish, and squid (NMFS 2018c). The potential effects of Action Proponent sonar on ESA-listed fishes was described in detail in Section 6.3.1. Similar effects would be anticipated to fishes that are prey to false killer whales. To summarize, direct injury or mortality from sonar and other transducers is extremely unlikely 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 (e.g., explosives). Direct injury or mortality from sound levels produced from the type of sonar the Action Proponents use has not been documented in fishes (Popper et al. 2007; Kane et al. 2010; Halvorsen et al. 2012; Popper et al. 2013; Popper et al. 2014a). For the same reasons (i.e., low peak pressures, slow rise times), we would not anticipate HCTT sonar sources would result in injury or mortality of IFKW invertebrate prey (e.g., cephalopods).

Based on best available information regarding the hearing capabilities of almost all IFKW prey species, these animals would not likely be able to detect most mid- and high-frequency sonar sources used by the Action Proponents. These are the most commonly used sources in the HRC and within IFKW designated critical habitat. False killer whale prey would be more likely to detect low-frequency sources, which are used far less frequently in the HRC (and designated critical habitat). False killer whale prey that are able to detect low-frequency sonar could experience brief periods of masking, or exhibit brief behavioral reactions, and stress responses. Prey items located closer to the sonar sound source would likely experience more significant responses, whereas animals located further away from the source are less likely to react to the sound levels. However, because the Action Proponent sonar is moving, and false killer whale prey are also capable of moving away from the disturbance, the overall exposure duration is expected to be brief and if masking did occur, it would not occur for a significant amount of time and not prevent the animals from detecting biologically relevant cues at meaningful levels. Additionally, any physiological stress responses or behavioral reactions are expected to be temporary, lasting only a few seconds or minutes during sonar pings. For these reasons, no long-term consequences for any exposed prey items are expected. Because of this, false killer whale prey items exposed to HCTT sonar would still be available in the environment for consumption by false killer whales following exposure. For this reason, the effects of HCTT sonar and other transducer use on the prey resource characteristic of designated critical habitat for MHI IFKWs are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated).

Sound Levels

In this section, we evaluate if HCTT sonar and other transducer use may result in sound levels within MHI IFKW designated critical habitat that would significantly impair the value of the habitat for IFKW use or occupancy. As presented in Section 6.1.1, exposure of marine mammals to sonar and other transducers could result in behavioral responses and sound-induced hearing loss (e.g., TTS), as well as other possible adverse effects (e.g., stress). When marine mammals experience these effects, their ability to carry out important life functions (e.g., foraging, navigating, or communicating) may be impaired for some period of time. Therefore, when HCTT sonar is introduced into designated critical habitat for MHI IFKWs, this has the potential to degrade the value of that habitat for important life functions. In other words, introduction of sonar into designated critical habitat may cause disruptions in use or occupancy of critical habitat, and thus diminish the value of critical habitat for the conservation of MHI IFKWs.

As discussed below, our analysis focuses on the extent of this diminishment based on a number of factors. We consider general characteristics of sound produced by sonar and other transducers, available information on HCTT sonar use within designated critical habitat (e.g., frequency and duration of use, spatial extent of sound propagation), potential and likely effects to IFKW habitat use or occupancy from exposure to sonar, and the NAEMO quantitative analysis estimating the number of instances in which sonar use in the HRC could be expected to result in disruptions in the way IFKWs use or occupy critical habitat (i.e., instances of harassment). The focus of the sound level characteristic is on the quality of the habitat for IFKW use and occupancy. As noted in the final rule to designate critical habitat, "*the mere presence of noise, or even noise which might cause harassment of the species, does not necessarily result in adverse modification*" (83Fed. Reg. 35062). In this analysis, we use the number of instances of harassment (i.e., significant disruptions of normal behavior patterns) as evidence that sonar and other transducers impairs use or occupancy because harassment within critical habitat is evidence of degraded habitat for use and occupancy. When evaluating whether sonar has decreased the value of critical habitat for conservation, we also consider timing, frequency, and duration of the sound and its resulting impacts on the population, as well as the spatial extent of impacts when they occur relative to the area designated as critical habitat.

Exposure Analysis

The Action Proponents have proposed activities using a variety of sonar sources of varying frequencies in the HRC, some of which will occur within MHI IFKW designated critical habitat. Mid-frequency (1 – 10 kHz) and high-frequency (10-100 kHz) sonar sources overlap the hearing and vocalization frequency ranges of IFKWs. The Action Proponents were unable to provide information on the specific quantities of anticipated sonar use for the duration of the 7-year MMPA rule and the reasonably foreseeable future within MHI IFKW designated critical habitat versus other portions of the action area given annual variation in individual unit training objectives, overall readiness and deployment cycles, and emergent variations in testing requirements. Similarly, the Action Proponents were unable to provide specific information on the quantities of anticipated sonar that is likely to propagate into critical habitat from sources

operated outside of critical habitat. Below we present information provided by the Action Proponents on previous sonar use within the HRC and where available, sonar use specifically expected to occur within MHI IFKW designated critical habitat in the future.

Approximately 22,600 sonar bin-hours and 13,000 sonar bin-counts are proposed in the HRC in a maximum year of activity in areas that completely or partially overlap the critical habitat (Navy 2025l). These quantities do not include sonar use proposed in areas that were excluded from the designation of MHI IFKW critical habitat and areas subject to the Joint Base Pearl Harbor-Hickam Integrated Natural Resource Management Plan (see Figure 32 for map of excluded areas). A large percentage of HRC designated subareas and the vast majority of seaspace where sonar use could potentially occur lie outside of designated critical habitat (Figure 77). Since 94% of the HRC does not overlap the critical habitat, it is likely that a large portion of the proposed sonar hours will occur outside of critical habitat (Navy 2025l).

Within the HRC, the majority of sonars used would be mid and high-frequency. Although any bin category could be used in critical habitat, a Navy review of classified data for typical mid-frequency active sonar sources (i.e., MF1, MF4, MF5) from 2012-2017 demonstrated that most use was outside of critical habitat (Navy 2025l). Large scale training events (e.g., RIMPAC, USWEX, etc.) using the more powerful hull-mounted mid-frequency active sonars would generally occur outside of critical habitat (Navy 2025l). High-frequency sonars associated with mine warfare activities would be more common in or near MHI IFKW critical habitat due to the shallow water requirements for these activities (Navy 2025l). High-frequency military sonars are generally less powerful than mid-frequency sonars, have shorter propagation ranges due to greater signal attenuation in the ocean, and are often used in directional sources rather than omni-directional sources (Navy 2025l). While high-frequency sonar can contribute to a reduction in communication space and detection space for IFKW foraging, the affected area would be both temporally and spatially limited compared to mid-frequency sources.

A thorough discussion on the potential effects of sonar on marine mammals, including IFKWs, is included in Section 6.1.1. This information is pertinent to evaluating how a HCTT activity using sonar may affect MHI IFKW critical habitat because the characteristic of the PBF under consideration is how sound levels may impact the animals themselves within the habitat. That is, the characteristic is defined as "…*sound levels that would not significantly impair false killer whales' use or occupancy*" of the habitat. As presented in Section 6.1.1, exposure of IFKWs to sonar and other transducers could result in behavioral responses and sound-induced hearing loss (e.g., TTS), as well as other possible adverse effects (e.g., stress). When IFKWs experience these effects, their ability to carry out important life functions (e.g., foraging, navigating, or communicating) is likely to be impaired for some period of time. Therefore, when HCTT sonar is introduced into designated critical habitat for MHI IFKWs, this has the potential to degrade the value of that habitat for important life functions (i.e. diminish the value of critical habitat for conservation of the species). An understanding of how MHI IFKWs react to sound created by sonar in critical habitat is essential to applying the sound element of the PBF. Meaningful conclusions cannot be drawn without understanding behavioral response as it is the most important indicator of whether use or occupancy of critical habitat is impaired by sonar use. Our

assumption is that if use or occupancy is impaired, then the value of critical habitat for the conservation of the species is diminished.

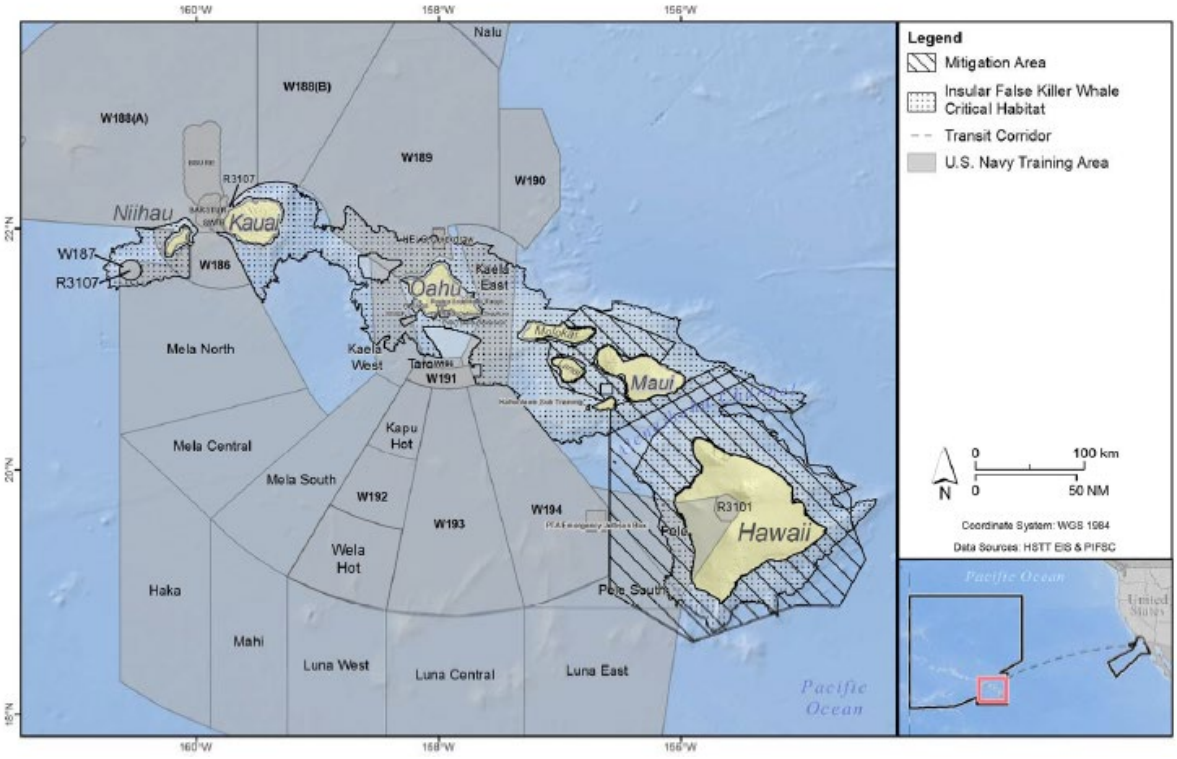


Figure 77. Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat and Hawaii Range Complex Subareas Used Most Often for Anti-submarine Warfare Training and Testing Activities (NMFS 2018b)

As documented previously, sonar exposures are more likely to be consequential to the animal if they are for long durations or occur repeatedly (e.g., Southall et al. 2007; Southall et al. 2016). Similarly, sonar use is more likely to result in significant impairment to the use or occupancy of the habitat at a population level if the habitat is subjected to long duration or repeated exposures. Sounds from sonar used by the Action Proponents recur in the action area many times annually and into the reasonably foreseeable future, but are also transient, and do not persist in the marine environment at times when the sonar source is not being operated. In general, sonar use in the action area, within the HRC, and within MHI IFKW designated critical habitat is temporary and transitory, and the habitat exposed to sonar would be expected to very quickly return to its prior state when sonar is not being transmitted.

HCTT sonar use recurs multiple times annually within the HRC and within critical habitat and is anticipated to continue into the reasonably foreseeable future. The final rule designating critical habitat explains: "*scientific information also indicates that the introduction of a permanent or chronic noise source can degrade the value of habitat by interfering with the sound-reliant animals ability to gain benefits from that habitat, impeding reproduction, foraging, or communication (i.e., altering the conservation value of the habitat)…chronic exposure to noise*

as well as persistent noise may impede the population's ability to use the habitat for foraging, navigating, and communicating, and may deter MHI IFKWs from using the habitat entirely." Thus, our analysis considers how the introduction of noise caused by HCTT sonar and other transducers may or may not impede the population's use or occupancy of designated critical habitat for important biological functions to determine whether the introduction of noise alters the conservation value of the habitat. Our analysis of individual animal response to HCTT sonar use can be used to indicate whether behavioral disturbances could significantly impair use or occupancy of the habitat at a population level.

While a majority of HCTT sonar use within the action area and within the HRC is anticipated to occur outside of MHI IFKW designated critical habitat, sonar and other transducers will be used in this area and as described previously, such activities have the potential to result in impacts to the value of such habitats for IFKW use or occupancy. Additionally, in some cases, sonar use outside of critical habitat is likely to result in sound propagating into critical habitat, potentially resulting in impacts to the value of the habitat for IFKW use or occupancy. Table 46 (Section 6.1.1) presents information on the expected number of instances of harassment (e.g., behavioral disruptions) of MHI IFKWs annually from sonar and other transducers. Of the estimated 105 annual exposures resulting in behavioral harassment, about 30% (31) are expected to occur within IFKW critical habitat. Of the estimated 64 annual exposures resulting in TTS, only one (less than 2%) is expected to occur within IFKW critical habitat. This represents the best available estimate of the number of instances in which disruptions of IFKW use (e.g., disruption of foraging, navigating, or communicating) or occupancy of critical habitat are likely to occur annually from Navy sonar and other transducers. This estimate accounts for sonar operated within critical habitat, but also sonar operated in relatively close proximity that may propagate into the area. Based on the estimated abundance of MHI IFKWs (i.e., 138 individuals) that spend all or a portion of their time in critical habitat, and the number of instances of behavioral disruption expected (i.e., estimates based on NAEMO) to occur in critical habitat, an average of 0.23 disruptions of critical habitat use annually per animal is expected.[29]

During an activity involving sonar, the sonar is typically only actively transmitted for a small part of the time. When sonar is being used it is duty-cycled such that the sound is only emitted for a short period of time, and then stopped, usually for a much longer period of time, in order for any return echoes to be received and interpreted. For example, 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 2025l). By contrast with continuous or high-duty (e.g., MF1C) active sonar sources, which have duty cycles ranging from 80-100%, sonar is actively

[29] Note that NMFS recognizes the calculation of the number of disruptions per animal is based on Navy modeling and is a rough approximation of what will occur during HCTT training and testing activities in IFKW critical habitat. Some individuals could experience a few more or less disruptions annually than what is estimated here. However, due to the limitations on acoustic exposure modeling capabilities, we are unable to identify which individuals from the 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 critical habitat will be exposed to and affected by HCTT sonar annually. This estimate is presented to indicate the relative magnitude of likely exposures on an annual basis for animals within the population.

transmitted for the majority of time that the sound sources is being used. As such, continuous and high-duty sonar sources have more potential to mask vocalizations, particularly for mid or high-frequency cetaceans (e.g., false killer whales), given the overlap in vocalization frequency with these sonar systems.

Table 50 shows the proportion of overall sonar use in the HRC with high duty or continuous duty cycle sound sources. Approximately 2% of sonar bin-counts and a small portion of bin-hours employ high-to-continuous duty cycle sources, particularly in mid- and high frequencies that are relevant to false killer whale communication, foraging, and hearing. As noted above, the Action Proponents could not provide us with more detailed information regarding the level of activity (e.g., number of events, proportion of sonar use, and hours of sonar use) involving continuous or high-duty active sonar within MHI IFKW critical habitat. Instead, we use information on the distribution of IFKW acoustic impacts within critical habitat, by activity and sonar bin category (Table 51), to help understand the relative amount of high duty and continuous active sonar source use that would be expected within critical habitat. A variety of activities involving multiple sonar source types (i.e., bins) contribute to the predicted impacts to MHI IFKWs within critical habitat. About one-half of the estimated impacts are from activities that include at least one high duty or continuous active sonar source in the mid to high-frequency range that overlaps with IFKW vocalizations and hearing.

In addition to the analysis of individual animal responses to HCTT sonar use, our analysis in this consultation also considers the temporal and spatial extent of sonar exposure to designated critical habitat. When considering potential impacts to MHI IFKW designated critical habitat, we focus on mid-frequency and high-frequency sonars as those are the sources within the primary hearing range of the species (i.e., 150 Hz to 160 kHz). The spatial extent of sonar exposure to habitat during each hour of sonar use would vary depending on source level and propagation of the sound within the water column. Ranges to effects to marine mammals from exposure to sonar and other transducers are provided in the *Navy's Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area* (Navy 2025f), and are incorporated into this opinion by reference (Section 2.5.1, pp. 2-91 to 2-139). When considering the spatial extent of exposure to designated critical habitat, the cutoff distances described in Section 6.1.1 must also be considered and applied. For odontocetes, the modeled cutoff was 15 km with a received level of 168 dB RMS; i.e., animals that are beyond 15 km from the source and below 168 dB re: 1 µPa RMS are unlikely to incur significant behavioral reactions. For both mysticetes and odontocetes, regardless of received level, the model considered animals beyond 100 km from the source as unlikely to incur significant behavioral responses. It is important to emphasize that these distances represent the maximum distance in which impacts to use or occupancy would be expected. At these larger distances, the likelihood of a significant behavioral disruption (and therefore impacts to use or occupancy) is relatively low for many sonars source classes. There would be an increased likelihood of significant behavioral disruptions (and therefore impacts to use or occupancy) at closer distances. Designated critical habitat for MHI IFKWs encompasses approximately 45,859 km^2. Based on the range to effects and the size of designated critical habitat, only a very small percentage of

critical habitat would be exposed to HCTT sonar that could result in impacts to use or occupancy during each hour of use.

Table 50. Proportion of Overall Sonar Use in the Hawaii Range Complex with High to Continuous Duty Cycles (Navy 2025l)

Source Class Category[1]	*Description*	*Duty Cycle*	*Percent*
Broadband Sources[2]			
LF	<205 dB	High	-
LF to HF		Continuous	2%
		High	3%
LF to MF		High	0%
MF to HF		High	18%
Low-Frequency Acoustic Sources			
LFL	160 dB to 185 dB	High	0%
LFM	185 dB to 205 dB	Continuous	0%
		High	12%
LFH	>205 dB	Continuous	0%
		High	2%
Mid-Frequency Acoustic Sources Other Than Hull-Mounted			
MFL	160 dB to 185 dB	High	2%
MFM	185 dB to 205 dB	Continuous	0%
		High	2%
MFH	>205 dB	Continuous	0%
		High	0%
Hull-Mounted Surface Ship Sonar			
MF1C	Hull-mounted surface ship sonar (previously MF11) with duty cycle >80%	High	2%
High-Frequency Acoustic Sources			
HFL	160 dB to 185 dB	High	0%
HFM	185 dB to 205 dB	Continuous	0%
		High	2%
HFH	>205 dB	Continuous	0%
		High	0%
Very High-Frequency Acoustic Sources			
VHFL	160 dB to 185 dB	High	-
VHFM	185 dB to 205 dB	High	-
VHFH	>205 dB	Continuous	0%
		High	0%

(-) means no hours or counts are proposed in this category in these areas.
[1] Bin MF1 and MF1K (hull-mounted sonar) are not included because they have a low duty cycle.
[2] Broadband sources have a range of duty cycles. For this analysis, they are all assumed to be high-to-continuous, which is an over-estimate.

Table 51. Estimated Effects to MHI IFKWs within Critical Habitat over a Maximum Year of Proposed Activities by Activity Type and Sonar Bin (Navy 2025c)

Activity (training or testing)	**Bins**	**Potential High or Continuous Duty Cycle Sonar Yes / No ?**	**Event Duration**	**Number of Behavioral Response Impacts**
Submarine Navigation (training)	MFH, HFH	No	hours	7
Surface Ship Object Detection-HCR (training)	MF1K	No	hours	5
Anti-Submarine Warfare Tracking Test (Rotary Wing)-HRC (testing)	MFM, MFH	Yes	hours	3
Multi-Domain Unmanned Autonomous Systems (training)	VHFH, Broadband (MF to HF); E5, E7	Yes	days	3
Medium Coordinated Anti-Submarine Warfare (training)	MF1, MF1C, MFM, MFH, HFH, Broadband (LF to HF), Broadband (MF to HF)	Yes	2-3 days	2
Surface Ship Object Detection-Pearl Harbor (training)	MF1K	No	hours	2
Vehicle Testing (testing)	LFH, MFL, MFH, HFL, HFM, VHFH	Yes	days	2
Anti-Submarine Warfare Torpedo Test (Aircraft)-HRC (testing)	MFM, MFH, HFH	Yes	days	1
Long Range Acoustic Communications (testing)	LFM	Yes	days	1
Mine Countermeasures - Ship Sonar-Pearl Harbor (training)	MF1K, HFH	No	hours	1
Submarine Sea Trials - Weapons System Testing (testing)	MFL, MFH, HFM, HFH, Broadband (LF to HF)	Yes	days	1
Multiple Activities	Various Sources	Unknown		4*
			Total:	32

* Includes one TTS impact

Response Analysis

During exposure in designated critical habitat, affected IFKWs may be utilizing the habitat to engage in any number of activities including, but not limited to, foraging, navigating, communicating, or resting. If IFKWs exhibited a behavioral response to HCTT sonar that affected how the animals were utilizing the habitat, these activities would be disrupted and it may pose some energetic cost. However, as noted previously in Section 6.1.1, most responses to HCTT sonar (both in and outside of critical habitat) are anticipated to be short term and instances of hearing impairment (i.e., TTS) are expected to be mild or moderate. Based on best available

information that indicates marine mammals typically resume normal behavior quickly after the cessation of sonar exposure, we anticipate that exposed animals will be able to return to normal behavioral patterns after this short duration exposure ceases. For example, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to 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). Baird et al. (2013; 2014; 2017a) tagged four shallow-diving odontocete species (including false killer whales) in the HRC off the Pacific Missile Range Facility before Navy training events using sonar. Consistent with most other studies looking at the response of marine mammals to sonar (e.g., Melcon et al. 2012; Goldbogen et al. 2013), none of the tagged animals demonstrated a large-scale or long-term avoidance response to the sonar as they moved on or near the training range where sonar was being used.

Additionally, based on best available information regarding exposure of IFKWs and other marine mammal species to HCTT sonar, these short term responses are not anticipated to lead to long duration (e.g., more than one day) abandonment of any particular area within critical habitat, or of critical habitat as a whole. As noted in Section 6.1.1, when marine mammals have been observed to leave a certain area following acoustic exposure, they are likely to return after the period of exposure ends (e.g., Tyack et al. 2011b; Baird et al. 2016). Even in intensively used sonar training areas, such as those locations considered in this consultation in Southern California, photo identification results have indicated long-term residency of odontocetes (i.e., Cuvier's beaked whales) considered particularly sensitive to disturbance from acoustic exposure (Falcone et al. 2009; Falcone and Schorr 2014). IFKWs are not known to be as sensitive to acoustic disturbance as beaked whales. In contrast, evidence suggests MHI IFKWs are fairly tolerant of disturbance from military sonar, perhaps because the animals in the population have been exposed to sonar multiple times on previous occasions (Baird et al. 2017a).

For another study (Baird et al. 2016), tags were deployed on seven false killer whales prior to a training event using sonar in the HRC. Only one individual's movements in space and time lent itself to detailed analysis of exposure and response to mid-frequency active sonar. The false killer whale passed through the training event twice, transiting away from an area of relatively low exposure (estimated received level of mean = 90.9 dB re: 1 µPa rms) and toward the training event, receiving an estimated median received level of 156 dB re: 1 µPa with a maximum estimated received level of 188 dB re: 1 µPa (Baird et al. 2016). The individual then moved away from the areas of sonar for several hours, then returned to this area of exposure to an estimated maximum mean received level of 150.8 dB, and then moved to an area of lower received levels (Baird et al. 2016). This exposure-response scenario is consistent with the discussion above in that short term responses were observed, but animals returned to the area following exposure. Also important to consider is that the Navy has been conducting training and testing activities in the HRC, and specifically within the area designated as critical habitat, for decades. Despite this, monitoring (e.g., Baird et al. 2013; Baird et al. 2014; Baird et al. 2017a) indicates that IFKWs continue to utilize these habitats.

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 of marine mammals to HCTT sonar. With respect to the characteristic of the PBF related to sound levels, the duration and frequency of the sound source being utilized in or near designated critical habitat is important in evaluating the extent of potential impacts to use and occupancy (83Fed. Reg. 35062). Most of the activities the modeling indicates are likely to result in significant disruptions of behavior patterns in designated critical habitat last less than one day. While these activities are anticipated to result in significant disruptions of behavior of the animals in designated critical habitat, and therefore disruptions of IFKW's use of the habitat, the short duration of these events limits the potential for these activities to result in long duration exposures to the animals or specific areas of designated critical habitat and, consistent with literature described previously on the response of marine mammals to sonar, we anticipate that exposed animals will be able to return to normal behavior patterns shortly after the exposure is over (minutes to hours; e.g., Goldbogen et al. 2013; Silve et al. 2016).

Some activities occurring in the action area and resulting in impacts to IFKWs will last for longer durations. For example, RIMPAC is expected to result in hundreds of hours of sonar activity involving multiple platforms (i.e., surface vessels, submarines, and aircraft) utilizing sonar. These exercises range in duration from two to over ten days, and therefore have the potential to result in sustained and/or repeat exposure. However, while MTEs may have a longer duration, they are not concentrated in small geographic areas over that time period. MTEs use thousands to tens of thousands of square miles of ocean space during the course of the event. As noted previously, while some portion of such exercises could occur in critical habitat, a majority would occur in more offshore training subareas of the HRC. This is particularly true for portions of the exercises that utilize the Navy's more powerful mid and high-frequency hull mounted sonars that have higher likelihoods of resulting in significant behavioral disruptions. There is no HCTT 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). Further, as described in the final biological report supporting the critical habitat designation (NMFS 2018c), MHI IFKWs circumnavigate the Main Hawaiian Islands and quickly move throughout their range (Baird et al. 2008; Baird et al. 2012; NMFS 2018c). One individual moved from Hawaii to Maui to Oahu to Molokai, covering at least 449 km over a 96 hour period (Baird et al. 2010; Oleson et al. 2010; NMFS 2018c). Therefore, there is a low likelihood that IFKWs and HCTT activities will co-occur for extended periods of time or repetitively over the duration of an activity in critical habitat due to the following: (1) the large area of habitat available to and utilized by MHI IFKWs (NMFS 2018c), (2) the relatively small area that will be affected by sonar at any given period of time, (3) that most HCTT activities resulting in significant disruptions of habitat use last less than one day, and (4) that long duration (i.e., more than one day) HCTT 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), largely outside of designated critical habitat. Because of

this, and the short duration of responses anticipated, we expect that effects to IFKW use or occupancy of designated critical habitat from HCTT sonar will be temporary when they occur.

The potential for masking is another important consideration when evaluating effects of anthropogenic noise on habitat use of marine mammals, including IFKWs. Masking was also discussed in Section 6.1.1 of this opinion. Some limited masking could occur due to the Action Proponent'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 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. Because most HCTT sonar sources 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 2013a). 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, but these effects would only happen close to the source. As discussed above in our exposure analysis, the quantitative acoustic effects analysis predicts impacts to IFKWs within critical habitat from a variety of activities involving multiple sonar bins that could potentially include high duty or continuous active sonar sources (Table 51). These sonars transmit more frequently (greater than 80% 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, IFKWs close to the sonar source could experience masking on a longer time scale than those exposed to traditional (i.e., low duty cycle) sonars. However, even with high duty and continuous sonar sources, the duration of masking would be limited since both the sonar source and the whale are moving within the environment, and we do not anticipate whales would remain in close proximity to a source for an extended period of time. Our exposure analysis above predicted a relatively small number of annual impacts (i.e., 31 behavioral, 1 TTS) to MHI IFKWs within critical habitat. Based on information provided by the Action Proponents, we estimate that less than half of these impacts would likely be from exposure to high duty cycle or continuous sonar. Although the effects of masking are expected to be limited in space and time, the use and occupancy of critical habitat may still be temporarily impaired if MHI IFKWs are actively avoiding areas where high duty or continuous active sonar is being transmitted into critical habitat. While most sonar sources used by the Action Proponents operate over a narrow band of frequencies which only overlap a small portion of the full range of MHI IFKW hearing, some activities that will overlap with critical habitat, including some involving high duty cycle and continuous sonar, also use broadband sonar sources (e.g., anti-submarine warfare training). Based on the factors discussed above, we anticipate that HCTT sonar will result in a limited amount of masking that could impair IFKW habitat use or occupancy. Masking as a result of mid-frequency (other than hull-mounted) and broadband sonar sources within the 4-10 kHz

range may impact MHI IFKW's ability to communicate and socialize. Mid-frequency sonar sources are not likely to affect their ability to find prey through echolocation since this species echolocates at higher frequency levels (i.e., typically > 17 kHz; Madsen et al. 2004; Thode et al. 2016). False killer whale echolocation clicks could be masked by sound from high-frequency or broadband, high-duty cycle sonar sources (see Table 51 for HCTT activities with high-frequency, high-duty cycle sonar sources with NAEMO predicted impacts to MHI IFKWs). However, as discussed in Section 6.1.1, any masking effects from high-frequency sonar, including high-duty cycle sources, would likely be minimal given the small zone of effect anticipated.

6.2 Sea Turtles

This section addresses the effects of acoustic (i.e., from sonar and other transducers, and air guns), explosive, and vessel strike stressors on ESA-listed sea turtles from the proposed HCTT activities. We start with an overview of the methodology for the Navy's quantitative analysis of the effects of these stressors on sea turtles.

6.2.1 Acoustic and Explosive Effects Analysis

To evaluate the impacts from acoustic stressors associated with proposed training and testing activities, the Navy performed a quantitative analysis to estimate the number of instances (i.e., exposures) that could affect ESA-listed sea turtles and the magnitude of that effect (e.g., injury, hearing loss, behavioral response). Similar to that described above for marine mammals (Section 6.1.1), the quantitative analysis utilizes NAEMO, and takes into account criteria and thresholds used to predict impacts in conjunction with spatial densities of sea turtles within the action area. One important difference from the quantitative analysis for marine mammals is that avoidance is not factored into NAEMO impacts estimates for sea turtles (i.e., sea turtle are assumed not to have the ability to avoid acoustic impacts by swimming away from a sound source).

We consider the modeling conclusions from the Navy's analysis to represent the best available data on exposure of ESA-listed sea turtles to acoustic and explosive stressors from the proposed action. We also find that the estimates of take resulting from this analysis are reasonably certain to occur. For more details on this approach see Section 6.1.1 above, or the Navy's technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training* (Navy 2024h).

Density Estimates

The density data used in the NAEMO quantitative impact analysis for each ESA-listed sea turtle species present in the HCTT action area is described in the Navy's density technical report (Navy 2025n). For most sea turtle species, abundance is estimated using line-transect surveys, or mark-recapture studies. Sea turtle abundance estimates have also been derived from other in-water survey data in selected areas that rely on published abundance and distribution data from nesting sites. As discussed above for marine mammals, the Navy compiled the best available

information on sea turtle species' densities in the action area into the NMSDD, which was used in NAEMO for estimating impacts to sea turtles from acoustic and explosive stressors.

The sea turtle density data used by the Navy to quantitatively assess impacts is primarily from NMFS's aerial surveys with supplemental data from shipboard surveys from NMFS and others. NAEMO outputs only represent estimates for larger sea turtles, since smaller turtles (i.e., including most less than about than 30 cm diameter, and some proportion less than 40 cm), are likely missed from aerial survey. For these reasons, neither age class nor size are explicitly accounted for in the sea turtle density data, although the larger size makes sightability and identification of age and species easier.

A guild was used previously to estimate sea turtle densities in Hawaii due to limited species-specific abundance and distribution data. The guild was essentially a proxy for all five sea turtle species potentially occurring in Hawaiian waters (Navy 2025n). Sea turtle densities were estimated in both nearshore (< 100 m depth contour) and offshore (> 100 m depth contour) regions of the action area. To determine NAEMO takes by species the predicted exposures were distributed among the five sea turtle species in the action area in proportion to the number of fisheries interactions reported by the longline fishery off Hawaii (Navy 2025n). The assumption was that the number of fishery interactions correlated positively with species occurrence and likelihood of exposure to acoustic stressors. Offshore (> 100 m depths), species-specific densities were based on NMFS's data on interaction with the longline fishery from 2010-2020 (Navy 2024d). In nearshore waters of the HRC, the NAEMO exposure analysis assumed 99% of sea turtles were green sea turtles, 0.9% were hawksbill sea turtles, and 0.1% were olive ridley sea turtles (Navy 2024d).

Similar to Phase III, green sea turtle density estimates off of California for Phase IV were limited to San Diego Bay. Densities for loggerhead and leatherback sea turtles within the California portion of the action area were added in Phase IV; densities were not estimated for either of these species in Phase III (Navy 2024d). The loggerhead sea turtle density was based on a NMFS survey in 2015 (Eguchi et al. 2018) during a strong El Nino when there were anomalously warm sea surface temperatures in the Southern California Bight, and should only apply during similar conditions. The loggerhead density used for NAEMO impact estimates was limited to September and October, when warmest sea surface temperatures typically occur off Southern California (Navy 2024d). Leatherback sea turtle densities were derived primarily from Benson et al. (2020) and in collaboration with NMFS scientists.

Criteria and Thresholds to Predict Impacts

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 sea turtle criteria and thresholds developed and applied for each sound source associated with the proposed HCTT activities. A detailed description of the sea turtle criteria and threshold development is included

in the Navy's technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV)* (Navy 2025a).

Non-auditory Injury and Mortality

The Navy's quantitative analysis assessed the potential for HCTT activities involving in-water explosives to physically injure or kill ESA-listed sea turtles. The criteria and thresholds for non-auditory injury and mortality in sea turtles are the same as those described for marine mammals in Section 6.1.1.

Auditory Impacts

Although no studies have directly measured underwater TTS in ESA-listed sea turtles, two recent studies examined underwater TTS in freshwater turtles. Salas et al. (2023b) exposed red-eared sliders (*Trachemys scripta elegans*) to sound exposure levels between 155-193 dB re: 1 lPa^2 s, and auditory sensitivity was measured at 400 Hz using auditory evoked potential methods. The mean predicted TTS onset was 160 dB re: 1 lPa^2 s and there were significant positive relationships between shift magnitude and exposure duration, amplitude, and SEL. The SELs at which TTS was observed in these freshwater turtle studies are considerably lower than the predicted threshold of 200 dB re: 1 lPa^2 s for ESA-listed sea turtles (Salas et al. 2023b), which was based on surrogate TTS data from freshwater fishes (U.S. Department of the Navy 2017). In a similar study using Eastern painted turtles, Salas et al. (2023b) reported similar results, with TTS onset occurring at much lower SELs than previously predicted.

Although sea turtles appear to be slightly less sensitive to sound (e.g., higher thresholds at some frequencies) than freshwater turtles (Piniak et al. 2016), Salas et al. (2023a) conclude that their findings "warrant caution and awareness" that ESA-listed sea turtles may incur hearing loss at lower sound levels than previously thought.

New criteria and thresholds for the onset of sea turtle TTS and AINJ were developed for Phase IV based on the best available data (Table 53).

Table 52. Comparison of Phase III and Phase IV Sea Turtle TTS and AINJ Onset Levels for Non-impulsive (sonar) and Impulsive (explosives, airguns) Sound Sources (weighted SEL onset units are dB re: 1 μPa^2s; impulsive peak SPL onsets units are dB re: 1 µPa)

	Phase 3		Phase 4	
Source Type and Metric	**TTS**	**AINJ (PTS^2)**	**TTS**	**AINJ (PTS)**
Non-impulsive onset SEL (weighted)[1]	200	220	178	198
Impulsive onset SEL (weighted)	189	204	169	184
Impulsive onset Peak SPL	226	232	224	230

[1]The weighted non-impulsive thresholds by themselves only indicate the TTS/AINJ threshold at the most susceptible frequency.

[2] Auditory injury (AINJ) was previously referred to as permanent threshold shift (PTS). The new terminology acknowledges that auditory injury may occur without PTS.

A new sea turtle composite audiogram was derived from existing hearing data (Figure 78). Based on the composite audiogram and TTS data from freshwater turtles, exposure functions were created for non-impulsive noise sources (Figure 79).

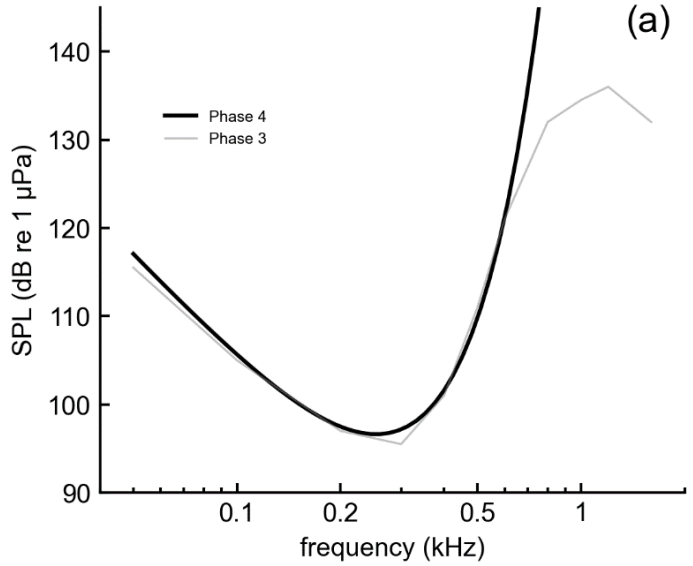


Figure 78. Comparison of Phase III (thin gray lines) and Phase IV (thick black lines) Sea Turtle Composite Audiograms (Navy 2025f)

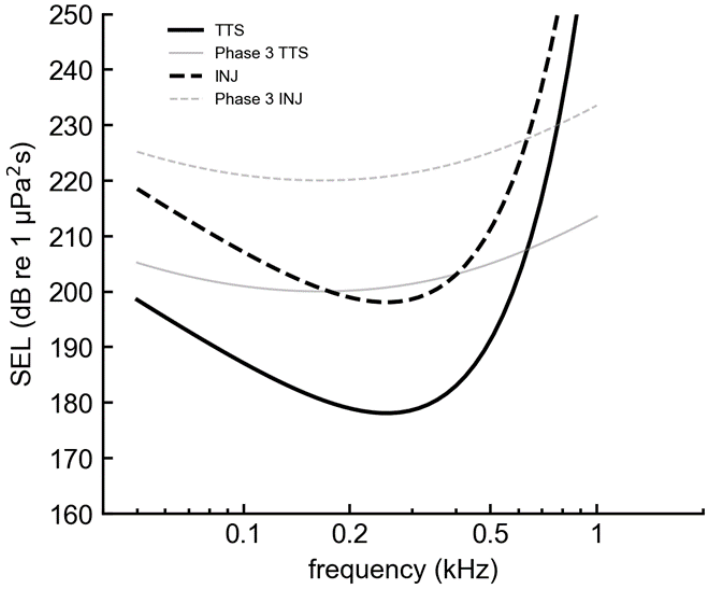


Figure 79. Comparison of Phase III (thin gray lines) and Phase IV (thick black lines) Sea Turtle Exposure Functions for Non-impulsive TTS (solid lines) and AINJ (dashed lines) (Navy 2025e)

Behavioral Response

The unweighted SPL_{rms} behavioral disturbance threshold of 175 dB for sonar (below 2 kHz) and airguns were carried over from Phase III for the NAEMO Phase IV analysis. For sea turtle exposures to multiple explosions a threshold of 164 dB re: 1 μPa^2s SEL was used. For a single explosion the behavioral response threshold is set to the impulsive TTS onset threshold of 169 dB re: 1 μPa^2s SEL.

Ranges to Effects

Ranges to effects to sea turtles from exposure to sonar and other transducers, airguns, and explosives are provided in the Navy's *Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area* (Navy 2025a), and are incorporated into this opinion by reference (Section 3.4, pp. 3-31 through 3-3).

Exposure Analysis

Section 6.2.1 presented information on the criteria and thresholds used to estimate impacts to ESA-listed sea turtles from acoustic and explosive stressors. Additional information on these criteria is described in the Navy's criteria and thresholds technical report (Navy 2025c). This section presents information on the estimated number of exposures of ESA-listed sea turtles to acoustic (i.e., sonar and other transducers, and air guns) and explosive stressors that are expected to result in a significant behavioral response, TTS, AINJ, non-auditory injury, or mortality. Exposure estimates were produced using the NAEMO approach as described in Section 6.1.1 above, and in the Navy's technical report (Navy 2024h). For each sea turtle species, a multi-sectioned table quantifies impacts as follows:

- The first section shows the number of instances of each effect type that could occur due to each substressor (sonar, air guns, or explosives) over a maximum year of activity. Impacts are shown by type of activity (i.e., Coast Guard training; other Action Proponent training; and testing).
- The second section shows the percent of total impacts that would occur within seasons (i.e., warm and cold) and the following general geographic areas: SOCAL RC, PMSR, NOCAL RC, HRC, and the high seas (transit lanes between the California and Hawaii portions of the action area).
- The third section shows which activities are most impactful to the species. Activities that cause 5% or more of total impacts to a species are shown here.
- The fourth section shows impacts in critical habitats where they are designated or proposed for ESA-listed sea turtles.

As discussed above, NAEMO outputs only represent estimates for larger sea turtles since density estimates are largely derived from aerial surveys, which typically can only detect the larger sea turtles (Epperly et al. 1995). Turtles smaller than about 40 cm are likely being missed by aerial surveys (regardless of altitude) to an unknown degree and represent a sizable proportion of the

population of all sea turtle species (DiMatteo et al. 2024). As such, NAEMO estimates, based on these sea turtle density models, underestimate impacts to an unknown extent.

Since data are not available to estimate the proportion of smaller sea turtle size classes that are missed, we cannot quantify impacts to smaller sea turtles at this time. Since we cannot quantify impacts to smaller sea turtles (i.e., less than ~ 40 cm) from sonar and explosives, we instead use a surrogate 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., significant behavioral response, TTS, AINJ, non-auditory injury, and mortality) from sonar and explosives in those areas occupied by sea turtles less than 40 cm, and 2) sonar and explosive activity levels, which is the total number of hours or count (by sonar or explosive bin type) that would be used in the action area as part of the proposed action.

During the early life histories of sea turtles, hatchlings and juveniles spend a majority of time passively floating in prevailing ocean currents and inhabiting floating vegetation mats. The Action Proponents have proposed activity-based mitigation measures (Section 2.4.2) aimed at minimizing impacts on floating vegetation used by hatchling and juvenile sea turtles. Although the density data does not quantitatively allow the separation of sea turtle size classes, the effects to small sea turtles (less than 30 cm diameter) is somewhat minimized through mitigation.

Green Sea Turtle – East Pacific DPS

Estimated exposures to East Pacific DPS green sea turtles from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 53. The large majority of impacts anticipated are TTS resulting from Navy sonar used for Acoustic and Oceanographic Research.

Table 53. Estimated Effects to East Pacific DPS Green Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	-	(1)	-	-	-
Explosive	Navy Training	9	8	1	(1)	0
Explosive	Navy Testing	2	7	1	0	0
Sonar	Navy Testing	42	622	7	-	-
Maximum Annual Total		**53**	**638**	**9**	**1**	**0**

	Percent of Total Impacts
Season	**SOCAL RC**
Warm	38%
Cold	62%

Activities with 5% or More of Total Impacts	Category	Percent of Total Impacts
Acoustic and Oceanographic Research (ONR)	Navy Testing	91%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury
For BEH, TTS, AINJ annual estimated impacts: Zero (0) impacts indicate total less than 0.5 and a dash (-) is a true zero.
The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Green Sea Turtle Central North Pacific DPS

Estimated exposures to Central North Pacific DPS green sea turtles from the use of sonar, air guns, and explosives as part of the proposed action are shown in Table 54. The large majority (83%) of impacts are from Navy explosives used for Obstacle Loading training. Nearly 80% of the predicted impacts will occur within shallow water areas of the HRC that have been proposed

by NMFS as critical habitat for the Central North Pacific DPS green sea turtle. While most exposures will result in either a significant behavioral response or auditory effects (i.e., TTS or AINJ), a small number non-auditory injuries (up to three per year) and mortalities (up to one per year) are also predicted.

Table 54. Estimated Effects to Central North Pacific DPS Green Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Air gun	Navy Testing	-	(1)	-	-	-
Explosive	Navy Training	2,019	1,061	41	2	1
Explosive	Navy Testing	32	58	4	(1)	0
Explosive	Army Training	(1)	(1)	-	-	-
Sonar	Navy Testing	16	50	-	-	-
Maximum Annual Total		**2,068**	**1,171**	**45**	**3**	**1**

	Percent of Total Effects
Season	**HRC**
Warm	53%
Cold	47%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Obstacle Loading	Navy Training	82%
Underwater Demolition Qualification and Certification	Navy Training	8%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat (Proposed)	Critical Habitat (All)	1,636	950	39	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5. The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Hawksbill Sea Turtle

Estimated exposures to hawksbill sea turtles from the use of sonar and explosives as part of the proposed action are shown in Table 55. The majority (68%) of impacts are from Navy explosives used for Obstacle Loading training, followed by Navy sonar used for Acoustic and Oceanographic Research (17%).

Table 55. Estimated Effects to Hawksbill Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	18	10	(1)	-	-
Explosive	Navy Testing	0	1	0	-	-
Explosive	Army Training	-	(1)	-	-	-
Sonar	Navy Testing	1	6	-	-	-
Maximum Annual Total		**19**	**18**	**1**	**-**	**-**

	Percent of Total Effects	
Season	**HRC**	**High Seas**
Warm	56%	0%
Cold	42%	1%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Obstacle Loading	Navy Training	68%
Acoustic and Oceanographic Research (ONR)	Navy Testing	17%
Underwater Demolition Qualification and Certification	Navy Training	7%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.

Source	Category	BEH	TTS	AINJ	INJ	MORT

The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Loggerhead Sea Turtle – North Pacific Ocean DPS

Estimated exposures to North Pacific DPS loggerhead sea turtles from the use of sonar and explosives as part of the proposed action are shown in Table 56. The large majority of impacts anticipated are TTS resulting from Navy testing activities using sonar, particularly Acoustic and Oceanographic Research. Navy training and testing activities involving explosives account for the majority of injurious impacts, including an estimated six AINJ and one non-auditory injury exposures during a maximum year of HCTT activities. About two-thirds of loggerhead impacts are anticipated in the California portion of the action areas and during the warm season (i.e., summer and fall).

Table 56. Estimated Effects to North Pacific Ocean DPS Loggerhead Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	36	60	3	(1)	0
Explosive	Navy Testing	31	82	3	1	0
Explosive	USCG Training	0	0	-	-	-
Explosive	Army Training	(1)	1	-	-	-
Sonar	Navy Training	1	(1)	-	-	-
Sonar	Navy Testing	66	583	3	-	-
Maximum Annual Total		**135**	**727**	**9**	**2**	**0**

	Percent of Total Effects			
Season	**SOCAL RC**	**PMSR**	**HRC**	**High Seas**
Warm	56%	10%	14%	2%
Cold	0%	0%	16%	1%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Acoustic and Oceanographic Research (ONR)	Navy Testing	70%
Vehicle Testing	Navy Testing	6%
Mine Neutralization Explosive Ordnance Disposal	Navy Training	5%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.
The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Olive Ridley Sea Turtle

Estimated exposures to olive ridley sea turtles from the use of sonar and explosives as part of the proposed action are shown in Table 57. The large majority of impacts anticipated are TTS resulting from Navy sonar testing activities, particularly Acoustic and Oceanographic Research.

Table 57. Estimated Effects to Olive Ridley Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	2	5	(1)	0	-
Explosive	Navy Testing	(1)	2	(1)	-	-
Explosive	USCG Training	0	-	-	-	-
Explosive	Army Training	(1)	2	(1)	-	-
Sonar	Navy Testing	33	215	2	-	-
Maximum Annual Total		**37**	**224**	**5**	**0**	**-**

	Percent of Total Effects	
Season	**HRC**	**High Seas**
Warm	44%	5%
Cold	47%	4%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Acoustic and Oceanographic Research (ONR)	Navy Testing	85%
Vehicle Testing	Navy Testing	11%

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.
The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Leatherback Sea Turtle

Estimated exposures to leatherback sea turtles from the use of sonar and explosives as part of the proposed action are shown in Table 58. The large majority of impacts anticipated are TTS resulting from Navy sonar testing activities, particularly Acoustic and Oceanographic Research. A small proportion (4%) of exposures are expected to occur within leatherback critical habitat off the coast of California. Leatherback impacts are fairly evenly distributed spatially among the SOCAL RC, NOCAL RC and HRC, and seasonally between warm and cold water months.

Table 58. Estimated Effects to Leatherback Sea Turtles over a Maximum Year of Proposed HCTT Activities (Navy 2025i)

Source	Category	BEH	TTS	AINJ	INJ	MORT
Explosive	Navy Training	3	2	(1)	-	-
Explosive	Navy Testing	2	5	(1)	0	-
Explosive	Army Training	(1)	(1)	(1)	0	-
Sonar	Navy Training	0	0	-	-	-
Sonar	Navy Testing	43	376	3	-	-
Maximum Annual Total		**49**	**384**	**6**	**0**	**-**

	Percent of Total Effects			
Season	**SOCAL RC**	**NOCAL RC**	**HRC**	**High Seas**
Warm	13%	14%	15%	3%
Cold	17%	16%	17%	4%

Activities Causing 5% or More of Total Effects	Category	Percent of Total Effects
Acoustic and Oceanographic Research (ONR)	Navy Testing	88%
Vehicle Testing	Navy Testing	9%

Area Type	Area Name (Active Months)	BEH	TTS	AINJ	INJ	MORT
Critical Habitat	CA Coastal Marine Waters (All)	1	18	0	-	-

BEH = Significant Behavioral Response, TTS = Temporary Threshold Shift, AINJ = Auditory Injury, INJ = Non-Auditory Injury, MORT = Mortality
For BEH, TTS, AINJ, INJ, MORT annual effects: Dash (-) indicates a (true zero) and zero (0) indicates a rounded value less than 0.5.
The value (1) indicates up to one exposure in a maximum year, but less than one per year, on average.

Response Analysis

Given the above estimated exposures of ESA-listed sea turtles to acoustic stressors associated with the proposed action, in this section we describe the range of responses to these exposures. This includes behavioral responses, sound-induced hearing loss (i.e., TTS and AINJ), non-auditory injury, physiological stress responses, and masking. As discussed above for marine mammals (Section 6.1.1), our aim with this response analysis is to assess the potential responses that might reduce the overall fitness of individual ESA-listed sea turtles. Such responses would likely rise to the level of take under the ESA.

Non-Auditory Injury and Mortality

From our exposure analysis above, a small number of exposures to explosives are expected to result in non-auditory injuries for the following ESA-listed sea turtle species: green sea turtle – East Pacific and Central North Pacific DPSs and loggerhead North Pacific Ocean DPS. A small number of Central North Pacific DPS green sea turtle mortalities from explosives are also anticipated as a result of the proposed action.

Based on what is known about potential sea turtle impacts from explosives, we assume underwater explosives can kill, injure, and impair sea turtles exposed to detonations. Lethal injuries can 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).

Non-lethal injuries can impact individual sea turtle fitness, causing a reduction in foraging success, reproduction, or increased susceptibility to predators. Depending on the type of injury (e.g., severity, location) and other relevant factors (e.g., animal life stage, size, condition prior to exposure), a reduction in fitness from non-auditory injury could either be temporary with recoverable impacts, or result in long-term fitness consequences for some individuals. Examples of nonlethal injuries include eardrum rupture, bruising, and immobilization of severely stunned animals. Minor organ injuries and contusions can occur as a result of underwater explosions; however, some sea turtles would be expected to recover over time through normal healing processes. Stunned animals beneath the water may drown or become vulnerable to other impacts (e.g., vessel strike, predation) while they are immobilized. Still, delayed complications arising from nonlethal injuries may ultimately result in the death of the animal because of increased risks from secondary infection, predation, or disease; and a reduced foraging capacity.

Hearing Loss and Auditory Injury

Sea turtles are likely susceptible to hearing loss when exposed to high levels of sound within their limited hearing range (most sensitive from 100 to 400 Hz and limited over 1 kHz). Explosives are a broadband source (Hildebrand 2009a), so if a sea turtle experiences TTS or AINJ from explosives, a greater frequency band would be affected as compared to TTS or AINJ from narrow-band sonar sources. From our exposure analysis above, the large majority of TTS

and AINJ impacts to sea turtles are from explosives. Central North Pacific DPS green sea turtles are the species that will be most impacted by TTS and AINJ from explosives (i.e., up to 1,119 TTS and 45 AINJ per years). Air guns are also an impulsive, broadband sound source dominated by lower frequencies within the hearing range of reptiles. However, from our exposure analysis above, only a small number of green sea turtles (i.e., less than one per year from each DPS) would be exposed to air gun noise at levels that could result in an adverse effect.

Because a greater frequency band would be affected due to explosives, there is increased chance that the hearing impairment will affect frequencies utilized by sea turtles for acoustic cues, such as the sound of waves, coastline noise, the presence of a vessel or predator. However, sea turtles are not known to rely heavily on sound for life functions (Popper et al. 2014a; Nelms et al. 2016), and instead, 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). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions. Sea turtles may use acoustic cues such as waves crashing, wind, vessel and/or predator noise to perceive the environment around them. If such cues increase survivorship (e.g., aid in avoiding predators, navigation), hearing loss may have effects on individual sea turtle fitness. TTS in sea turtles is expected to only last for a few hours to days, depending on the severity. Given this short period of time, and that sea turtles are not known to rely heavily on acoustic cues, we do not anticipate that TTS exposure would have fitness impacts on individual turtles. AINJ could permanently impair a sea turtle's ability to hear environmental cues, depending on the frequency of the cue and the frequencies affected by the hearing impairment. Given this longer time frame, we anticipate that at least some sea turtles that experience AINJ may have a reduction in fitness either through some slight decrease in survivorship (e.g., decreased ability to hear predators or hazards such as vessels) or reproduction (e.g., minor effects to navigation that may reduce mating opportunities).

Behavioral Responses

Any acoustic stimuli within sea turtle hearing ranges in the marine environment could elicit behavioral responses in sea turtles, including noise from sonar and explosions. Based on a limited number of studies, sea turtle behavioral responses to non-impulsive sounds could consist of temporary avoidance, increased swim speed, or no observable response (Lenhardt 1994; Lenhardt 2002; Kastelein et al. 2024). Similarly, based on a limited number of studies, sea turtle behavioral responses to impulsive sounds (i.e., air guns) could consist of temporary avoidance, increased swim speed, startle response, dive response, changes in depth; or there may be no observable response (O'Hara and Wilcox 1990; McCauley et al. 2000; DeRuiter and Doukara 2012; Kastelein et al. 2024). There is no evidence in the available literature to suggest that any sea turtle behavioral response to acoustic stressors would persist after the sound exposure.

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. Exposure to a single blast during

an event, which is the most probable scenario during HCTT 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. This is unlikely to occur since most activities involving explosives 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, while a large number of sea turtles may experience a behavioral response from exposure to explosives, the anticipated impacts on fitness and survival are minor and short-term.

Physiological Stress

ESA-listed sea turtles that experience either TTS, AINJ, or a significant behavioral response are also expected to experience a physiological stress response. A short, low level stress response may be adaptive and beneficial for sea turtles in that it may result in sea turtles avoiding the stressor and minimizing their exposure. 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 acoustic and explosive stressors resulting from HCTT activities are expected to be short-term in nature given that, in most cases, sea turtles would not experience repeated exposure to these stressors over a long period of time. As such, we do not anticipate stress responses would be chronic, involve distress, or have negative long-term impacts on any individual sea turtle's fitness.

Masking

Sea turtles likely use their hearing to detect broadband low-frequency sounds in their environment so the potential for masking would be limited to sound exposures that have similar characteristics (i.e., frequency, duration, and amplitude). Continuous and near-continuous human-generated sounds that have a significant low-frequency component, are not brief, and are of sufficient received level, are most likely to result in masking (e.g., proximate vessel noise and high-duty cycle or continuous active sonar). Other intermittent, short-duration sound sources with low-frequency components (e.g., air guns, aircraft noise, weapons noise, and explosives) would have limited potential for masking. Because sea turtles may rely primarily on senses other than hearing for interacting with their environment, any effect of masking may be mediated by reliance on other environmental inputs.

6.2.2 Vessel Strike Analysis

The majority of the proposed HCTT military readiness activities considered in this biological opinion involve vessel activity. While commercial vessel traffic is relatively steady throughout the year, Action Proponent vessel use within the action area is episodic, based on specific exercises being conducted at different times of the year and in different locations. Sea turtles must surface to breathe, and several species are known to either bask or feed at the surface for long periods. As such, sea turtles are vulnerable to being hit by a vessel or USV transiting

through areas where they may be resurfacing, resting or feeding at the surface. All of the ESA-listed sea turtles species present in the action area can occur at or near the surface, and therefore may be susceptible to strike from manned or unmanned surface vessels as part of the proposed action. Unlike when a vessel strikes a large whale, in many cases it may be difficult for an Action Proponent vessel to detect when it has struck a sea turtle. This is due to the relatively small size of a sea turtle compared to most of the vessels used by the Action Proponents for military readiness training and testing activities. However, reported sea turtle strandings with vessel strike injuries in inshore or nearshore areas used primarily (or exclusively) by military vessels (e.g., portions of Pearl Harbor and Kane'ohe Bay) can, with some degree of certainty, be attributed to Action proponent vessel activity (see Figures 1 and 2 in Kelly 2020).

Although activity-based mitigation measures for underway vessels will be implemented (see Table 6), sea turtles are often difficult to sight from a moving vessel, which may significantly reduce the effectiveness of Lookouts in terms of minimizing vessel strike risk. In order to protect sea turtles from propeller strikes, JBPHH has installed prop guards on all small boats under Naval Station's control (NAVFAC 2024). Vessel speed restrictions exist within the entirety of the Pearl Harbor Naval Defensive Sea Area, including the harbor mouth. In most areas the speed of larger vessels is limited to 10 knots.

Vessel use for HCTT activities resulting in strikes of sea turtles would most likely occur in areas that overlap high density sea turtle habitats, particularly nearshore foraging areas or directly off nesting beaches. Sea turtles are expected to be more highly dispersed in deeper offshore waters and, given the large open-ocean areas over which Action Proponent vessels could potentially conduct testing and training activities, the likelihood of co-occurrence is much lower in offshore waters. Leatherback turtles, in particular, could be impacted by offshore vessel movement given this species' preference for open-ocean habitats and its surface foraging behavior.

Sea turtle vulnerability to vessel strike increases with vessel speed. High speed vessel movements increase the potential risk of vessel strikes by reducing the available reaction time of both the sea turtle and vessel operators to an impending strike (Hazel et al. 2007). Hazel et al. (2007) found that vessel operators could not rely on turtles at the surface to actively avoid being struck for vessel speeds greater than 4 km per hour, well below the typical operating speeds of most Action Proponent vessels. Hazel et al. (2007) examined vessel strike risk to green sea turtles and suggested that sea turtles may habituate to vessel sound and are more likely to respond to the sight of a vessel rather than the sound of a vessel, although both may play a role in eliciting responses (Hazel et al. 2007). Regardless of what specific stressor associated with vessels turtles are responding, they only appear to show responses (avoidance behavior) at approximately 10 m or closer (Hazel et al. 2007). This is a concern because faster vessel speeds also have the potential to result in more serious injuries (Work et al. 2010).

In inshore and nearshore waters of the action area where higher sea turtle abundance makes vessel encounters more likely, Action Proponent vessel use occurs more regularly and is mainly from small, high-speed (greater than 10 knots) vessels. Propulsion testing, which can involve

vessels operating at speeds in excess of 30 knots, may pose an even greater strike risk of sea turtle strike due to the very high vessel speeds. More information on Navy and Cost Guard vessel activity in the HCTT action area can be found above in Section 2.5.2 (Physical Disturbance and Strike Stressors – Vessels) and in the HCTT Phase IV BA (Navy 2024d). For a more detailed discussion of the general threat of vessel strike to ESA-listed sea turtles in the action area see Section 5.14 (Environmental Baseline: Vessel Strike).

Exposure Analysis

General Approach

The approach used to estimate sea turtle vessel and USV strikes was based on available strandings information (including cause of strandings) and the relative proportion of all vessel activity (e.g., commercial fishing vessels, non-fishing commercial vessels, recreational boats, cargo ships, ferries, cruise ships, and military vessels) within portions of the action area attributed to Action Proponent vessel activity. Our analysis focuses on the areas of greatest overlap between HCTT vessel activity and ESA-listed sea turtles (i.e., nearshore waters in close proximity to Navy or Coast Guard ports). For the HRC, our analysis focuses on nearshore waters off the main Hawaiian Islands, since many Action Proponent vessels are based out of Pearl Harbor, Oahu and vessel traffic is concentrated in this part of the Hawaiian Archipelago (Mintz 2016; Starcovic and Mintz 2021). Green sea turtle strandings are also higher on Oahu compared to other Hawaiian Islands. From 2008-2018, Oahu accounted for 77% of all documented strandings of green sea turtles with vessel strike injuries in Hawaii (Kelly 2020). Kelly (2020) identified Kane'ohe Bay and Pearl Harbor as sea turtle vessel strike "hot spots" around Oahu based on strandings location data.

Sea turtle densities are substantially lower in offshore waters compared to nearshore areas of the HRC. Sea turtles struck by vessels far offshore within the HRC are less likely to strand than those struck in more nearshore waters. Based on data from 1982-2003, Chaloupka et al. (2008) reported that 99% of green turtle strandings recorded in the Archipelago occurred around the four main islands of Kauai, Oahu, Maui (including Molokai and Lanai) and Hawaii, with most of these (75%) occurring around Oahu. We have no empirical data to indicate that Action Proponent vessels strike turtles in offshore waters of the HRC, and also no data to indicate what percent of those that may be struck in offshore waters strand. Therefore, turtle vessel strike in the HRC is calculated solely based on stranding data for the Main Hawaiian Islands.

For the California portion of the action area, HCTT traffic is heaviest in the easternmost part of the SOCAL RC. For HSTT Phase III, our sea turtle vessel strike analysis focused on San Diego Bay since Action Proponent vessel activity is concentrated in this part of the action area, which is also home to a resident foraging aggregation of about 60 East Pacific DPS green sea turtles. This sub-population co-existed with a power plant located in south San Diego Bay, California, for about 50 years until the plant was decommissioned in 2010 (Eguchi et al. 2020). A study by Eguchi et al. (2020) suggests that the turtles expanded their home ranges within the bay after the plant was closed, which may have increased their risk of vessel strike. For our sea turtle vessel

strike analysis, we include strandings data from counties north of San Diego to account for the expansion of the action area for HCTT Phase IV. Available survey data, strandings data, and fishery bycatch data indicate that sea turtles are rarely observed (alive or stranded) off the California coast north of San Luis Obispo County (Eguchi and Seminoff 2011; Welch et al. 2019; Moore 2021; NMFS 2021b), indicating very sparse density and low population abundance. Therefore, our analysis focuses on sea turtle vessel strikes within the nearshore and inshore portions of the PMSR and SOCAL RC, including San Diego Bay. We have no empirical data to indicate Action Proponent vessels strike sea turtles in the offshore portions of the SOCAL RC or PMSR, and also no data to indicate what percent of those that may be struck in offshore waters strand.

In addition to the range complexes within the HCTT action area, Action Proponent vessels would also traverse the transit corridor between Hawaii and Southern California. While a military vessel moving between range complexes could strike a sea turtle, the likelihood of this occurring is extremely small given the low sea turtle densities anticipated within the transit corridor and the small number of transits that would occur. We have no empirical data to indicate Action Proponent vessels strike turtles within the transit corridor. As such, we do not anticipate any such strikes in the transit corridor portion of the action area (i.e., discountable) and, therefore, do not discuss this area further in our sea turtle vessel strike analysis.

A few studies have reported live sea turtles with vessel strike injuries (Norem 2005; Deem et al. 2006; Blumenthal et al. 2009; Denkinger et al. 2013; Ataman et al. 2021). This indicates that under some circumstances vessel strikes can result in non-lethal effects to sea turtles. However, a study of sea turtle strandings records in southwest Florida from 1987-2021 suggests that sea turtles encountered as strandings have a very low likelihood of surviving vessel strikes long-term (Kincaid et al. (in press)). Approximately 95% of sea turtles with vessel strike injuries in this study were either dead upon recovery or died subsequent to rescue. Sea turtle injuries from vessel strike can lethally damage vital organs, result in fatal blood loss, lead to secondary infection, and cause chronic impairment and diminished fitness, sometimes leading to mortality later in life (Casal and Orós 2009; Foley et al. 2019; Phu and Palaniappan 2019). In addition, turtles who survive a vessel strike injury long enough for it to heal may suffer secondary health issues over time, such as reduced mobility which may increase the risk of future vessel strikes or predation, and neurogenic bowel dysfunction, which can ultimately lead to death (Kincaid et al. (in press)). While it is difficult to account for turtles who survive a vessel strike injury and never strand, results from this study suggest that, even though initial survivorship may be as high as 25%, the chance of long-term survival is extremely low, in general. The best available data suggest that collisions with vessels even at very low speeds can be lethal for sea turtles (Hazel et al. 2007; Work et al. 2010). Thus, in most instances, sea turtle survival of a vessel strike would only be possible for strikes by small vessels operating at slow speeds (i.e., less than 5-10 knots). For the proposed action, the large majority of Action Proponent vessel activity will be from large vessels and/or vessels operating above 10-15 knots. Given this, and the lack of data on non-lethal vessel strikes of sea turtles in the HCTT action area, for our effects analysis we focus on

estimating lethal vessel strikes, which includes strikes resulting in immediate mortality and strikes resulting in a serious injury that leads to mortality in the future. In doing so we rely on stranding data of sea turtles with evidence of vessel strike, and we apply a correction factor (i.e., a strandings probability described below) for unobserved vessel strikes that we believe reasonably captures and corrects for any non-lethal strikes that may occur as a result of the proposed action.

Strandings Probabilities

Strandings data can provide valuable information on minimum mortality at sea and likely causes of death attributed to both anthropogenic and natural factors (e.g., fishery bycatch, disease, or vessel strike). Autopsies of stranded turtles can often indicate the likely cause of stranding, including whether or not the turtle was struck by a vessel. Since it is possible that a vessel strike can occur post-mortem, vessel strike may not be the proximal cause of death in all stranded turtles exhibiting vessel strike wounds. However, for our analysis it is reasonable to assume that vessel strike was the likely cause of mortality for any stranded turtle with signs of vessel strike.

Estimating total at sea mortality based on reported strandings can prove challenging since stranding probabilities are usually very low and highly variable in space and time (Koch et al. 2013). Although sea turtle stranding rates are variable, strandings typically represent only a small portion of the total mortality, as predators, scavengers, wind, and currents prevent carcasses from reaching the shore (Koch et al. 2013). Previous studies suggest that the stranding probability of a sea turtle that dies at sea usually does not exceed 10-20% of total at-sea mortality, even in nearshore waters (Epperly et al. 1996; Hart et al. 2006; Mancini et al. 2012). In addition, many strandings may never be noticed or recorded in a database, particularly in more remote areas or areas without sea turtle stranding monitoring programs. In such areas the observed stranding rate is likely even smaller than the stranding probabilities predicted by experimental studies.

For the Main Hawaiian Islands we use 10% as a reasonable, science-based estimate of the observed stranding probability. This estimate considers (1) the large geographic span of the islands, (2) the physical factors (e.g., wind, currents, and bathymetry) in this region that may prevent a carcass from stranding, and (3) the remote, less populated areas within the Main Hawaiian Islands, some of which are used for Navy or other vessel activities, where unobserved strandings may occur. Available information does not allow us to estimate the percentage of vessel struck turtles that are observed stranded off of California. However, we expect this observed stranding rate to be somewhat higher than the Main Hawaiian Islands due to the large human populations along the Southern California coastal counties of San Diego, Orange, and Los Angeles. Therefore, based on the available information, we use 20% as a reasonable, science-based estimate of the observed stranding probability for sea turtle stranding off of California.

Estimating HCTT Vessel Activity

Combining available information on the number of reported strandings with evidence of vessel strike and the estimated stranding probability, we can arrive at an estimate of the total number of sea turtle vessel strikes within the action area. For purposes of our effects analysis, we then need to determine what proportion of the estimated sea turtles vessel strikes in the action area are attributable to Action Proponents vessels or USVs as part of the proposed action. To estimate vessel strikes by Action Proponent vessels and USVs we need to determine the proportional level of HCTT vessel/USV activity relative to all vessel activity that can result in sea turtle vessel strike. Vessel activity can be measured several ways, including number of ships/vessels, number of transits in and out of ports, or number of ship-hours on the water. Based on the information available for our analysis, we determined that ship-hours on the water was the best correlate of sea turtle ship strike risk. All other things being equal (e.g., vessel speed, vessel size, vessel noise, locations, and sea turtle densities), the number of vessel strikes should be roughly proportional to the number of vessel hours on the water.

Based on 2016-2023 vessel traffic (i.e., at-sea vessel hours) information provided by the Navy (C. Johnson, U.S. Navy Pacific Fleet, personal communication email sent to R. Salz, NMFS OPR, March 27, 2025)(Navy 2025k), Action Proponent vessels combined account for an estimated 6.7% of total vessel hours (i.e., military and non-military combined) throughout the entire HCTT action area. This estimate is based on data for large vessels, defined as commercial and military vessels greater than 65 ft in length (note: Navy vessels included in this analysis are all greater than 387 ft, whereas Coast Guard vessels range from 87 ft to 420 ft). While the 6.7% represents the proportion of Action Proponent vessel activity throughout the entire HCTT action area, for purposes of our sea turtle vessel strike analysis, we are primarily interested in vessel activity in the HRC and SOCAL RC. Action Proponent vessels account for 22.0% of overall vessel traffic (i.e., vessels greater than 65 ft) in the SOCAL RC and 2.4% of overall vessel traffic in the HRC. We use these percentages in our effects analysis as a proxy for the proportion of all sea turtles killed by vessel strike attributable to Action Proponent vessels. Lacking more detailed information on the relative percent of HCTT vessel traffic in only the nearshore portions of the HRC or SOCAL RC where sea turtle densities are highest and vessel strike risk is greatest, we assume that these percentages for each range complex are a close approximation of the percentages in these smaller areas. We believe this is a reasonable assumption since both HCTT and non-military vessel traffic are concentrated in these smaller, nearshore areas.

As noted above, this analysis is based on positional vessel data that only includes large vessels, generally those over 65 ft in length. Thus, smaller military boats, small commercial vessels and most recreational boats are not included. We anticipate that, due to their generally larger sizes and faster speeds, a large proportion of sea turtle vessel strikes within the action area would be from large commercial and military vessels. However, we recognize that sea turtles are also susceptible to vessel strike by smaller vessels, including small military vessels, recreational boats, and small fishing vessels. As such, we consider the percentages calculated above as a

reasonable, science-based estimate of the proportion of sea turtle vessel strikes that are attributable to HCTT vessels of all sizes.

Estimating Sea Turtle Vessel Strikes

We combined the available information from strandings reports with our estimated strandings probabilities and estimated proportion of vessel traffic attributed to Action Proponent vessels to arrive at an estimate of vessel strikes as a result of the action.

Based on updated information provided by the NMFS Pacific Islands Fisheries Science Center (PIFSC), the average annual number of green sea turtle strandings in the Main Hawaiian Islands due to boat strike in the 3-year period from 2022-2024 is 12 turtles (PIFSC 2025). Based on our strandings probability analysis above, we estimate that the 12 green sea turtles struck by all vessels annually represent about 10% of the total number of green sea turtles in Hawaiian waters that are struck annually by all vessels. Thus, our estimate of annual average number of green sea turtle vessel strikes (by all vessels) in Hawaiian waters is 120 turtles.

The next step is to determine the proportion of this total vessel strike estimate attributable to Action Proponent vessels and USVs as part of the action. For Hawaii, from above we estimated that 2.4% of all vessel traffic (based on vessel hours) is by Action Proponent vessels. Therefore, the estimated number of Central North Pacific DPS green sea turtle vessel or USV strikes annually by Action Proponent vessels operating in the HRC is 2.9 turtles (i.e., 120 * 0.024), which we round up to 3.

Sea turtles that strand due to vessel strike are nearly all found dead. Of all the possible causes of stranding, vessel strike was the most likely to result in a dead stranded green sea turtle in Hawaii (shark attack was the second most likely; Chaloupka et al. 2008). In a long-term study (1982-2003) of green sea turtle strandings in Hawaii, Chaloupka et al. (2008) found that over 95% of vessel strike caused strandings resulted in mortality. More recently, in 2014 100% of the vessel strike caused strandings of green sea turtle in Hawaii were found dead. The few turtles struck by vessels that strand alive would likely have serious injuries that may result in reduced fitness, increased vulnerability to other threats (e.g., predation and disease) and eventual mortality. Therefore, we assume that all three of the estimated green sea turtles that strand annually due to an Action Proponent vessel strike in the HRC would either be killed or seriously injured, which would likely lead to delayed mortality.

Green sea turtles account for over 95% of all sea turtle strandings in Hawaii (Chaloupka et al. 2008; NMFS 2015b). Based on historical data, hawksbills are the second most likely sea turtle species to strand, accounting for between 1% and 4% of strandings in Hawaii (Chaloupka et al. 2008; PIFSC 2025). For purposes of our analysis, we assume 4% of strandings in a given year could be hawksbills. To estimate hawksbill vessel strikes we apply the ratio of historical hawksbill strandings to green sea turtle stranding to our estimated number of green sea turtle vessel strikes by Action Proponent vessels from above. Therefore, the estimated number of

hawksbill sea turtle vessel or USV strikes annually by Action Proponent vessels operating in the HRC is 0.33 turtles (i.e., 4/96 * 8).

Based on historical data, olive ridley sea turtles likely account for less than 1% of all turtle strandings in Hawaii (Chaloupka et al. 2008). For purposes of our analysis, we assume 1% of strandings could be olive ridleys in a given year. Therefore, the estimated number of olive ridley sea turtle vessel or USV strikes annually by Navy vessels operating in the HRC is 0.1 turtles (i.e., 1/96 * 8 = 0.3).

Leatherbacks and loggerhead are very rarely found stranded in Hawaii and historically account an extremely small proportion (less than 0.1%) of observed sea turtle strandings (Chaloupka et al. 2008). This is likely a function of the low densities of these species in the HRC, particularly in nearshore areas with heavy vessel traffic. We have no information to indicate Action Proponent vessels strike these species within the HRC, although strandings probabilities for these species would likely be very low given their preference for offshore habitats. Based on the best available information, we find that the likelihood of a HCTT vessel o USV strike of a loggerhead or leatherback sea turtle in the HRC as part of the action to be extremely unlikely, and thus discountable.

For our vessel strike analysis in the California portion of the action area we used sea turtle strandings data for California for the years 2018-2024 provided by the NMFS Southwest Fisheries Science Center (LaCasella 2025). While sea turtles strand for a variety of reasons, for purposes of our analysis we only used records where the field “Boat Collision” was marked as “Yes.” While this field is not conclusive evidence that vessel strike was the proximal cause of death, it represents the best available information on which to base of vessel strike quantitative analysis. Although the time series used for our analysis includes years during the peak of the COVID pandemic (i.e., 2020-2021), the average annual number of turtle strandings with signs of vessel strike for these years was not significantly different than the average number of turtle strandings with signs of vessel strike for the years before (i.e., 2018-2019) or after (2023-2024) the pandemic.

From 2018-2024, there were 48 stranded sea turtles with signs of vessel strike off of California: 42 green, 5 olive ridley, and 1 loggerhead. San Diego County accounted nearly half (46%) of these strandings, followed by Los Angeles County (29%; primarily Long Beach), Orange County (19%), and Ventura and San Luis Obispo Counties (less than 1% each).

Based on our strandings probability analysis above, we estimate that stranded turtles off California (42 green, 5 olive ridley, and 1 loggerhead over seven years) represent about 20% of the total number of sea turtles that are struck annually by vessels. Thus, our estimate of the total number of sea turtle vessel strikes (by all vessels) within the California portion of the action area from 2018-2024 is 210 green, 25 olive ridley, and 5 loggerhead. On an annual basis we estimate 30 green, 3.6 olive ridley, and 0.7 loggerhead vessel strikes within the California portion of the action area.

Next, we determined the proportion of the California vessel strike estimate attributable to Action Proponent vessels as part of the action. Based on 2016-2023 vessel traffic (i.e., at-sea vessel hours) information provided by the Navy (C. Johnson, U.S. Navy Pacific Fleet, personal communication email sent to R. Salz, NMFS OPR, March 27, 2025)(Navy 2025k), Action Proponent vessels account for an estimated 22% of all vessel traffic, for vessels greater than 65 ft in length, within the SOCAL RC. Using this estimate of proportional Action Proponent vessel traffic, we estimate the number of sea turtle vessel and USV strikes annually by HCTT vessels and USVs operating in the SOCAL RC by species is as follows: 6.7 East Pacific DPS green sea turtles; 0.8 olive ridley sea turtles; and 0.2 North Pacific Ocean DPS loggerhead sea turtles. As discussed above for Hawaii, we assume that all stranded turtles due to vessel strikes in the SOCAL RC would result in mortality.

There were no records of leatherback or hawksbill sea turtles in the California strandings database for the years 2018-2024 (LaCasella 2025). This is likely a function of the very low densities of these species off California. Hawksbill sea turtles are considered a tropical or subtropical species and their range in the Eastern Pacific is primarily south of the action area. We have no information to indicate Action Proponent vessels strike leatherback sea turtles in the California portion of the action area, although strandings probabilities for this species would likely be very low given their preference for offshore habitats. Based on the best available information, we find that the likelihood of a HCTT vessel strike of a leatherback or hawksbill sea turtle in the California portion of the action are to be extremely unlikely, and thus discountable.

Anticipated Life Stages and Sex Ratios of Sea Turtle Vessel Strikes

Next, we consider the life stages of sea turtles that could be impacted by vessel strike as a result of the proposed action. Sea turtles vessel strikes are most likely occur in areas of overlap between high densities of sea turtles, high volume of HCTT vessel traffic, and military readiness activities involving vessels operating at high speeds. Within the HRC, coastal foraging habitats exist for green, hawksbill, and olive ridley sea turtles in nearshore and inland waters surrounding the Main Hawaiian Islands. In these areas, neritic juveniles, sub-adults, and adults of all three species are at risk of vessel strike. Based on the average size (i.e., 75.2 cm) of green sea turtle vessel strike strandings in Hawaii from 2014-2018, and the nesting female size range (i.e., 74.6 cm to 105.5 cm) for the Central North Pacific DPS (Balazs 2015), Kelly (2020) concluded that at least half of all vessel strike related green sea turtle strandings in Hawaii may affect breeding adults.

Near nesting beaches, sea turtle hatchlings would also be present, but very briefly as they leave the nest, enter the water, and move to offshore areas to mature. While the majority of Hawaiian green sea turtles nest in the Northwestern Hawaiian Islands, specifically French Frigate Shoals, some nesting also occurs in the Main Hawaiian Islands (Seminoff et al. 2015). Hawksbill sea turtles nest in the Main Hawaiian Islands, predominately on the Big Island of Hawaii, although each island has a few annual nesters. Given the limited time window and spatial overlap (i.e., adjacent to nesting beaches) with HCTT vessel traffic, we anticipate a very small number of

green and hawksbill hatchlings would be exposed to vessel strike within the HRC as a result of the proposed action. While olive ridley nesting has been documented in the Main Hawaiian Islands, it is considered an extremely rare occurrence such that the likelihood of an olive ridley hatchling vessel risk is discountable. Oceanic phase juveniles and hatchlings from all three species could be exposed to vessel strike from training and testing exercises occurring in the deeper, offshore waters of the HRC. However, sea turtles are expected to be highly dispersed in deeper, offshore waters and, given the large area over which Action Proponent vessels could potentially conduct HCTT activities, the likelihood of co-occurrence is much lower in these area as compared to nearshore coastal and inland areas.

We used information from the PIFSC sea turtle strandings database from 2019-2023 (PIFSC 2025) to approximate the proportional distribution of HCTT sea turtle vessel strikes within Hawaii as a result of the proposed action between males and females for each species. For the majority of strandings records, sex is recorded as "unknown" because, without an internal examination, it typically can only be determined if the animal is an adult and if the carcass is in good condition. However, we still consider this database to be the best available information to approximate the distribution of vessel strikes by sex, within the HRC. Green sea turtle sex ratios based on documented vessel strike strandings records in Hawaii are biased towards female (i.e., 66%). This is consistent with skewed sea turtle sex ratios reported in the published literature, likely due to environmental warming of nesting beaches (Laloë et al. 2024). For example, Hays et al. (2014) searched the published literature to assemble a database on hatchling sex ratios for sea turtle breeding sites across the globe. Out of 46 articles, including data collected from 1977-2011, a male biased sex ratio was reported in just four records, and only two studies supported a balanced sex ratio (Hays et al. 2014).

Within the California portion of action area, coastal foraging habitats exist for green sea turtles within nearshore and inshore waters and over the continental shelf. In these areas, neritic juveniles, putative adult, and adult green sea turtles are vulnerable to vessel strike. Olive ridleys are primarily a pelagic sea turtle, meaning they spend a large part of their lives in the open ocean, though they also inhabit coastal areas. Based on the California sea turtle strandings database (LaCasella 2025) for the years 2018-2024, the juvenile life stage account for about 82% of green and 73% of olive ridley sea turtle strandings due to vessel strike, with adults accounting for the other 18% and 27% for each species, respectively. Pre-recruitment (oceanic phase) juvenile green and olive ridley sea turtles could potentially be exposed to vessel strike from training and testing exercises occurring in the deeper, offshore waters of the SOCAL RC. However, sea turtles are expected to be highly dispersed in deeper, offshore waters and, given the large area over which Action Proponent vessels could potentially conduct HCTT activities, the likelihood of co-occurrence is much lower in these areas as compared to nearshore coastal and inland areas. Loggerhead sea turtles are more likely to be found farther offshore as compared to green or olive ridleys, particularly during periods of elevated ocean temperatures (e.g., marine heatwaves). Based on information from the loggerhead strandings database for California, we anticipated the large majority of HCTT vessel or USV strikes of this species to be juveniles.

We used information from the California sea turtle strandings database from 2018-2024 (LaCasella 2025) to approximate the proportional distribution of HCTT sea turtle vessel strikes within California as a result of the proposed action between males and females for each species. Similar to the Hawaii dataset, although sex is recorded as "unknown" for the majority of California strandings records, we still consider this database to represent the best available information for estimating the sex ratio of vessel struck sea turtles within the California portion of the HCTT action area. For all three sea turtle species that we anticipate HCTT vessel or USV strikes of in the California, sex ratios based on strandings records are biased towards female: green 66% (n=62), olive ridley 53% (n=60), and loggerhead 83% (n=12).

Sea Turtle Vessel Strike Exposure Analysis: Summary

Based on our exposure analysis, we find that vessel and USV strike of ESA-listed sea turtles by Action Proponent vessels is reasonably certain to occur as a result of the action. Most of the anticipated vessel or USV strikes will be green sea turtles from both the Central North Pacific DPS (up to three per year) and Eastern Pacific DPS (up to seven per year). We expect a much smaller number of hawksbill, olive ridley, and North Pacific DPS loggerhead sea turtles will be struck by Action Proponent vessels as a result of the proposed action (i.e., up to one strike per year per species or DPS). It is extremely unlikely that an Action Proponent vessel will strike a leatherback sea turtle as part of the action and thus, the effects of vessel strike on this species are considered discountable.

For both the Hawaii and California portions of the action area, the large majority of vessel strikes are expected to occur in nearshore coastal and inland sea turtle foraging habitats. Life stages that will be impacted by vessel strikes in these areas include neritic juvenile, subadult and adult green, hawksbill and olive ridley sea turtles. A very small number of green and hawksbill hatchlings could also be struck off of nesting beaches within the HRC as a result of the proposed action. We expect the large majority of loggerhead sea turtles struck by Action Proponent vessels to be juveniles.

Due to the anticipated female biased sex ratios of the affected sea turtle populations, we expect the majority of vessel strikes for all species and DPSs will be with females. For both DPSs of green sea turtle that will be impacted, sex ratios from strandings data indicate that about two-thirds of the anticipated vessel strikes will be female.

Response Analysis

From our exposure analysis above, we assume that any sea turtle strike as a result of the proposed action would result in either immediate or delayed mortality. Collisions with vessels would likely result in blunt trauma, lacerations, or mortality. Some sea turtles may be compromised and sustain infection, have reduced foraging abilities, experience higher predation risks, or die some time later as a result of vessel strike injuries. While some sea turtles may initially survive a vessel strike injury, given the average size and speed of Action Proponent vessels, the chance of long-term survival is extremely low (Kincaid et al. (in press)).

6.3 Fishes

This section addresses the effects of sonar and other transducers, air guns, in-water explosives, and vessel strikes on the ESA-listed fish in the action area.

6.3.1 Acoustic and Explosives Effects Analysis Approach

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. For many of the HCTT acoustic stressors affecting fishes in the action area we rely on recommendations in the 2014 ANSI Technical Report (*Sound Exposure Guidelines for Fishes and Sea Turtles*) as a starting point (Popper et al. 2014a). More recent information from published research has been incorporated into this analysis, as available and relevant for purposes of our fish acoustics effects analysis.

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 a cumulative pressure metric (SEL_{cum}) threshold. NMFS also applies a RMS sound pressure level threshold for some acoustics sources to assess whether behavioral responses may be elicited during some sound exposures.

Sonar

General categories and characteristics of the sonar and other transducer systems proposed for use during HCTT activities are described in Sections 2.2 and 2.5.1. Because of the sheer number and diversity of fishes, only a limited number of species have had hearing capabilities tested. Figure 80 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 80 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 HCTT defined sonar classes described in Section 2.5.1.

To evaluate the effects of sonar use during HCTT activities, we use the criteria for sonar and fishes based upon the recommendations provided in the 2014 ANSI Technical Report (Popper et al. 2014a). The recommended threshold for onset of TTS for fish with a swim bladder that is not

involved in hearing would be low-frequency sonar exposure levels greater than 210 dB SEL_{cum} re 1 μPa^2-s. TTS has not been observed from exposure to mid-frequency active sonar in fish species within this hearing group. Fishes within this hearing group, which include the ESA-listed sturgeon and salmon species considered in this opinion, 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 have been proposed for fishes with a swim bladder that is not involved in hearing from exposure to mid-frequency active sonar. Since the other ESA-listed fish considered in this opinion (i.e., sharks and rays) lack a swim bladder their sensitivity to low-frequency sonar is expected to be less than for those species with a swim bladder.

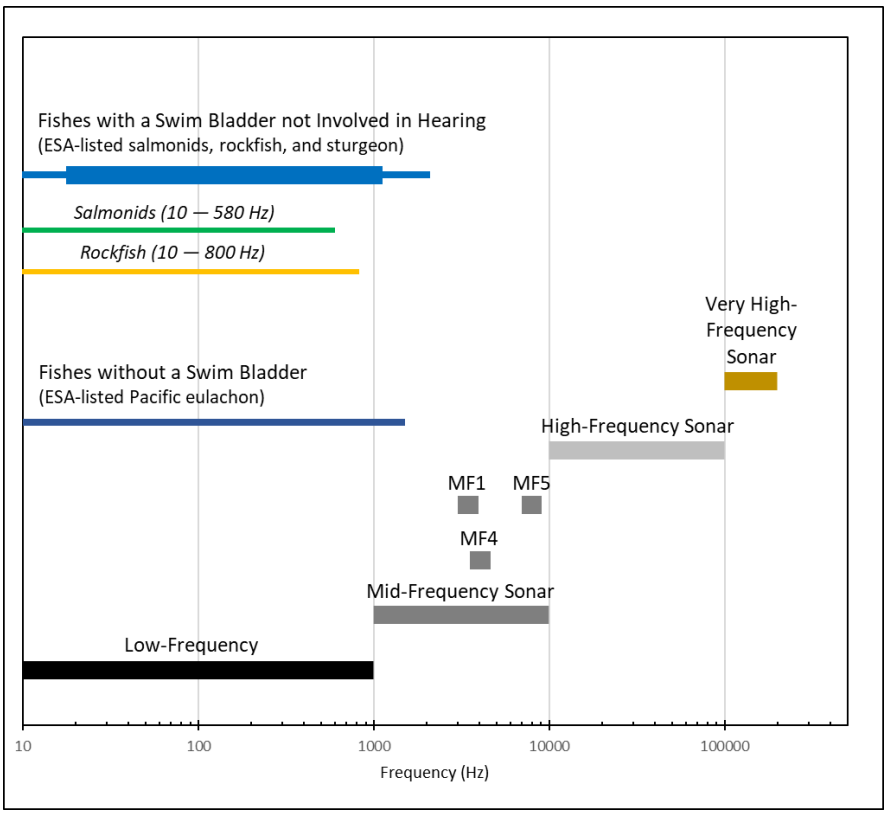


Figure 80. Hearing Frequency Ranges (pressure) of some ESA-listed Fish Groups Compared to Navy Defined Sonar Classes. Sources: (Tavolga and Wodinsky 1963; Hawkins and Johnstone 1978; Popper et al. 2007; Popper 2008; Popper et al. 2014a).

Figure 80 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 kHz 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. MF1 = 3.5 kHz, MF4 = 4 kHz, MF5 = 8 kHz.

All ESA-listed fishes considered in this opinion have the potential to be exposed to sonar and other transducers during HCTT activities in the action area. 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 HCTT activities possess any hearing specializations, and all hear primarily below 2 kHz. 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 analysis that possess a swim bladder (i.e., eulachon, steelhead, Chinook salmon, and coho salmon), 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 fishes with swim bladders not involved in hearing.

Exposure to Navy Surveillance Towed Array Sensor System (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 (Popper et al. 2007; Kane et al. 2010). 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% 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 kHz) to study the effects of the exposure on the survival, development, and behavior. The fish were placed in plastic bags

3 m 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 kHz. 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%. 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 kHz) sonar at received SPLs of 210 dB re: 1 uPa, resulting in cumulative SELs of 220 dB re: 1 uPa. The researchers 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 HCTT 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 HCTT proposed activities (i.e., due to the transient nature of HCTT 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) 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%) 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 gourami 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 Action Proponents. 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 proposed HCTT activities. In addition, the relative risk of injury or mortality to fish with no swim bladders (i.e., sharks and rays) exposed to low and mid-frequency sonar is lower than fish with swim bladders, no matter the distance from the source.

Based on the studies discussed above and other research, direct injury from sound levels produced from the type of sonar the Action Proponents use has not been documented in fishes (Popper et al. 2007; Kane et al. 2010; Halvorsen et al. 2012; Popper et al. 2013; Popper et al. 2014a). These sound sources are considered to pose less risk to fish species because the sound produced from sonar and other transducers characteristically has lower peak pressures and slower rise times than other acoustic stressors that are known to injure and kill fish (e.g., impulsive sounds from pile driving, or the strong shock waves produced from detonation of explosives). As such, injury of ESA-listed fish from sonar is extremely unlikely (i.e., discountable), and is not considered further in this analysis. Therefore, potential impacts from exposure to HCTT sonar on ESA-listed fish are limited to TTS, masking, physiological response (including stress), and behavioral reactions.

Most marine fishes, including all ESA-listed species within the action, are either hearing generalists or lack a swim bladder, and would be unable to detect frequencies greater than approximately 2 kHz. Based upon the fish hearing and frequency overlap, the ESA-listed fishes considered in this biological opinion would have limited ability to detect mid-frequency active sonar frequencies, and would not likely be able to detect mid-frequency active sonar sources proposed for HCTT. Further, none of the fish species considered in this opinion can detect high- and very high-frequency sonars and other transducers. Therefore, ESA-listed fish species considered in this opinion are extremely unlikely to be affected by either mid-frequency or high-frequency sonar sources.

The HCTT proposed action includes a relatively small number of activities involving low-frequency sonar use in areas that would overlap with ESA-listed fish species. ESA-listed fish

species may only detect low-frequency sonar systems with higher source levels within a few km; and most other, less powerful low-frequency sonar systems, at much shorter ranges. Based on calculated ranges to effects for fishes exposed to military sonar in other Navy action areas (Navy 2024d), TTS is not anticipated from HCTT low-frequency sonar sources. 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 HCTT sonar sources are continuously 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 HCTT sonar and other transducers on ESA-listed fish are considered insignificant, and thus not likely to cause adverse effects.

Air Guns

As described in 2.5.1.5, the proposed action includes the use of small air guns with capacities up to 60 in^3 for testing activities in various offshore portions of the action area. Air guns could be used up to 57 days per year in the HRC, up to 57 days per years in the NOCAL RC, and up to 44 days per years in the SOCAL RC (Navy 2025e). The introduction of anthropogenic noise from airgun operations or other non-explosive impulsive sounds into marine environments can result in adverse effects to fish including mortality, injury and behavioral modifications depending on the distance from the source and the received sound pressure levels. The Navy calculated the ranges to effects for air guns by modeling the distance that sound would need to propagate to reach exposure level thresholds specific to a fish hearing group or category that would cause TTS, injury, and mortality (Navy 2025e). Range to effects for injury and mortality from HCTT air guns are 2 m or less for all fish functional hearing groups for all depths and cluster sizes (i.e., 1 and 10 shots) modeled, based on calculations using both SPL and SEL exposures (Navy 2025e). For TTS, the calculated range to effects for fishes with a swim bladder is 5 m for all depths and with a cluster size of 10 air gun shots, based on calculations using both SPL and SEL exposures (Navy 2025e). Based on these ranges to effects, an ESA-listed fish would need to be in very close proximity to an air gun blast for exposure to result in mortality, injury or TTS. Given the anticipated low densities of ESA-listed fish species in the portions of the action area where HCTT air guns would be used, the dispersed nature of activities involving air guns throughout the vast action area, and the natural avoidance behavioral traits of ESA-listed fish species to swim away from the noise and physical disturbance produced by large military

vessels, we find it extremely unlikely (i.e., discountable) that an ESA-listed fish would be in close enough proximity to a HCTT air gun shot such that sound exposure levels would result in either mortality, injury, or TTS.

Although not calculated for HCTT, ranges to effects for behavioral responses from air guns are likely much larger than those calculated for mortality, injury or TTS. For example, in one study, fish behavioral responses to seismic airgun arrays operating in water depths of 100-120 m were observed at distances of approximately 2 km (Weilgart 2013). Fish behavioral responses to air guns suggest that the impact zone extends well beyond the immediate vicinity of the sound source. Fish demonstrate several immediate behavioral responses to air gun noise that reflect their acute sensitivity to acoustic disturbance. The behavioral impacts of airgun exposure extend beyond direct physiological effects. Typical fish responses to airgun operations include initial startle response, altered swimming patterns, moving toward the bottom of the water column, swimming at accelerated speeds, changes in schooling behavior, and modifications to feeding activities (Peng et al. 2015; Slabbekoorn et al. 2019). Research has shown that these changes in swimming behavior can persist throughout the duration of noise exposure, with fish exhibiting increased alarm responses and modified movement patterns (Peng et al. 2015). Airgun noise can also significantly influence how fish utilize their habitat.

The intensity of these responses appears to vary among species, suggesting that behavioral reactions to noise are influenced by multiple variables including the fish's location, physiological state, age, body size, and school size. Mortality and potential damage to the cells of the lateral line have been observed in fish larvae, fry, and embryos after exposure to single shots from a seismic air gun within close proximity (i.e., 0.1 to 6 m) to the sound source (Cox et al. 2012). However, exposure of adult pallid sturgeon (*Scaphirhynchus albus*) and paddlefish (*Polyodon spathula*) to a single shot from an air gun array (four air guns) within similar ranges (6 m) did not result in any signs of mortality within seven days after exposure (Popper et al. 2016).

Elasmobranchs (e.g., sharks and rays), like all fish, have an inner ear capable of detecting sound and a lateral line capable of detecting water motion caused by sound (Hastings 2005; Popper 2009). Data for elasmobranch fishes suggest they are capable of detecting sounds from approximately 20 Hz to 1 kHz with the highest sensitivity to sounds at lower ranges (Casper et al. 2012; Casper and Mann 2009; Casper 2006; Ladich and Fay 2013; Myrberg Jr. 2001; Yan 2003). However, unlike most teleost fish, elasmobranchs do not have swim bladders (or any other air-filled cavity), and thus are unable to detect sound pressure (Casper et al. 2012). Particle motion is presumably the only sound stimulus that can be detected by elasmobranchs (Casper et al. 2012). Given their assumed hearing range, elasmobranchs are anticipated to be able to detect the low frequency sound from an airgun array, if exposed. However, the duration and intensity of low-frequency acoustic stressors will likely minimize the effect this stressor has on elasmobranchs. Furthermore, although some elasmobranchs have been known to respond to anthropogenic noise, in general elasmobranchs are not considered particularly sensitive to noise (Casper et al. 2012).

The precise expected response of ESA-listed fishes to low-frequency acoustic energy is not completely understood due to a lack of sufficient experimental and observational data for these species. However, given the signal type and level of exposure to the low-frequency signals used in seismic survey activities, we do not expect adverse effects (including significant behavioral adjustments, TTS, PTS, injury, or mortality). As summarized in the *ANSI Sound Exposure Guideline Technical Report* (Popper et al. 2014a), exposure to high intensity and long duration impact pile driving or air gun shots has not caused mortality, and fishes typically recovered from injuries in controlled laboratory settings. Barring other proxies to rely upon, species tested to date can be used as surrogates for investigating injury in other species exposed to similar sources (Popper et al. 2014a). The most likely response of ESA-listed fishes exposed to HCTT air gun operations, if any, will be minor, temporary changes in their behavior including increased swimming rate, avoidance of the sound source, or changes in orientation to the sound source, none of which are expected to result in fitness impacts for the individual exposed.

In summary, we find that any effects to ESA-listed fish from HCTT air gun use are insignificant due to the: 1) limited use and relatively small footprint of air guns throughout the HCTT action area; 2) anticipated low densities of ESA-listed fish in areas that may overlap with air gun use; and 3) very small ranges to effects for impacts that could result in adverse effects with fitness consequences. Therefore, we conclude that noise from air guns used as part of the proposed action may affect, but is not likely to adversely affect ESA-listed fish within the HCTT action area.

Explosives – Fishes

All ESA-listed fishes that may be present in the action area are capable of detecting sound produced by explosions. In general, the introduction of explosive energy into marine environments through military readiness training and testing activities presents significant risks to fish species. This determination stems from the extensive range of potential impacts, which include direct mortality, non-lethal injury, temporary hearing loss, physiological stress, acoustic masking, and various behavioral modifications. The severity of these effects varies based on multiple factors, including the explosive characteristics, environmental conditions, and the physiological attributes of exposed species. Understanding these impacts requires careful examination of both the nature of explosive exposure and the biological responses of different fish species. This analysis draws from recent assessments to provide an updated understanding of explosive effects on ESA-listed fish species in the action area.

NMFS does not currently have “formal” criteria established for explosives thresholds and effects on fishes, and in most cases bases interim thresholds upon the lowest level of sound where onset of injury may occur. For effects of explosives on fish, we used the mortality criteria provided in the 2014 ANSI Guidelines (Popper et al. 2014a). Based on these guidelines, the peak pressure mortality threshold of 229 dB re: 1 μPa is applied for fish with and without swim bladders. The 2014 ANSI Technical Report did not suggest numeric thresholds for fish injury or TTS due to explosives (only mortality).

For HCTT Phase III (and other Navy Phase III consultations) we used the impact pile driving and airgun injury thresholds suggested by the ANSI Technical Report as a surrogate for explosives. However, for this consultation we have revised that threshold based on new information from studies on the effects of explosives on fish.

Dahl et al. (2020) reported the effects of underwater explosions on Pacific sardines (*Sardinops sagax*), a species with a physostomous swim bladder (i.e., an open swim bladder with direct connection to the gut via pneumatic duct). 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 m), there were inconsistent findings at the 50-154 m range (e.g., higher rates of injury were shown at a distance of 154 m compared to 68 m). The acoustic analysis led to the suggestion that, at a range of 154 m, 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 m despite the higher peak pressures observed at 68 m, 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).

Jenkins et al. (2022) studied the physical effects on non-auditory tissues of Pacific mackerel (*Scomber japonicus*), another hearing generalist, exposed to sound from underwater explosions. Primary effects were damage to the swim bladder and kidney that exceeded control levels at ≤333 m from the explosion (peak sound pressure level 226 dB re: 1 µPa, SEL 196 dB re: 1 μPa^2 s, pressure impulse 98 Pa s). A proportion of fish were dead upon retrieval at 26–40 minutes post exposure in 6 of 12 cages located ≤157 m from the explosion. All fish that died within this period suffered severe injuries, especially swim bladder and kidney rupture. The authors note that, based on the limited range in fish mass and length ratios utilized in the study, fish size was not important in determining susceptibility to injury and that peak pressure and SEL were better predictors of injury than was pressure impulse. In a related study, Ryan et al. (2022) looked at the physical effects on the inner ear of Pacific mackerel exposed to sound from underwater explosions (i.e., a single detonation of 4.5 kg of C4 explosives). Hair cell ciliary bundles counts were significantly reduced in fish exposed to the explosion as compared to the controls. The extent of effect also varied in animals at different distances from the explosion, with damage occurring in fish as far away as 400 m. Ryan et al. (2022) observed physical damage in the inner ear of Pacific mackerel, exposed to underwater explosions starting at received peak to peak SPLs of 220 dB re: 1 µPa.

Bowman et al. (2024) placed caged Pacific mackerel at targeted ranges of 150–800 m from an explosive source Animals were left at depth for about three hours post-exposure to determine immediate effects on survival. Fish were then retrieved and assessed for physical damage. They found that peak acoustic pressure levels that produced consistently high injury rates were about

230 dB re: µPa (Figure 81), which were in effect at ranges from the source of less than about 200 m (Bowman et al. 2024). In contrast, the peak levels for seismic sources are generally smaller, being, for example, approximately 214 dB re: µPa at range 50 m (Vereide et al. 2023). These results suggest that using pile driving or airgun injury threshold data as a surrogate for explosive thresholds may overestimate impacts to fish. While the type and severity of non-ear injuries reported during simulated impact pile driving studies are similar to those reported in explosive experiments, Bowman et al. (2024) note that caution must be taken in comparing results between effects of pile driving and explosions for several reasons. First, it must be noted that effects from explosions are from a single blast, whereas effects from pile driving come from the exposure to multiple strikes over minutes or hours. Second, there are significant and substantial differences in the characteristics of the pulses between pile driving and explosions, and these may result in very different effects on the swim bladder and how it interacts with surrounding tissues (Bowman et al. 2024).

From Bowman et al. (2024), as shown in Figure 81, injury rates for "any injury" do not approach 50% until around 220 dB re: µPa. Data from other recent explosive exposure studies also show reported rates of injury significantly different than controls starting at peak sound pressure levels of around 226 dB (Dahl et al. 2020; Jenkins et al. 2022; Jenkins et al. 2023). Therefore, based on the best available information, we apply a peak pressure fish injury threshold of 220 dB for our acoustic effects analysis for this consultation.

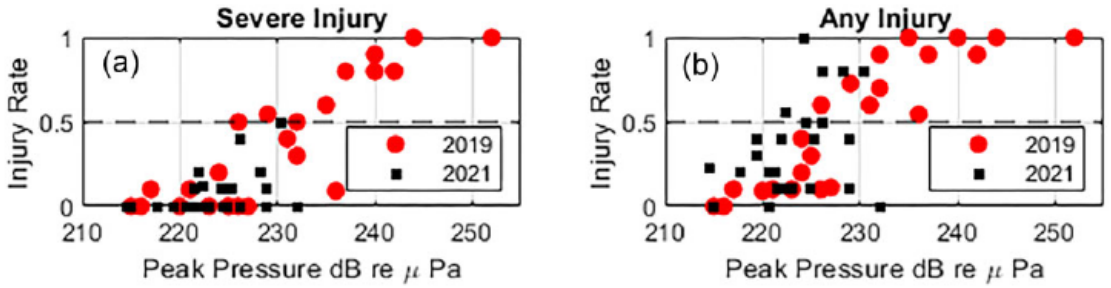


Figure 81. Injury Rates for Severe Injury (a) and Any Injury (b) of Pacific Mackerel Exposed to Explosive Detonations at Varying Sound Pressure Levels

Notes on Figure 81: Data represents 10 individual trial days (5 per year), using ~10 fish per cage across 50 cages (n = 501 fish). Points represent the mean injury response within each treatment cage. The dashed, horizontal lines denote the 50% injury rate.

Though there are no direct measurements of TTS in fishes exposed to explosive sources, it is assumed that fish would demonstrate similar effects on auditory detection as those exposed to other impulsive sources such as those described above. These received sound levels likely represent thresholds at which hearing effects may occur. Auditory injury has not been known to occur in fishes tested to date. Any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al. 2014a). The lowest SEL at which TTS has been observed in fishes with hearing specializations

exposed to air gun signals is 186 dB re: 1 μPa^2s. As reviewed in the *ANSI Sound Exposure Guideline* technical report (Popper et al. 2014a), hearing generalists would be less susceptible to TTS than specialists, even at higher levels and longer durations. Therefore, for the ESA-listed fish with swim bladders (i.e., sturgeon and salmon), we anticipate that TTS would occur at received levels above 186 dB. Consistent with the recommendations in the guidelines, fishes without a swim bladder (i.e., sharks, rays, and sawfish) would not be susceptible to TTS and therefore no criteria are proposed.

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).

Exposure Analysis

Fish species demonstrate varying levels of susceptibility to explosive impacts based primarily on their anatomical features. Species with different hearing specializations and swim bladder configurations demonstrate varying levels of susceptibility to explosive effects. The presence and type of swim bladder plays a particularly crucial role in determining sensitivity to explosive pressure waves, as these organs can amplify the effects of underwater explosions on surrounding tissues. The 2014 ANSI Guidelines establish distinct categories for assessment, recognizing that different fish species respond differently to explosive exposure. These categories include fish with no swim bladder, fish with swim bladders not involved in hearing, and fish with swim bladders involved in hearing (Popper et al. 2014a). This classification system acknowledges that species with swim bladders are particularly vulnerable to barotrauma and sudden pressure changes from impulsive sounds, which can result in immediate or delayed mortality (Popper et al. 2014a). Additionally, the fish's size, proximity to the explosive source, position in the water column (i.e., depth), and geometry of exposure can all significantly influence the potential for injury or mortality.

The Navy used NAEMO to calculate ranges to effects for ESA-listed fish species based upon the explosive criteria thresholds discussed above. Fishes within these ranges would be predicted to receive the associated effect. Ranges may vary greatly depending on factors such as the cluster size of the explosives, location, depth, and season of the activity. For purposes of this analysis, in-air sources detonating at or near (within 10 m) the surface were modeled as if detonating completely underwater at a source depth of 0.1 m, with all energy reflected into the water rather than released into the air (Navy 2025a). Therefore, the amount of explosive and acoustic energy entering the water, and consequently the estimated ranges to effects, are likely to be overestimated for detonations at or near the water surface (Navy 2025a).

Explosive ranges to effects for ESA-listed fish species without a swim bladder (i.e., giant manta ray, oceanic whitetip sharks, and scalloped hammerhead sharks) and hearing generalist species with a swim bladder (i.e., eulachon, steelhead, Chinook salmon, and coho salmon) are presented in Table 59 and Table 60, respectively.

Table 59. HCTT Explosive Ranges to Effects for Fishes without a Swim Bladder (Navy 2025a)

Bin	Depth	Cluster Size	TTS	INJ	MORT
E1	≤200 m	1	NA	86 m (5 m)	15 m (4 m)
	>200 m	1	NA	88 m (5 m)	18 m (4 m)
E2	≤200 m	1	NA	136 m (15 m)	38 m (6 m)
	>200 m	1	NA	137 m (15 m)	39 m (7 m)
E3	≤200 m	1	NA	246 m (10 m)	79 m (9 m)
	>200 m	1	NA	247 m (13 m)	76 m (8 m)
E4	≤200 m	1	NA	445 m (32 m)	162 m (16 m)
	>200 m	1	NA	446 m (38 m)	157 m (14 m)
E5	≤200 m	1	NA	415 m (32 m)	151 m (14 m)
	>200 m	1	NA	398 m (29 m)	146 m (11 m)
E6	≤200 m	1	NA	580 m (58 m)	221 m (27 m)
	>200 m	1	NA	579 m (53 m)	224 m (25 m)
E7	≤200 m	1	NA	702 m (26 m)	280 m (10 m)
	>200 m	1	NA	708 m (25 m)	281 m (10 m)
E8	≤200 m	1	NA	901 m (53 m)	354 m (10 m)
	>200 m	1	NA	895 m (53 m)	353 m (11 m)
E9	≤200 m	1	NA	951 m (48 m)	454 m (20 m)
	>200 m	1	NA	951 m (52 m)	456 m (21 m)
E10	≤200 m	1	NA	1,275 m (113 m)	576 m (58 m)
	>200 m	1	NA	1,263 m (123 m)	576 m (53 m)
E11	≤200 m	1	NA	2,059 m (85 m)	739 m (14 m)
	>200 m	1	NA	2,028 m (156 m)	747 m (37 m)
E12	≤200 m	1	NA	1,750 m (7 m)	760 m (2 m)
	>200 m	1	NA	1,726 m (32 m)	758 m (11 m)

Bin	Depth	Cluster Size	TTS	INJ	MORT
E13	≤200 m	1	NA	6,486 m (348 m)	2,972 m (132 m)
E16	>200 m	1	NA	9,576 m (645 m)	3,757 m (168 m)

Median ranges with standard deviation ranges in parentheses, TTS ranges are SEL-based and for fishes with a swim bladder only
TTS = Temporary Threshold Shift, INJ = Injury, MORT = Mortality, NA = not applicable
version.20250226

Table 60. HCTT Explosive Ranges to Effects for Fishes with a Swim Bladder (Navy 2025a)

Bin	Depth	Cluster Size	TTS	INJ	MORT
E1	≤200 m	1	< 45 m (7 m)	86 m (5 m)	15 m (4 m)
		5	< 90 m (18 m)	NA	NA
		25	< 187 m (50 m)	NA	NA
		50	< 254 m (33 m)	NA	NA
	>200 m	1	< 2 m (15 m)	88 m (5 m)	18 m (4 m)
		5	< 67 m (22 m)	NA	NA
		25	< 170 m (9 m)	NA	NA
		50	< 240 m (8 m)	NA	NA
E2	≤200 m	1	< 45 m (7 m)	136 m (15 m)	38 m (6 m)
	>200 m	1	< 46 m (8 m)	137 m (15 m)	39 m (7 m)
E3	≤200 m	1	< 97 m (34 m)	246 m (10 m)	79 m (9 m)
		5	< 199 m (73 m)	NA	NA
		25	< 388 m (146 m)	NA	NA
	>200 m	1	< 100 m (22 m)	247 m (13 m)	76 m (8 m)
		5	< 180 m (6 m)	NA	NA
		25	< 384 m (10 m)	NA	NA
E4	≤200 m	1	< 286 m (124 m)	445 m (32 m)	162 m (16 m)
	>200 m	1	< 170 m (16 m)	446 m (38 m)	157 m (14 m)
E5	≤200 m	1	< 160 m (198 m)	415 m (32 m)	151 m (14 m)
		5	< 302 m (57 m)	NA	NA
	>200 m	1	< 140 m (7 m)	398 m (29 m)	146 m (11 m)
		5	< 300 m (8 m)	NA	NA

Bin	Depth	Cluster Size	TTS	INJ	MORT
		20	< 550 m (12 m)	NA	NA
E6	≤200 m	1	< 457 m (312 m)	580 m (58 m)	221 m (27 m)
		15	< 1,827 m (157 m)	NA	NA
	>200 m	1	< 330 m (86 m)	579 m (53 m)	224 m (25 m)
E7	≤200 m	1	< 280 m (55 m)	702 m (26 m)	280 m (10 m)
	>200 m	1	< 270 m (81 m)	708 m (25 m)	281 m (10 m)
E8	≤200 m	1	< 489 m (57 m)	901 m (53 m)	354 m (10 m)
	>200 m	1	< 483 m (54 m)	895 m (53 m)	353 m (11 m)
E9	≤200 m	1	< 610 m (348 m)	951 m (48 m)	454 m (20 m)
	>200 m	1	< 437 m (14 m)	951 m (52 m)	456 m (21 m)
E10	≤200 m	1	< 694 m (139 m)	1,275 m (113 m)	576 m (58 m)
	>200 m	1	< 695 m (132 m)	1,263 m (123 m)	576 m (53 m)
E11	≤200 m	1	< 1,881 m (81 m)	2,059 m (85 m)	739 m (14 m)
	>200 m	1	< 1,806 m (107 m)	2,028 m (156 m)	747 m (37 m)
E12	≤200 m	1	< 677 m (1 m)	1,750 m (7 m)	760 m (2 m)
	>200 m	1	< 678 m (12 m)	1,726 m (32 m)	758 m (11 m)
E13	≤200 m	1	< 7,875 m (202 m)	6,486 m (348 m)	2,972 m (132 m)
E16	>200 m	1	< 10,965 m (491 m)	9,576 m (645 m)	3,757 m (168 m)

Median ranges with standard deviation ranges in parentheses, TTS ranges are SEL-based and for fishes with a swim bladder only
TTS = Temporary Threshold Shift, INJ = Injury, MORT = Mortality, NA = not applicable
version.20250226

Most HCTT explosive activities will occur in the SOCAL RC, the HRC, and PMSR (Navy 2025a). Only a small number of explosive activities are planned within the NOCAL RC (see Table 31). Most activities involving in-water explosives associated with large caliber naval gunfire, or the launching of targets, missiles, bombs, or other munitions, are conducted more than 12 nm from shore (Navy 2025a). Small Ship Shock Trials could occur in the SOCAL RC greater than 12 nm from shore and Sinking Exercises are conducted greater than 50 nm from shore (Navy 2025a). Activities with explosives that could be conducted closer to shore include Mine Warfare and Expeditionary Warfare activities. In the Hawaii Study Area explosive activities could occur at specified ranges and designated locations around Oahu, including the Puuloa Underwater Range and designated locations in and near Pearl Harbor (Navy 2025a). In the SOCAL RC, explosive activities could occur near San Clemente Island, in the Silver Strand

Training Complex, and in other designated mine training areas along the southern California coast (Navy 2025a).

The large majority of explosive munitions used for HCTT activities will detonate at or just above the water surface (greater than 90% by count) (Navy 2025a). These include those used during surface warfare activities, such as explosive gunnery, bombs, and missiles. Sound and energy from in-air detonations would be reflected at the water surface and, therefore, would be less likely to adversely affect fish species located throughout the water column. By contrast, since the sound and energy produced by in-water explosives is not dissipated by the air-water interface, these stressors have a much greater potential to impact fish. The types of activities with detonations below the surface include Mine Warfare, activities using explosive torpedoes, and ship shock trials, as well as specific training and testing activities (Navy 2025a).

The Action Proponents will implement mitigation measures to avoid impacts from explosive military readiness activities on artificial reefs, hard bottom substrate, and shipwrecks throughout the Study Area (see the Section 2.4.2 for details), which consequently, will help avoid potential impacts on fishes that shelter and feed within those habitats. In addition, hammerhead sharks are included in the activity-based mitigation measure for explosive mine neutralization events involving Navy divers (described in Section 2.4.2, Table 5), which may further reduce the number and severity of exposures to HCTT explosive stressors for ESA-listed scalloped hammerhead sharks.

Surrogate for Impacts to ESA-listed Fish from Explosives

Due to the lack of available density or abundance information for ESA-listed fish species in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish from each species that would be exposed to stressors from 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 the number of fish suffering mortality, injury, TTS, and behavioral disruption) or to monitor take-related impacts in terms of individuals of the ESA-listed fish species in the action area.

Because it is not practical to express the amount of anticipated take of these fish species, we instead use a surrogate measure to express the amount or extent of incidental take. The surrogate for the incidental take of the ESA-listed fish listed above is based on: 1) the ranges to effects (Table 59 and Table 60), 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 adults of these species, and 2) explosive activity level, which is the total number of explosives (by bin type) that would be detonated as part of the proposed action and overlap with areas occupied by ESA-listed fish species.

Below we discuss the anticipated overlap between ESA-listed fish species in the action area and HCTT explosive stressors. This qualitative analysis is based on available information on the

species' range, distribution, life history, and position in the water column, and the Action Proponent's proposed explosive activities.

Chinook Salmon

California Coastal, Central Valley spring-run, and Sacramento River winter-run ESUs of Chinook salmon could be exposed to sound and energy from explosives associated with HCTT activities in the NOCAL RC, and potentially the northernmost part of the PMSR. Overall, there are very few activities that utilize explosives in the NOCAL RC (see Table 31) compared to other locations, and the munitions used during these activities are considered small (E3 [> 0.5–2.5 lb. NEW] or below) (Navy 2025a). In addition, all explosives in the NOCAL RC will be conducted at least 12 nm from the closest point of land, which will avoid or reduce impacts on Chinook salmon, particularly juveniles, present in nearshore habitats. Although there are higher quantities of explosives used in the PMSR compared to the NOCAL RC (Navy 2025a), the densities of ESA-listed Chinook ESUs within the PMSR are anticipated to be very low, as this area represents the southern most extent of these populations' ranges. In addition, the larger detonations proposed for PMSR would all occur beyond 12 nm from shore (Navy 2025a) where densities of migrating Chinook salmon densities are expected to be very low as compared to more nearshore areas.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual Chinook salmon (Navy 2025a). Chinook salmon from the California Coastal, Central Valley spring-run, and Sacramento River winter-run ESUs that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. The Chinook salmon life stages most impacted will be adults and subadults, as juveniles generally prefer more nearshore areas where explosives will not be used. The number of ESA-listed Chinook salmon exposed to explosive stressors within a range that could result in such adverse effects is expected to be minimal due to the: 1) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 2) the relatively small NEWs of these explosives; and 3) the anticipated low densities of these ESA-listed ESUs in areas where explosives would be used.

Coho Salmon

Oregon Coast, Southern Oregon and Northern California Coast, and Central California Coast ESUs of coho salmon could be exposed to sound and energy from explosives associated with HCTT activities in the NOCAL RC. The coho salmon life stages most impacted will be adults and subadults, as juveniles generally prefer more nearshore areas where explosives will not be used. Although subadult and adult coho salmon may be exposed to detonations placed throughout the water column (i.e., near the surface to depths of 50 m), they are very surface oriented and, therefore, are more likely to be exposed to explosives detonated in the upper portion of the water column, or those at the water surface. Overall, there are very few activities that utilize explosives in the NOCAL RC (see Table 31) compared to other locations, and the

munitions used during these activities are considered small (E3 [> 0.5–2.5 lb. NEW] or below) (Navy 2025a). In addition, all explosives in the NOCAL RC will be conducted at least 12 nm from the closest point of land, which will avoid or reduce impacts on migrating coho salmon.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual coho salmon (Navy 2025a). Coho salmon from the Oregon Coast, Southern Oregon and Northern California Coast, and Central California Coast ESUs that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. The number of ESA-listed coho salmon exposed to explosive stressors within a range that could result in such adverse effects is expected to be minimal due to the: 1) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 2) the relatively small NEWs of these explosives; and 3) the anticipated low densities of these ESA-listed ESUs in areas where explosives would be used.

Steelhead

Northern California, California Central Valley, Central California Coast, South-Central California Coast, and Southern California DPSs of steelhead could be exposed to sound and energy from explosives associated with HCTT activities in the NOCAL RC. Because steelhead are highly surface oriented, they are most likely to be exposed to explosives detonated in the upper portion of the water column or at the water surface. Both outmigrating steelhead and adults returning to spawn are expected to occur in the action area. Given their offshore distribution, juvenile steelhead are also susceptible to the effects of HCTT explosive activities.

Overall, there are very few activities that utilize explosives in the NOCAL RC (see Table 31) compared to other locations, and the munitions used during these activities are considered small (E3 [> 0.5–2.5 lb. NEW] or below) (Navy 2025a). In addition, all explosives in the NOCAL RC will be conducted at least 12 nm from the closest point of land, which will avoid or reduce impacts on migrating steelhead. In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual steelhead (Navy 2025a). Steelhead from the Northern California, California Central Valley, Central California Coast, South-Central California Coast, and Southern California DPSs that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. The number of ESA-listed steelhead exposed to explosive stressors in the NOCAL RC within a range that could result in such adverse effects is expected to be minimal due to the: 1) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 2) the relatively small NEWs of these explosives; and 3) the anticipated low densities of these ESA-listed steelhead DPSs in areas where explosives would be used.

Southern California DPS could also be exposed to stressors from explosives detonated in the SOCAL RC and PMSR. Although there are higher quantities of explosives used in the SOCAL RC and PMSR (compared to the NOCAL RC), explosive activities in these areas are widely

dispersed in space and time, thus reducing the likelihood that explosions would co-occur with individual Southern California DPS steelhead (Navy 2025a). Furthermore, most of the explosive munitions used in this location are considered small (E5 [> 5–10 lb. NEW] or below) (Navy 2025a). Some steelhead could also be exposed to large detonations during activities such as oceanographic research (E7 [> 20–60 lb. NEW]) and Torpedo Testing (E8 [> 60–100 lb. NEW] or E11 [> 500–675 lb. NEW]). However, these larger detonations are typically used beyond 12 nm from shore, and are used much less often than smaller ones, reducing the potential for overlap with migrating steelhead (Navy 2025a).

Considering the populations' status and life history, densities of Southern California DPS steelhead are expected to be extremely low within the portion of the HCTT action area where overlap with explosive activities could occur. The majority of this DPSs life history occurs outside of the action area where it is not susceptible to potential effects from HCTT explosive use. Steelhead are thought to rely heavily on offshore marine waters for feeding, typically in northern latitudes, with high seas tagging programs indicating steelhead make more rapid, extensive migrations into offshore oceanic feeding grounds in their first year than any other Pacific salmonids (Quinn and Myers 2004; Daly et al. 2014). Because of this life history, we would not anticipate outmigrating Southern California DPS steelhead to spend extended period of time in portions of the SOCAL RC or PMSR where explosives are used. Instead, we anticipate outmigrating individuals will quickly move offshore and into northern latitudes, including NOCAL RC and north of the action area, to forage (Myers et al. 1996). This is where available information suggests steelhead from this DPS spend the majority of their lives while in marine environments, before returning to spawn. Adult Southern California DPS steelhead returning to spawn would pass through the Southern California Range Complex and PMSR and could be susceptible to effects from explosives if the animals were to co-occur with HCTT explosive stressors in space and time. However, due to the low number of steelhead that return to spawn in rivers in close proximity to the action area, the likelihood of an adult steelhead co-occurring with HCTT explosive stressors is very low. The available information on the abundance of Southern California DPS steelhead indicates that they are rare in the action area (see Section 4.2.19 for details). For this reason, and the intermittent nature of HCTT explosives in the action area, instances where Southern California DPS steelhead are exposed to explosive stressors would be rare. For all of these reasons, though information is not available to estimate the number of Southern California DPS steelhead that are likely to be injured or killed, we anticipate a very small percentage of the Southern California DPS to be exposed and adversely affected by HCTT explosive stressors. We anticipate that the large majority of Southern California DPS steelhead migrating through the action area would not co-occur with explosive stressors within the range to adverse effects as described in Table 60.

Eulachon

Southern DPS eulachon could be exposed to sound and energy from explosives associated with HCTT activities in the NOCAL RC. Eulachon are typically distributed in deeper coastal waters

and near benthic habitats in the open ocean at a wide range of depths (i.e., from 20 to 500 m) with an average depth around 150 m. Overall, there are very few activities that utilize explosives in the NOCAL RC (see Table 31) compared to other locations, and the munitions used during these activities are considered small (E3 [> 0.5–2.5 lb. NEW] or below) (Navy 2025a). In addition, all explosives in the NOCAL RC will be conducted at least 12 nm from the closest point of land, which will avoid or reduce impacts on migrating eulachon.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual eulachon (Navy 2025a). Southern DPS eulachon that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. The number of eulachon exposed to explosive stressors within a range that could result in such adverse effects is expected to be minimal due to the: 1) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 2) the relatively small NEWs of these explosives; 3) the anticipated low densities of eulachon in areas where explosives would be used; and 4) the preferred depths of eulachon, which minimize potential impacts from explosives detonated at or near the surface.

Oceanic Whitetip Sharks

Oceanic whitetip sharks could be exposed to sound and energy from HCTT explosives in the HRC, SOCAL RC, and the HCTT Transit Corridor. Oceanic whitetip sharks in deeper, offshore waters spend much of their time at the surface, potentially increasing the risk of exposure to surface detonations, though they could be exposed throughout the water column as they also frequent deep ocean waters.

The majority of the explosives used in the action area can be categorized in, or below, E5 (> 5 to 10 lb. NEW), with only occasional detonations of larger charge sizes (e.g., bins E8 and E11). These smaller bins produce smaller ranges to higher order effects such as mortality or injury compared to larger bin sizes, thus further reducing the potential that oceanic whitetip sharks would incur impacts that could lead to fitness consequences. Although some oceanic whitetip sharks may be present where Ship Shock Trials occur, this activity would only be conducted once over a seven-year period.

Both the SOCAL RC and the HRC are at the northern extent of the range of this species in the Pacific Ocean. For this reason, most oceanic whitetip sharks in the Pacific Ocean likely do not occur at any time within the action area and would not be susceptible to HCTT explosive use. Due to the dispersed, infrequent occurrence and short duration of explosives use throughout the SOCAL RC and Hawaii ranges, and the rarity of oceanic whitetip shark presence in these areas, exposure of this species to explosive stressors is not expected to be common. In addition, oceanic whitetip sharks do not posses a swim bladder, so are less susceptible to injury or mortality from explosives than fish species with swim bladders.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual oceanic whitetip

sharks (Navy 2025a). Oceanic whitetip sharks that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. However, for all of the reasons above, though information is not available to estimate the number of oceanic whitetip sharks that are likely to be injured or killed from HCTT explosives, we anticipate an extremely small percentage of the North Pacific population to be exposed and adversely affected by HCTT explosives stressors. We anticipate that a relatively small number of oceanic whitetip sharks in the action area would co-occur with explosive stressors within the range to adverse effects as described in Table 59.

Scalloped Hammerhead Sharks

Eastern Pacific DPS scalloped hammerhead sharks could be exposed to sound and energy from HCTT explosives in the HRC, SOCAL RC, and the HCTT Transit Corridor. Although most explosive activities are conducted offshore, some HCTT activities with explosives may also be conducted close to shore where scalloped hammerhead sharks could occur. This includes certain Mine Warfare and Expeditionary Warfare activities (Navy 2025a). In the HRC explosive activities could occur at specified ranges and designated locations around Oahu, including the Puuloa Underwater Range and designated locations in and near Pearl Harbor (Navy 2025a). Mitigation measures within the nearshore mitigation areas surrounding the Hawaiian Islands (Section 2.4.2) and for Mine Warfare activities involving Navy divers (Table 5) may also minimize impacts to scalloped hammerhead sharks from HCTT explosives. In the SOCAL RC, explosive activities could occur near San Clemente Island, in the Silver Strand Training Complex, and in other designated mine training areas along the southern California coast (Navy 2025a). However, sightings of scalloped hammerhead sharks in southern California are considered rare, in general. Although some scalloped hammerhead sharks may be present farther offshore where Ship Shock Trials occur, this activity would only be conducted once over a seven-year period.

The majority of the explosives used in the action area can be categorized in, or below, E5 (> 5 to 10 lb. NEW), with only occasional detonations of larger charge sizes (e.g., bins E8 and E11). These smaller bins produce smaller ranges to higher order effects such as mortality or injury compared to larger bin sizes, thus further reducing the potential that scalloped hammerhead sharks would incur impacts that could lead to fitness consequences. In addition, scalloped hammerhead sharks do not posses a swim bladder, so are less susceptible to injury or mortality from explosives than fish species with swim bladders.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual scalloped hammerhead sharks (Navy 2025a). Scalloped hammerheads that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. However, for all of the reasons above, though information is not available to estimate the number of scalloped hammerhead sharks that are likely to be injured or killed from HCTT explosives, we anticipate an extremely small percentage of the Eastern Pacific DPS to be exposed and adversely affected by HCTT explosives stressors. We anticipate that a relatively small number of sharks

from this DPS would co-occur with explosive stressors within the range to adverse effects as described in Table 59.

Giant Manta Ray

Giant manta rays could be exposed to sound and energy from HCTT explosives in the HRC and SOCAL RC. Giant manta rays have the potential to be exposed to detonations placed throughout the water column, including near the surface or on the seafloor. Although most explosive activities are conducted offshore, some HCTT activities with explosives may also be conducted close to shore where manta rays could occur. This includes certain Mine Warfare and Expeditionary Warfare activities (Navy 2025e). In the HRC explosive activities could occur at specified ranges and designated locations around Oahu, including the Puuloa Underwater Range and designated locations in and near Pearl Harbor (Navy 2025a).

Adult giant manta rays are typically found offshore but occasionally visit coastal areas where upwelling occurs. In particular, large aggregations of giant manta rays are known to occur along the Kona coast off the Big Island of Hawaii. Mitigation measures within the nearshore mitigation areas surrounding the Hawaiian Islands (Section 2.4.2) may minimize impacts to manta rays from HCTT explosives. The Action Proponents will not use explosives in the Hawaii Island Mitigation Area during training and testing activities. This area is inclusive of the locations where large aggregations of giant manta rays are known to occur.

In the SOCAL RC, explosive activities could occur near San Clemente Island, in the Silver Strand Training Complex, and in other designated mine training areas along the southern California coast (Navy 2025a). The likelihood of giant manta rays co-occurring with these activities would be limited as the SOCAL RC represents the northern edge of the specie's distribution in the Eastern Pacific.

The majority of the explosives used in the action area can be categorized in, or below, E5 (> 5 to 10 lb. NEW), with only occasional detonations of larger charge sizes (e.g., bins E8 and E11). These smaller bins produce smaller ranges to higher order effects such as mortality or injury compared to larger bin sizes, thus further reducing the potential that giant manta rays would incur impacts that could lead to fitness consequences. In addition, manta rays do not posses a swim bladder, so are less susceptible to injury or mortality from explosives than fish species with swim bladders.

In general, HCTT explosive activities are widely dispersed in space and time, which further reduces the likelihood that explosive stressors would co-occur with individual manta rays (Navy 2025a). Giant manta rays that are co-located with HCTT explosive activities may experience adverse effects including TTS, injury or mortality. However, due to the dispersed, infrequent occurrence and short duration of explosives use throughout the SOCAL RC and HRC, the rarity of giant manta rays in the SOCAL RC, and the Navy's mitigation in the HRC to avoid explosive use in areas around Hawaii Island where this species congregates, exposure of this species to explosive stressors is not expected to be common. Though information is not available to estimate the number of giant manta rays that are likely to be injured or killed from HCTT

explosives, we anticipate an extremely small percentage of the giant manta ray population within the action area to be exposed and adversely affected by HCTT explosives stressors. We anticipate that a relatively small number of giant manta rays from this DPS would co-occur with explosive stressors within the range to adverse effects as described in Table 59.

Response Analysis

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 thousands of 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 each seven-year period of HCTT training and testing. Some ranges will also vary depending upon the number of explosions in a single activity, depth and weight of the charge, etc. Fishes without a swim bladder, adult or fully developed fishes, and larger species are assumed to be generally less susceptible to injury and mortality from explosions compared to small, juvenile or larval fishes (Popper et al. 2014a). Other factors also influence the degree of sensitivity such as state of buoyancy, proximity to the blast (e.g., depth in the water, bodily alignment, etc.), and condition of the fish during the exposure event.

Explosive detonations can cause extensive damage to non-auditory tissues throughout a fish's body. This damage occurs through multiple mechanisms, including direct pressure effects and indirect trauma from rapid pressure changes. The intense pressure waves generated by underwater explosions can cause immediate tissue damage, particularly in gas-filled organs (Jenkins et al. 2022). These effects can manifest even at considerable distances from the explosion source, though their severity typically decreases with distance. The most immediate and severe physiological responses to explosive exposure occur in fish tissues and organs. Research has shown that swim bladders and kidneys are particularly vulnerable to damage from explosive pressure waves. In controlled studies, fish exposed to underwater explosions exhibited severe injuries to these organs, with damage patterns correlating directly to their proximity to the explosion source. Recent research documented that fish within 333 m of explosive detonations showed significant swim bladder and kidney damage, with peak sound pressure levels reaching 226 dB re: 1 µPa (Jenkins et al. 2022). These injuries often proved fatal, particularly in cases where fish were located within 157 m of the explosion site.

The relationship between explosive exposure and fish mortality follows distinct patterns based on several key factors. Notably, recent studies have revealed that fish size or mass does not significantly influence susceptibility to injury from explosions (Jenkins et al. 2022). Instead, peak pressure and SELs serve as more reliable predictors of injury severity. However, the overall size range of the fish in the study may not have varied enough to be able to detect a difference in susceptibility with differing weight or length (Jenkins et al. 2022).

For elasmobranch species, to date, no hearing loss has been demonstrated when exposed to other acoustic stressors such as air guns. Rather, the risk of it occurring for these species is much lower

than those fish that do possess swim bladders. Therefore, ranges for these species would likely be lower than what is calculated given the fact TTS has not been demonstrated at the thresholds, and the criteria for TTS is already based upon a very conservative value for more sensitive fish species with swim bladders. For fishes with swim bladders, the ESA-listed fish species that may be present in the action area do not have any hearing specializations, and do not have swim bladders involved in hearing. Similar to elasmobranchs, we are unaware of any research demonstrating TTS in these species 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. If TTS does occur, it would likely co-occur with barotraumas, and therefore would be within the range of other injuries these fishes are likely to experience from blast exposures. 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). Most TTS however, would likely be restored to normal hearing ranges within a few hours or days.

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. These effects, depending on the severity and duration could lead to fitness consequences such as reduced survival, growth, or reproductive capacity. 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 time, such as over the course of a day or consecutive days, fishes may also choose to avoid the area of disturbance. HCTT 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 in-water 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.

Vessel Strikes – Fishes

Vessel traffic during HCTT activities would be concentrated in certain portions of the action area such as areas near ports or naval installations and ranges (e.g., Pearl Harbor, San Diego, San Diego Bay, and San Clemente Island) and Coast Guard Stations, but could occur throughout the action area. Each of the ESA-listed fish species considered in this opinion are thought to spend at

least some time in the upper portions of the water column, where they may be susceptible to vessel strike. Oceanic whitetip sharks can be found at the ocean surface and down to at least 152 m deep, but most frequently stay between depths of 25.5 and 50 m (Carlson and Gulak 2012; Young et al. 2017). 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% of its time in the top 1 m of the water column, with few movements below 7 m. Scalloped hammerhead sharks may occur in the upper portions of the water column as well. Though tagging studies indicate giant manta rays are capable of descending to depths of hundreds of meters, they are also known to occur in surface waters and be susceptible to vessel strike (82Fed. Reg. 3694). Although subadult and adult coho salmon may be found at various depth throughout the water column (i.e., near the surface to depths of 50 m), they are typically very surface oriented.

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 fishes 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 160–490 ft (50–350 m). When the vessel passed over them, some fish responded with sudden escape responses with movement away from the vessel laterally or through downward compression of the school. Chapman and Hawkins (1973) observed avoidance responses of herring from the low-frequency sounds of large vessels or accelerating small vessels. Avoidance responses quickly ended within ten seconds after the vessel departed. Conversely, Rostad (2006) observed that some fish are attracted to different types of vessels (e.g., research vessels, commercial vessels) of varying sizes, noise levels, and habitat locations.

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). Potential implications of behavioral avoidance response to vessels was addressed in Section 4.1.1.

Based on the available information, we find that it is extremely unlikely (i.e., discountable) that an Action Proponent vessel will strike an ESA-listed fish species due to the following reasons: 1) the generally low abundance of the ESA-listed fish species in the action area, particularly around Navy and Coast Guard ports or installations where the majority of vessel traffic occurs; 2) the ability of these fish species to detect vessels from a distance and the agility to maneuver to avoid being struck by an oncoming vessel; 3) the low number of vessels associated with HCTT activities relative to non-military traffic in the area; and 4) the lack of documented cases of

military vessels striking these species (or any other fish species) in the action area. Therefore, we conclude that stressors from vessel strike as a result of the proposed action may affect, but are not likely to adversely affect, the following ESA-listed fish species: Southern DPS eulachon; Southern California DPS steelhead; California Central Valley DPS steelhead; Central California Coast DPS steelhead; Northern California DPS steelhead; South-Central California Coast DPS steelhead; California Coastal ESU Chinook; Central Valley Spring-Run ESU Chinook; Sacramento River Winter-Run ESU Chinook; Central California Coast ESU coho; Oregon Coast ESU coho; Southern Oregon and Northern California Coasts ESU coho; Eastern Pacific DPS scalloped hammerhead shark; oceanic whitetip shark; and giant manta ray.

6.4 Summary of Effects Determinations

Table 61. NMFS ESA effects determinations by stressor for each ESA-listed species and critical habitat, and overall effects determination for each species and critical habitat (LAA = likely to adversely affect; NLAA = not likely to adversely affect; NE = no effect)

ESA-Listed Resource	Overall Determination	Acoustic Stressors						Explosive Stressors	Energy Stressors	Physical Disturbance and Strike Stressors			Entanglement Stressors			Ingestion Stressors		Secondary Stressors: Effects to Habitat and Prey
		Sonar & Other Transducers	Air Guns	Pile Driving	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators / Parachutes	Nets Used for Obstacle Avoidance	Military Expended Materials – Munitions	Military Expended Materials – Other than Munitions	
Marine Invertebrates																		
Black abalone	**NLAA**	NLAA	NE	NE	NLAA	NE	NE	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA
White abalone	**NLAA**	NLAA	NE	NE	NLAA	NE	NE	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA
Sunflower sea star	**NLAA**	NLAA	NE	NE	NLAA	NE	NE	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA
Fishes																		
Scalloped hammerhead shark – Eastern Pacific DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Giant Manta Ray	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Oceanic Whitetip Shark	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead Trout – Southern California DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – California Central Valley DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Central California Coast DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Northern California DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – South-Central California Coast DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – California Coastal ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chinook salmon – Central Valley Spring-Run ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Resource	Overall Determination	Acoustic Stressors						Explosive Stressors	Energy Stressors	Physical Disturbance and Strike Stressors			Entanglement Stressors			Ingestion Stressors		Secondary Stressors: Effects to Habitat and Prey
		Sonar & Other Transducers	Air Guns	Pile Driving	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators / Parachutes	Nets Used for Obstacle Avoidance	Military Expended Materials – Munitions	Military Expended Materials – Other than Munitions	
Chinook salmon Sacramento River Winter-Run ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Central California Coast ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Oregon Coast ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Southern Oregon and Northern California Coasts ESU	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Green Sturgeon – Southern DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Eulachon – Southern DPS	**LAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sea Turtles																		
Green turtle – Central North Pacific DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA
Green turtle - Eastern Pacific DPS	**LAA**	LAA	LAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Hawksbill turtle	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA
Leatherback turtle	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Loggerhead turtle – North Pacific DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Olive ridley turtle	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Marine Mammals																		
Killer whale – Southern Resident DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
North Pacific Right Whale	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Blue whale	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Fin whale	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

ESA-Listed Resource	Overall Determination	Acoustic Stressors						Explosive Stressors	Energy Stressors	Physical Disturbance and Strike Stressors			Entanglement Stressors			Ingestion Stressors		Secondary Stressors: Effects to Habitat and Prey
		Sonar & Other Transducers	Air Guns	Pile Driving	Vessel Noise	Aircraft Noise	Weapons Noise	Explosions	In-Water Electromagnetic Devices	Vessels & In-Water Devices	Military Expended Material	Seafloor Devices	Wires & Cables	Decelerators / Parachutes	Nets Used for Obstacle Avoidance	Military Expended Materials – Munitions	Military Expended Materials – Other than Munitions	
Gray whale - Western North Pacific DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale - Mexico DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale – Central America DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sei whale	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sperm whale	**LAA**	LAA	LAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
False killer whale - Main Hawaiian Islands Insular DPS	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA
Guadalupe fur seal	**LAA**	LAA	LAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Hawaiian monk seal	**LAA**	LAA	NLAA	NE	NLAA	NLAA	NLAA	LAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA
Critical Habitat																		
Black abalone	**NLAA**	NE	NE	NE	NE	NE	NE	NLAA	NE	NLAA	NLAA	NLAA	NE	NLAA	NE	NE	NE	NLAA
Green Sturgeon- Southern DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NE	NE	NE	NLAA
Leatherback Sea Turtle	**NLAA**	NLAA	NLAA	NE	NE	NE	NE	NLAA	NE	NLAA	NLAA	NE	NE	NE	NE	NLAA	NLAA	NLAA
Killer whale – Southern Resident DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Humpback Whale – Central America DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Humpback Whale – Mexico DPS	**NLAA**	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
False killer whale - Main Hawaiian Islands Insular DPS	**LAA**	LAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NE	NE	NE	NE	NE	NE	NE	NLAA
Hawaiian Monk Seal	**NLAA**	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NE	NE	NE	NLAA

7 CUMULATIVE EFFECTS

"Cumulative effects" are defined in the regulations as "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 CFR §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(a)(2) of the ESA.

We assessed the action area of this consultation for any non-Federal activities that are reasonably certain to occur. The past and ongoing impact of existing actions was described in the Environmental Baseline (Section 5). During this consultation, we searched for information on future state, tribal, local, or private (non-Federal) actions reasonably certain to occur in the action area. We did not find any information about non-Federal actions other than what has already been described in the Environmental Baseline, most of which we expect will continue in the future. An increase in these activities could similarly increase their effects on ESA-protected resources and for some, an increase in the future is considered reasonably certain to occur. Given current trends in global population growth, threats associated with changing environmental conditions, ocean pollution, fisheries bycatch, vessel strike, offshore wind development, and anthropogenic sound are likely to continue to increase in the future, although any increase in effects may be somewhat countered by an increase in conservation and management activities. In contrast, more historic threats such as whaling and sea turtle harvest are likely to remain low or potentially decrease. For the remaining 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 ESA-protected resources. Thus, this consultation assumes future effects will be similar to those in the past and, therefore, are reflected in the anticipated trends described in the Status of the Species and Critical Habitat Likely to be Adversely Affected (Section 4.2.1) and Environmental Baseline (Section 5) of this opinion.

8 INTEGRATION AND SYNTHESIS

This opinion includes a jeopardy analysis for the ESA-listed threatened and endangered species and a destruction or adverse modification analysis for designated critical habitat that are likely to be adversely affected by the action. Section 7(a)(2) of the Act and its implementing regulations require every Federal agency, in consultation with and with the assistance of the Secretary (16 U.S.C. §1532(15)), to insure that any action it authorizes, funds, or carries out, in whole or in part, in the United States or upon the high seas, is not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. The jeopardy analysis, therefore, relies upon the regulatory definitions of *jeopardize the continued existence of* and *destruction or adverse modification*.

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 a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR §402.02). Recovery, used in that definition, means "improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in section 4(a)(1) of the Act" (50 CFR §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 a listed species" (50 CFR §402.02). Conservation, used in that definition, means "to use and the use of all methods and procedures which are necessary to bring any endangered species or threatened species to the point at which the measures provided pursuant to this Act are no longer necessary" (16 U.S.C. §1532(3)).

The Integration and Synthesis is the final step in our jeopardy and adverse modification analyses. In this section, we add the effects of the action (Section 6) to the environmental baseline (Section 5) and the cumulative effects (Section 7), taking into account the status of the species, critical habitat, and recovery planning (Section 4), to formulate the agency's biological opinion as to whether the Action Proponents can insure their proposed actions are not likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) appreciably diminish the value of designated critical habitat as a whole for the conservation of the species.

8.1 Jeopardy Analysis

The jeopardy analysis assesses the proposed actions' effects to ESA-listed species' survival and recovery. The following sections summarize the relevant information in this opinion for each individual species considered.

8.1.1 Blue Whale

The current blue whale global mature population size is uncertain, but estimated to be in the range of 5,000-15,000 mature individuals (NMFS 2020d). The best abundance estimate for blue whales in the Hawaii portion of the action area is 133 animals (Bradford et al. 2017) and the best abundance estimate in the California portion of the action area (i.e., CA/OR/WA stock) is 1,898 animals (Calambokidis and Barlow 2020). Current estimates indicate a growth rate of around 3-4% per year for the eastern North Pacific stock (Calambokidis et al. 2009b) (Carretta 2021b).

Blue whales are only present in the offshore portion of the HCTT action area and may be exposed to acoustic stressors associated with training and testing activities throughout the year. Blue whales are expected to experience AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives. The large majority of these impacts will occur to blue whales from the Eastern North Pacific Stock in the California portion of the action area, although impacts will also occur to blue whales from the Central North Pacific Stock off of Hawaii. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 1,464

instances of behavioral harassment, 3,199 instances of TTS, and 28 instances of AINJ are reasonably certain to occur to blue whales annually, in a maximum year of HCTT sonar and explosive activities. Based on the best available information on the exposure of blue whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no non-auditory injury or mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of blue whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to 28 instances of blue whale AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some blue whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. A small number of blue whales could experience moderate to high severity PTS, most likely from exposure to broadband sound sources (e.g., explosives), since there is an increased chance that the resulting hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of blue whales that could experience fitness consequences from PTS is relatively small (i.e., likely less than three per year), compared to the overall blue whale population size, we do not anticipate population level impacts as a result. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of blue whales in the North Pacific Ocean or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in blue whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Blue whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for blue whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency blue whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably impacts to the fitness of individual blue whales. From our acoustic effects analysis, blue whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer

lasting compared to pulsed active sonar (e.g., MF1). While masking from high duty cycle or continuous active sonar could lead to longer-term behavioral disruptions affecting blue whale communication and social interactions, low-frequency cetaceans (including blue whales) are likely less susceptible to masking effects, given the limited overlap in vocalization frequency with these military sonar sources, which typically operate at higher frequencies (i.e., mid-frequency and high-frequency sonars).

From our vessel strike analysis (Section 6.1.3), we anticipate up to two (one Navy and one Coast Guard) blue whale lethal vessel strikes will occur within the California portion of the action area as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. We anticipate that the death of a male blue whale would have substantially less of an effect on the population than the loss of a female. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years. Best available information suggests the rangewide blue whale population is at least 5,000 individuals and the best estimate of current blue whale abundance for CA/OR/WA waters is 1,898 whales (Calambokidis and Barlow 2013; Calambokidis and Barlow 2020). Assuming a balanced sex ratio, this means 2,500 females in the range-wide population and 949 females in the CA/OR/WA population. If the one blue whale that is expected to be struck over seven years is a female of early reproductive age, this would reduce the reproductive potential of the range-wide population by 0.08% and the CA/OR/WA population by 0.11% (note: this is considered an overestimate since some blue whale vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or reproductive capability of the CA/OR/WA stock or the rangewide population. If the analyzed rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes would not affect the current geographic range of blue whales and no reduction in the distribution of this species is expected as a result of the action.

The 2020 blue whale recovery plan outlines downlisting and delisting criteria, which include increasing blue whale resiliency and ensuring geographic and ecological representation, minimizing anthropogenic effects, and ensuring anthropogenic activities are not contributing to the species being in danger of extinction within the foreseeable future throughout all or a significant portion of its range. The recovery plan lists several stressors potentially affecting the status of blue whales in the North Pacific Ocean that are relevant to HCTT activities including vessel strike, vessel disturbance, and military operations. No appreciable reduction in the number of blue whales is expected to occur from the action. While military readiness activities have been conducted in the HCTT action area for decades, Monnahan et al. (2014) suggested that the blue

whale population in the Eastern North Pacific, inclusive of the SOCAL RC portion of the action area, is at carrying capacity and recovered to pre-whaling levels. Current estimates indicate a growth rate of around 3-4% per year for the eastern North Pacific stock (Calambokidis et al. 2009a; Carretta 2022). This suggests that blue whales are likely somewhat resilient to the impacts incurred from HCTT activities in the action area. In addition, the Action Proponents will continue to implement restrictions on sonar and explosive use in some areas within the SOCAL RC identified as BIA's for blue whales (Calambokidis et al. 2015), thereby reducing potential effects of HCTT activities to this species in locations where we know they are likely to be foraging.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the 7-year MMPA rule period, 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. Similarly, the effects from ongoing HCTT 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. Therefore, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of blue whales in the wild.

8.1.2 Fin Whale

There are an estimated 10,000 fin whales in U.S. Pacific Ocean waters (Nadeem et al. 2016). Three fin whale stocks have been identified in U.S. Pacific Ocean waters: Northeast Pacific (minimum 1,368 individuals), Hawaii (approximately 203 individuals, N_{min} = 101) and California/Oregon/Washington (approximately 11,065 individuals, N_{min} = 7,970) (Becker et al. 2020; Bradford et al. 2021; Carretta et al. 2023b). There is now strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 (Nadeem et al. 2016; Becker et al. 2020; Carretta 2022). Estimated annual rates of increase in the California Current averaged 7.5% from 1991-2014 (Nadeem et al. 2016). However, it is unknown how much of this growth is due to immigration rather than birth and death processes (Carretta 2022).

Fin whales are expected to experience non-auditory injury, AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives. The large majority of these impacts will occur to fin whales from the California, Oregon, and Washington Stock in the California portion of the action area, although a small number of fin whale impacts will also occur off of Hawaii. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 3,721 instances of behavioral harassment, 9,862 instances of TTS, and 55 instances of AINJ are reasonably certain to occur to fin whales annually, in a maximum year of HCTT sonar and explosive activities. The quantitative analysis also predicts one fin whale non-auditory injury

from Navy explosives training every seven years of the proposed action. Based on the best available information on the exposure of fin whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of fin whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to 55 instances of fin whale AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some fin whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. Although we anticipate that a large majority of PTS exposures will be of mild severity, some fin whales will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of fin whales that could experience long-term fitness consequences from PTS is relatively small (i.e., likely less than 11 per year), compared to the overall fin whale population size, we do not anticipate population level impacts as a result. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of fin whales in the North Pacific Ocean or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in fin whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Fin whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for fin whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency fin whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably impacts to the fitness of individual fin whales. From our acoustic effects analysis, fin whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1). While masking from high duty cycle or continuous active sonar could lead to longer-term behavioral disruptions affecting fin whale communication and social

interactions, low-frequency cetaceans (including fin whales) are likely less susceptible to masking effects, given the limited overlap in vocalization frequency with these military sonar sources, which typically operate at higher frequencies (i.e., mid-frequency and high-frequency sonars).

As discussed above (Section 6.1.1), we expect that fin whales experiencing a non-auditory injury would be temporarily impaired as a result. While recovering from its injury, we assume the animal's ability to conduct important life functions (e.g., breeding, feeding) would be diminished, but that the animal would survive. For a slight lung injury, which defines the threshold for onset of non-auditory injury, it is reasonable to assume the whale would recover within several months. A more severe injury could take longer until full recovery, which could result in a decrease in the animal's reproductive potential, particularly if it occurs to a mature female. However, given their long life spans, it is unlikely that a single injury would result in a significant impact on the life-time reproductive potential of an individual fin whale. Since non-auditory injuries are expected to occur rarely (i.e., one every seven years), it is extremely unlikely that the same individual fin whale will be exposed to such an injury more than once as a result of the HCTT proposed action.

From our vessel strike analysis (Section 6.1.3), we anticipate up to six (five Navy and one Coast Guard) fin whale lethal vessel strikes to occur within the California portion of the action area as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual with the loss of a female having the potential to be more consequential to the population. Best available information suggests the fin whale population occurring in the California portion of the action area consists of 11,065 animals. Assuming a balanced sex ratio, this means there are about 5,532 females in the population. If all six fin whales that are expected to be struck by Action Proponent vessels are females of early reproductive age, this would reduce the reproductive potential of the eastern north Pacific population by approximately 0.11% over seven years (note: this is considered an overestimate since some fin whale vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or the reproductive capability of fin whales in the California portion of the action area. The impact of this level of vessel strike would be even less consequential for the range-wide population. If the analyzed rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes would not affect the current geographic range of fin whales and no reduction in the distribution of this species is expected as a result of the action.

The 2010 fin whale recovery plan (NMFS 2010b) includes the following downlisting criteria: 1) Factors that may limit population growth have been identified and are being or have been addressed to the extent that they allow for continued growth of populations in each ocean basin

in which it occurs (North Atlantic, North Pacific and Southern Hemisphere), and 2) The population in each ocean basin satisfies the risk analysis standard for threatened status (has no more than a 1% chance of extinction in 100 years) and has at least 500 mature, reproductive individuals (consisting of at least 250 mature females and at least 250 mature males). In the latest 5-year review (NMFS 2019e), NMFS recommended that the fin whale be downlisted from endangered to threatened based on meeting the downlisting criteria. As discussed above, we do not anticipate that the threats to fin whales (i.e., non-auditory injury, AINJ, TTS, behavioral response, and physiological stress from acoustic stressors, and vessel strike) resulting from the proposed action would appreciably limit the growth of the North Pacific population.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the 7-year MMPA rule period, 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. Similarly, the effects from ongoing HCTT 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. Therefore, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of fin whales in the wild.

8.1.3 Gray Whale – Western North Pacific DPS

The minimum population size estimate for the Western North Pacific stock, based on Cooke et al. (2017), is 271 animals (Carretta 2021b). The combined Sakhalin Island and Kamchatka populations were estimated to be increasing from 2005 through 2016 at an average rate between 2-5% annually (Cooke et al. 2017). Only a subset of the population would be expected to occur in the action area (i.e., the subset of individuals that winters off the U.S. west coast). Since the majority of this DPS' range is outside of the action area, for the subset of individual whales that do migrate to the U.S. West Coast, we anticipate the amount of time spent within the action area is relatively small, thus reducing the likelihood of exposure to HCTT stressors.

Western North Pacific DPS gray whales are expected to experience AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 72 instances of behavioral harassment, 97 instances of TTS, and two instances of AINJ are reasonably certain to occur to this DPS annually, in a maximum year of HCTT sonar and explosive activities. Based on the best available information on the exposure of fin whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no non-auditory injury or mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of

Western North Pacific DPS gray whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to two instances of Western North Pacific DPS gray whale AINJ annually, but, as discussed in more detail in Section 6.1.1, the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some Western North Pacific DPS gray whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Western North Pacific DPS gray whales in the action area or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in gray whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Gray whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for gray whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency gray whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably impacts to the fitness of individual gray whales.

Based on the best available information on the exposure of Western North Pacific DPS gray whales to vessel strike (See Section 6.1.3), no injury or mortality of this species is reasonably certain to occur from this stressor.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the 7-year MMPA rule period, 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 Western North Pacific DPS gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 Western North Pacific DPS gray whales in the wild. Therefore, we determine that

the proposed action would not appreciably reduce the likelihood of both survival and recovery of Western North Pacific DPS gray whales in the wild.

8.1.4 Humpback Whale – Mexico DPS

There is currently no abundance estimate for the humpback whale Mexico DPS, although an estimated 3,477 whales from this DPS feed off the U.S. West Coast (Calambokidis and Barlow 2020; Curtis et al. 2022). Curtis et al. (2022) derived an annual population growth rate for the Central America / Southern Mexico humpback whale stock of 1.6% (Carretta et al. 2023a). By comparison, the growth rate of the California/Oregon/Washington stock, which is comprised of both Mexico and Central America DPS whales, is estimated to be around 8% since the late 1980s (Calambokidis and Barlow 2020).

Mexico DPS humpback whales are expected to experience non-auditory injury, AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 1,274 instances of behavioral harassment, 3,175 instances of TTS, and 43 instances of AINJ are reasonably certain to occur to whales from this DPS annually, in a maximum year of HCTT sonar and explosive activities. The quantitative analysis also predicts one Mexico DPS humpback whale non-auditory injury from Navy explosives testing every seven years of the proposed action. Based on the best available information on the exposure of Mexico DPS humpback whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of Mexico DPS humpback whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to 43 instances of Mexico DPS humpback whale AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. Although we anticipate that a large majority of PTS exposures will be of mild severity, some Mexico DPS humpback whales will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized

by animals for acoustic cues. However, since the number of Mexico DPS humpback whales that could experience long-term fitness consequences from PTS is relatively small (i.e., likely less than four per year), compared to the overall DPS size, we do not anticipate population level impacts as a result. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Mexico DPS humpback whales in the action area or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in Mexico DPS humpback whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Humpback whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for humpback whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency humpback whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciable fitness impacts. From our acoustic effects analysis, Mexico DPS humpback whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1). While masking from high duty cycle or continuous active sonar could lead to longer-term behavioral disruptions affecting whale communication and social interactions, low-frequency cetaceans (including humpbacks) are likely less susceptible to masking effects, given the limited overlap in vocalization frequency with these military sonar sources, which typically operate at higher frequencies (i.e., mid-frequency and high-frequency sonars).

As discussed above (Section 6.1.1), we expect that humpback whales experiencing a non-auditory injury would be temporarily impaired as a result. While recovering from its injury, we assume the animal's ability to conduct important life functions (e.g., breeding, feeding) would be diminished, but that the animal would survive. For a slight lung injury, which defines the threshold for onset of non-auditory injury, it is reasonable to assume the whale would recover within several months. A more severe injury could take longer until full recovery, which could result in a decrease in the animal's reproductive potential, particularly if it occurs to a mature female. However, given their long life spans, it is unlikely that a single injury would result in a significant impact on the life-time reproductive potential of an individual humpback whale. Since non-auditory injuries are expected to occur rarely (i.e., one every seven years), it is extremely unlikely that the same individual humpback whale will be exposed to such an injury more than once as a result of the HCTT proposed action.

From our vessel strike analysis (Section 6.1.3), we anticipate up to two (one Navy and one Coast Guard) Mexico DPS humpback whale lethal vessel strikes to occur within the California portion of the action area as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual with the loss of a female having the potential to

be more consequential to the population. Best available information suggests that an estimated 3,477 humpback whales from the Mexico DPS feed off the U.S. West Coast (Calambokidis and Barlow 2020; Curtis et al. 2022). Assuming a balanced sex ratio, this means 1,738 females in the population. If the two Mexico DPS humpback whales that are expected to be struck over seven years are both females of early reproductive age, this would reduce the reproductive potential of the Mexico DPS by about 0.06% (note: this is considered an overestimate since some humpback whale vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or the reproductive capability of Mexico DPS humpback whales. If the anticipated rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the current geographic range of Mexico DPS humpback whales and no reduction in the distribution of this species is expected as a result of the action.

Stated goals of the 2022 NMFS recovery outline for Mexico DPS humpback whales include management activities that continue to protect humpback whales and their critical habitat by reducing medium and high risk threats to the population (NMFS 2022d). Of the threats cited in the recovery outline as medium to high risk, only vessel strike is relevant to HCTT activities. As discussed above, the small number of Mexico DPS humpback whale vessel strikes anticipated as a result of the action would not appreciably reduce the species' ability to recover.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the 7-year MMPA rule period, 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 Mexico DPS humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 Mexico DPS humpback whales in the wild. Therefore, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Mexico DPS humpback whales in the wild.

8.1.5 Humpback Whale – Central America DPS

The estimated abundance of the Central America DPS is 1,494 whales (Curtis et al. 2022), with an estimated annual growth rate of 1.8% (Carretta 2022). The estimated growth rate of the California, Oregon, Washington stock, which includes both Mexico and Central America DPS whales, since the late 1980s is around 8% (Calambokidis and Barlow 2020).

Central America DPS humpback whales are expected to experience AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 547 instances of behavioral harassment, 1,341 instances of TTS, and 19 instances of AINJ are reasonably certain to occur to whales from this DPS annually, in a maximum year of HCTT sonar and explosive activities. Based on the best available information on the exposure of Central America DPS humpback whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no non-auditory injury or mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of Central America DPS humpback whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to 19 instances of Central America DPS humpback whale AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. Although we anticipate that a large majority of PTS exposures will be of mild severity, some Central America DPS humpback whales will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of Central America DPS humpback whales that could experience long-term fitness consequences from PTS is relatively small (i.e., likely less than two per year), compared to the overall DPS size, we do not anticipate population level impacts as a result. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Central America DPS humpback whales in the action area or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in Central America DPS humpback whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Humpback whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for humpback whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency

humpback whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably fitness impacts. From our acoustic effects analysis, Central America DPS humpback whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1). While masking from high duty cycle or continuous active sonar could lead to longer-term behavioral disruptions affecting whale communication and social interactions, low-frequency cetaceans (including humpbacks) are likely less susceptible to masking effects, given the limited overlap in vocalization frequency with these military sonar sources, which typically operate at higher frequencies (i.e., mid-frequency and high-frequency sonars).

From our vessel strike analysis (Section 6.1.3), we anticipate up to two (one Navy and one Coast Guard) Central America DPS humpback whale lethal vessel strikes to occur within the California portion of the action area as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual with the loss of a female having the potential to be more consequential to the population. The estimated abundance of the Central America DPS is 1,494 whales (Calambokidis and Barlow 2020; Curtis et al. 2022). Assuming a balanced sex ratio, this means 747 females in the population. If the two Central America DPS humpback whales that are expected to be struck over seven years are both females of early reproductive age, this would reduce the reproductive potential of the Central America DPS by about 0.27% every seven years (note: this is considered an overestimate since some humpback whale vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or the reproductive capability of Central America DPS humpback whales. If the anticipated rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the current geographic range of Central America DPS humpback whales and no reduction in the distribution of this species is expected as a result of the action.

Goals of the 2022 NMFS recovery outline for Central America DPS humpback whales include management activities that continue to protect humpback whales and their critical habitat by reducing medium and high risk threats to the population (NMFS 2022d). Of the threats cited in the recovery outline as medium to high risk, only vessel strike is relevant to HCTT activities. As discussed above, the small number of Central America DPS humpback whale vessel strikes anticipated as a result of the action would not appreciably reduce the species' ability to recover. Despite ongoing threats to this DPS from vessel strike (military and nonmilitary) and other stressors, the estimated annual growth rate of this DPS is 1.8% (Carretta 2022).

In summary, 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 HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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 humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the 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 Central America DPS humpback whales in the wild. Thus we determine that the action would not appreciably reduce the likelihood of both survival and recovery of Central American DPS humpback whales in the wild.

8.1.6 Sei Whale

The most recent abundance estimate for sei whales in the North Pacific Ocean is about 35,000 whales (NMFS 2021g). Within the action area, the best abundance estimates for sei whales are 864 whales of the U.S. West Coast (Carretta et al. 2025), and 391 whales in the Hawaii sei whale stock (Bradford et al. 2017). There are no estimates of the growth rate or population trends of sei whale populations in the North Pacific or particular to the action area.

Sei whales are expected to experience AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the HRC and the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 121 instances of behavioral harassment, 434 instances of TTS, and five instances of AINJ are reasonably certain to occur to sei whales annually, in a maximum year of HCTT sonar and explosive activities. Based on the best available information on the exposure of sei whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no non-auditory injury or mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of sei whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to five instances sei whale AINJ annually, but, as discussed in more detail in Section 6.1.1, the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate

conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in sei whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Sei whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking would be limited for sei whales since it could only occur if there is overlap between the sonar sound source frequencies and the low frequency sei whale vocalizations. In addition, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably fitness impacts. From our acoustic effects analysis, sei whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1). While masking from high duty cycle or continuous active sonar could lead to longer-term behavioral disruptions affecting whale communication and social interactions, low-frequency cetaceans (including sei whales) are likely less susceptible to masking effects, given the limited overlap in vocalization frequency with these military sonar sources, which typically operate at higher frequencies (i.e., mid-frequency and high-frequency sonars).

From our vessel strike analysis (Section 6.1.3), we anticipate up to two (one Navy and one Coast Guard) sei whale lethal vessel strikes to occur as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. We anticipate that the death of a male sei whale would have substantially less of an effect on the population than the loss of a female. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years. Our best available estimate of the total number of sei whales in the North Pacific Ocean is about 35,000 (Hakamada and Matsuoka 2016; NMFS 2021g). Assuming a balanced sex ratio, this means there are about 17,500 female sei whales in the North Pacific Ocean. If the two sei whales that are expected to be struck over seven years are both females of early reproductive age, this would reduce the reproductive potential of this population of the North Pacific Ocean population by about 0.01% (note: this is considered an overestimate since some sei vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or the reproductive capability of sei whales in the North Pacific Ocean or range wide population. If the analyzed rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes would not affect the current geographic range of sei whales and no reduction in the distribution of this species is expected as a result of the action.

Based on the 2011 sei whale recovery plan (NMFS 2011), downlisting criteria (i.e., from endangered to threatened) include: populations in each ocean basin (North Atlantic, North Pacific and Southern Hemisphere) that have no more than a 1% chance of extinction in 100 years; the global population has at least 1,500 mature, reproductive individuals, consisting of at least 250 mature females and at least 250 mature males in each ocean basin; and none of the known threats to sei whales are known to limit the continued growth of populations. For delisting consideration, in addition to the above, the total sei whale abundance in each ocean basin must have less than a 10% probability of becoming endangered in 20 years. As discussed above, we do not anticipate that the threats to sei whales (i.e., vessel strike, TTS, behavioral response, and physiological stress from acoustic stressors) resulting from the action would appreciably limit the growth of the North Pacific population.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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. Similarly, the effects from ongoing HCTT 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 sei whales in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of sei whales in the wild.

8.1.7 Sperm Whale

The current global sperm whale estimate is about 850,000 whales (Whitehead and Shin 2022). Population estimates for the two U.S. stocks within the action area are 2,606 whales for the California/Oregon/ Washington stock (Becker et al. 2020) and 5,387 for the Hawaii stock (Becker et al. 2021). Population growth rates for this species or individual stocks are not available.

Sperm whales are expected to experience AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar, explosives, and air guns in the HRC and the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 4,236 instances of behavioral harassment, 1,304 instances of TTS, and four instances of AINJ are reasonably certain to occur to sperm whales annually, in a maximum year of HCTT activities. Based on the best available information on the exposure of sperm whales to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no non-auditory injury or mortality of this species is reasonably certain to occur from these stressors. As described in greater detail in Section 6.1.1, we do not anticipate these instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of sperm whales. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to four instances of sperm whale AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some whales are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual whales affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. Although we anticipate that a large majority of PTS exposures will be of mild severity, some sperm whales will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of sperm whales that could experience long-term fitness consequences from PTS is relatively small (i.e., less than one per year), compared to the overall sperm whale population size, we do not anticipate population level impacts as a result. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of sperm whales in the North Pacific Ocean or range wide.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in sperm whales given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Sperm whales exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). Odontocete species (including sperm whales), which rely on echolocation to find prey, are more susceptible to masking vocalization given the overlap in vocalization frequency with these sonar systems. However, since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciably fitness impacts. From our acoustic effects analysis, sperm whales will likely be exposed to high duty cycle or continuous active sonar (i.e., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1). Masking from high duty cycle or continuous active sonar with frequency ranges that overlap sperm whale vocalizations could lead to longer-term behavioral disruptions, affecting sperm whale foraging and communication. However, even with high duty cycle and continuous sonar sources, the duration of masking would be limited since both the sonar source and the whale are moving within the environment, and we do not anticipate whales would remain in close proximity to a source for an extended period of time. While high duty cycle and continuous active sonars transmit more frequently than traditional sonars, they typically operate at a substantially lower source level. Although the Action Proponents have proposed an increase in the use of high duty cycle and continuous active sonar from Phase III, the large majority of active sonar for HCTT Phase IV is pulsed sonar with low duty cycles (see

Table 50). In addition, most sonar sources used by the Action Proponents operate over a narrow band of frequencies which only overlap a small portion of the full range of sperm whale hearing, although some activities involving high duty cycle and continuous sonar also use broadband sonar sources (e.g., anti-submarine warfare training). Based on the estimated population sizes within the action area and NAEMO impacts analysis (see Table 44 and Table 45), we expect that individual sperm whales within the action area will be adversely affected by HCTT acoustic stressors less than once per year, on average (i.e., some individuals may be impacted more than others). Therefore, while masking may result in fitness consequences (including potential impacts on foraging efficiency), the duration, magnitude and scope of these impacts on individual sperm whales will be limited, and are not expected to result in impacts to reproduction or survival of sperm whales at the population level.

From our vessel strike analysis (Section 6.1.3), we anticipate up to one Navy sperm whale lethal vessel strike to occur as a result of the proposed action every seven years into the foreseeable future. Death would result in lost reproductive potential that the individual might have contributed to the population or subpopulation. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. We anticipate that the death of a male sperm whale would have substantially less of an effect on the population than the loss of a female. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years. Our best available estimate of the total number of sperm whales in the two stocks that occur within the action area is about 7,813 (Carretta et al. 2021). Assuming a balanced sex ratio, this means there are about 3,907 female sperm whales in these two stocks. If the one sperm whale that is expected to be struck over seven years is a female of early reproductive age, this would reduce the reproductive potential of the sperm whale stocks in the action area by 0.03% (note: this is considered an overestimate since some sperm whale vessel strikes will likely be of males). This is not an appreciable reduction in the numbers or the reproductive capability of sperm whales in the action area or the much larger global population. If the analyzed rate of vessel strike for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential resulting from vessel strikes would not affect the current geographic range of sperm whales and no reduction in the distribution of this species is expected as a result of the action.

The 2010 sperm whale recovery plan (NMFS 2010a) 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, changing environmental conditions, direct harvest, oil spills, anthropogenic noise, and vessel strike. Of these, anthropogenic noise and vessel strike are relevant to HCTT activities. 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. No appreciable reduction in the number of sperm whales is expected to occur from HCTT activities.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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. Similarly, the effects from ongoing HCTT 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. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of sperm whales in the wild.

8.1.8 Guadalupe Fur Seal

The best estimate of Guadalupe fur seal abundance is the Juárez-Ruiz et al. (2022) 2019 estimate of 63,850 animals (Carretta et al. 2025). The annual estimated growth rate for the 1991–2019 period is 8.4%, including a steep increase in pup production from 2014-2019 (Lotze et al. 2011; Juárez-Ruiz et al. 2022). However, since about 2010, a number of potential impacts to the population have occurred, including dramatic shifts in species distribution and abundance, and the mass strandings events along the U.S. West Coast as described in Section 6.3.8 (McCue 2021). Since Guadalupe fur seals rely on one major rookery where greater than 99% of the pups are born, they may be especially vulnerable to threats if they were to occur at this rookery. The Guadalupe fur seal does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established for this species.

Guadalupe fur seals are expected to experience mortality, non-auditory injury, AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar, explosives, and air guns in the California portion of the action area. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 266,199 instances of behavioral harassment, 81,354 instances of TTS, 54 instances of AINJ, and three non-auditory injuries are reasonably certain to occur to this species annually, in a maximum year of HCTT activities. In addition, Navy explosive training activities are expected to result in an estimated one Guadalupe fur seal mortality every seven years of the proposed action. Based on the best available information on the exposure of Guadalupe fur seals to HCTT vessel strike (See Section 6.1.3), no injury or mortality of this species is reasonably certain to occur from this stressor.

As described in greater detail in Section 6.1.1, we do not anticipate instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of Guadalupe fur seals. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term. Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action

area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to 54 instances of Guadalupe fur seal AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some Guadalupe fur seals are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual seals affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Guadalupe fur seals in the action area or range wide.

Although we anticipate that a large majority of PTS exposures will be of mild severity, some Guadalupe fur seals will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of fur seals that could experience long-term fitness consequences from PTS is relatively small (i.e., likely less than 16 per year), compared to the overall Guadalupe fur seals population size, we do not anticipate population level impacts as a result. In addition, compared to cetaceans, fur seals are not known to rely as heavily on sound for important biological functions, although PTS could affect Guadalupe fur seals communication with conspecifics. For these reasons, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Guadalupe fur seals in the action area or throughout their range.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in Guadalupe fur seals given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Guadalupe fur seals exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking effects would be limited for Guadalupe fur seals since the effects only occur in the presence of the masking noise and this species is not known to rely heavily on hearing or sound production for important biological functions. In most instances masking would be minor and short-term, with no appreciably impacts to the fitness of individual fur seals.

As discussed above, Navy explosive training activities are expected to result in an estimated one Guadalupe fur seal mortality every seven years of the proposed action. Death would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to the population. Our best available estimate of the total number of Guadalupe fur seals is 34,187 (García-Aguilar et al. 2018; Carretta 2021b). Assuming a balanced sex ratio, this means there are about 17,094 female Guadalupe fur seals in the population. A reasonably predictable outcome is that the one Guadalupe fur seal mortalities every seven years from Navy explosives would be a female of early reproductive age. This would reduce the reproductive

potential of the Guadalupe fur seals in the action area by less than 0.01%. This is not an appreciable reduction in the numbers or the reproductive capability of the Guadalupe fur seal population. If the analyzed rate of mortality for this species was to recur in subsequent 7-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. In addition, the relatively small loss of reproductive potential would not affect the current geographic range of Guadalupe fur seals and no reduction in the distribution of this species is expected as a result of the action.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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. Similarly, the effects from ongoing HCTT 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. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of Guadalupe fur seals in the wild.

8.1.9 Hawaiian Monk Seal

The population summary for Hawaiian monk seals in 2023 provides the best estimate for the species abundance as 1,605 (with 95% confidence that the true number is between 1,512 and 1,743) (Carretta et al. 2025). The population has grown at an average annual rate of about 2% from 2013-2022 (Carretta et al. 2023a; NMFS 2023e; NOAA 2023).

Hawaiian monk seals are expected to experience non-auditory injury, AINJ, TTS, behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the HRC. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 691 instances of behavioral harassment, 198 instances of TTS, and five instances of AINJ are reasonably certain to occur to this species annually, in a maximum year of HCTT sonar and explosive activities. In addition, Navy explosive training activities are expected to result in an estimated one Hawaiian monk seal non-auditory injury every seven years of the proposed action. We also estimate up to 360 instances of behavioral harassment to hauled out Hawaiian monk seals annually due to in-air acoustic effects from PMRF weapons firing noise (Section 6.1.2). Based on the best available information on the exposure of Hawaiian monk seals to HCTT vessel strike (See Section 6.1.3), no injury or mortality of this species is reasonably certain to occur from this stressor.

As described in greater detail in Section 6.1.1, we do not anticipate instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or survival of Hawaiian monk seals. Any adverse effects resulting from TTS or behavioral

responses are expected to be relatively minor and short-term. Unlike TTS, AINJ could be permanent (e.g., PTS), meaning the effects last well beyond the duration of the action and outside of the action area as animals migrate. As such, PTS has the potential to affect aspects of the affected animal's life functions that do not overlap in time and space with the action. From our effects analysis we estimate up to five instances of Hawaiian monk seal AINJ annually, but, as discussed in more detail in Section 6.1.1, in most instances the magnitude of this impairment is expected to be minor due to the methodology used to calculate impacts and the Action Proponents' proposed mitigation. With this anticipated minor degree of AINJ (i.e., mild severity PTS), even though some Hawaiian monk seals are expected to experience a minor reduction in fitness, we would not expect such impacts to have meaningful effects at the population level. That is, individual seals affected could be less efficient at locating conspecifics or have decreased ability to detect threats at long distances, but these animals are still expected to be able to locate conspecifics to socialize and reproduce, and will still be able to detect threats with enough time to avoid injury. For this reason, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Hawaiian monk seals in the action area or range wide.

Although we anticipate that a large majority of PTS exposures will be of mild severity, some Hawaiian monk seals will likely experience moderate to high severity PTS from exposure to broadband sound sources (e.g., explosives), which increase the chance that hearing impairment will affect frequencies utilized by animals for acoustic cues. However, since the number of monk seals that could experience long-term fitness consequences from PTS is relatively small (i.e., likely less than four per year), compared to the overall Hawaiian monk seals population size, we do not anticipate population level impacts as a result. In addition, compared to cetaceans, monk seals are not known to rely as heavily on sound for important biological functions, although PTS could affect Hawaiian monk seal communication with conspecifics. For these reasons, we do not anticipate that instances of AINJ will result in changes in the number, distribution, or reproductive potential of Hawaiian monk seals in the action area or throughout their range.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in Hawaiian monk seals given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. Hawaiian monk seals exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). However, masking effects would be limited for Hawaiian monk seals since the effects only occur in the presence of the masking noise, and this species is not known to rely heavily on hearing or sound production for important biological functions. In most instances masking would be minor and short-term, with no appreciably impacts to the fitness of individual monk seals.

Downlisting criteria provided in the 2007 Hawaiian monk seal recovery plan include: 1) aggregate numbers exceed 2,900 total individuals in the NWHI; 2) at least five of the six main subpopulations in the NWHI are above 100 individuals and the MHI population is above 500; and 3) survivorship of females in each subpopulation in the NWHI and in the MHI is high enough that, in conjunction with the birth rates in each subpopulation, the calculated population

growth rate for each subpopulation is not negative. The population will be considered for a delisting if the downlisting criteria continue to be met for 20 consecutive years without new crucial or serious threats being identified. The plan includes 14 categories of actions that are necessary for the recovery of the Hawaiian monk seal. Of these, only the category "reduce the likelihood and impact of human interactions" is relevant to the activities and stressors associated with the proposed action. From our effects analysis, we concluded no mortality of individuals would occur and that effects from acoustic stressors would be temporary, with only minor fitness consequences from PTS to a small number of individuals. In the absence of significant 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.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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 Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 Hawaiian monk seals in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of Hawaiian monk seals in the wild.

8.1.10 False Killer Whale – Main Hawaiian Islands Insular DPS

Badger et al. (2025) estimate 139 individuals (95% CRI: [120, 160]) in the population in 2021. The population appears to be in decline, with an estimated trend of -3.51 (95% CRI: [-8.40, 2.04]) over the time series (1991–2021) and -5.53, 95% CRI: [-9.91, -1.61]) over the last 10 years.

MHI IFKWs are expected to experience TTS and behavioral and physiological stress responses as a result of exposures to HCTT sonar and explosives in the HRC. Based on the quantitative acoustic effects analysis (i.e., NAEMO), 105 instances of behavioral harassment and 64 instances of TTS are reasonably certain to occur to this DPS annually, in a maximum year of HCTT sonar and explosive activities. Based on the best available information on the exposure of MHI IFKWs to sonar, air guns, and explosives, and as detailed in Section 6.1.1, no injury (auditory or non-auditory) or mortality of this DPS is reasonably certain to occur from these stressors. Based on the best available information on the exposure of MHI IFKWs to HCTT vessel strike (See Section 6.1.3), no injury or mortality of this species is reasonably certain to occur from this stressor.

As described in greater detail in Section 6.1.1, we do not anticipate instances of TTS or behavioral harassment to result in long-term fitness consequences affecting the reproduction or

survival of MHI IFKWs. Any adverse effects resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Long-term or chronic physiological stress responses that could potentially result in fitness consequences are not anticipated to occur in MHI IFKWs given the types of HCTT sonar systems proposed and the dispersed nature of sonar use in space and time. MHI IFKWs exposed to military sonar can experience masking if the sonar interferes with the animal's ability to communicate with conspecifics or detect biologically relevant sounds (see Section 6.1.1). Odontocete species (including false killer whales), which rely on echolocation to find prey, are more susceptible to masking vocalization given the overlap in vocalization frequency with these mid to high-frequency sonar systems. Since masking only occurs in the presence of the masking noise, most instances would be minor and short-term, with no appreciable fitness impacts. From our acoustic effects analysis, MHI IFKWs will likely be exposed to high duty cycle or continuous active sonar (e.g., MF1C) at levels that could result in masking effects that are longer lasting compared to pulsed active sonar (e.g., MF1; see Section 6.1.1 for details). However, even with high duty cycle and continuous sonar sources, the duration of masking would be limited since both the sonar source and the animal are moving within the environment, and we do not anticipate false killer whales would remain in close proximity to a source for an extended period of time.

The HCTT activities with mid-frequency (MF) and high-frequency (HF), high-duty cycle sonar sources that NAEMO predicted impacts to MHI IFKWs from are shown in Table 51. Masking from high duty cycle or continuous active sonar with frequency ranges that overlap MHI IFKW vocalizations could lead to longer-term behavioral disruptions, affecting false killer whale foraging and communication. Masking as a result of Navy mid-frequency (other than hull-mounted), high duty cycle sonar sources within the 4-10 kHz range may impact MHI IFKW's ability to communicate and socialize. False killer whale echolocation clicks, which have been recorded from 17-104 kHz (Madsen et al. 2004; Thode et al. 2016), could potentially be masked by sound from high-frequency (or broadband) sonar, although any masking effects from high-frequency sonar, including high-duty cycle sources, would likely be minimal given the smaller zone of effect and lower sources levels, as compared to mid-frequency anti-submarine warfare sonar.

While high duty cycle and continuous active sonars transmit more frequently, they typically operate at a substantially lower source level as compared to traditional pulsed sonar. Although the Action Proponents have proposed an increase in the use of high duty cycle and continuous active sonar from Phase III, the large majority of active sonar for HCTT Phase IV within the HRC is pulsed sonar with low duty cycles (see Table 50). In addition, most sonar sources used by the Action Proponents operate over a narrow band of frequencies which only overlap a small portion of the full range of IFKW hearing, although some activities involving high duty cycle and continuous sonar also use broadband sonar sources (e.g., anti-submarine warfare training). For example, Navy surface ship hull-mounted mid-frequency sonar (i.e., MF1 low duty cycle; MF1C high duty cycle) is a narrow band frequency source centered around or about 3 kHz, which is just below the frequency range of false killer whale whistles and well below the

frequency range of echolocation clicks. Based on the estimated MHI IFKW population size and NAEMO impacts analysis (see Table 46), we expect that individual MHI IFKWs within the action area will be adversely affected by HCTT acoustic stressors less than twice per year, on average (i.e., some individuals may be impacted more than others). Therefore, while masking may result in fitness consequences (including potential impacts on foraging efficiency), the duration, magnitude and scope of these impacts on individual false killer whales will be limited, and are not expected to result in impacts to reproduction or survival of this ESA-listed DPS at the population level.

Since spatial use among the four MHI IFKW social clusters varies among the Main Hawaiian Islands (Baird et al. 2023), acoustic impacts are likely not distributed evenly across the clusters. Historical data provided by the Navy indicate very low levels of mid-frequency active sonar use within the Hawaii 4-Islands Marine Mammal Mitigation Area (see Section 2.4.2) (Navy 2025l). This geographic mitigation area was expanded for HCTT Phase IV to include more high use areas within the identified MHI IFKW child BIA, including areas to the southwest of Lanai and north of Molokai. Considering this, and the seasonal restriction on mid-frequency active sonar within this mitigation area, clusters primarily seen within this mitigation area may have a lower risk of HCTT sonar exposures as compared to clusters that spend more time around islands that are either not protected by geographic mitigation measures (e.g., Oahu), and/or in areas identified by the Navy as particularly important for sonar training events (e.g., Alenuihaha Channel between Hawaii and Maui; Kaiwi Channel between Molokai and Oahu). Mahaffy et al. (2023) found that: Cluster 1 (n=62 individuals) was seen primarily off the northwest side of Hawaii, including the Alenuihaha Channel, but also in the 4-islands region; Cluster 2 (n=15 individuals) was primarily seen between Hawaii and Maui, including the Alenuihaha Channel; Cluster 3 (n=60 individuals) was seen across all Main Hawaiian Islands but primarily in the area between Oahu and Molokai, including the Kaiwi Channel; and Cluster 4 (n=37 individuals) was seen primarily off the western sides of Molokai and Lanai, including the Kaiwi Channel. While an analysis of acoustic impacts by IFKW social cluster may help inform our DPS risk analysis, the available information does not support a more detailed assessment of the spatial overlap between Navy sonar use and social cluster occurrence.

The final recovery plan for the MHI IFKW DPS identified delisting criteria for seven demographic and threats based recovery objectives. Of these, the only objective directly relevant to the Navy's action is to address threats from anthropogenic noise. The delisting criteria for the anthropogenic noise objective is as follows: *Management actions sufficiently address the effects of anthropogenic ocean noise (e.g., vessel traffic, sonar, alternative energy development) on MHI IFKWs and their habitat such that it is not adversely affecting and/or reducing their ability to successfully travel, communicate, and forage, and is not causing population-level effects*. As discussed above, while individual MHI IFKWs are likely to be adversely affected by HCTT sonar and explosives, we do not anticipate such effects will result in long-term fitness consequences that would appreciably affect their ability perform important life functions. In the absence of long-term 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. The effects of noise from the proposed action on MHI IFKW critical habitat are discussed in Section 8.2.1 below.

In summary, 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 Action Proponents will conduct in the HCTT action area on an annual basis, cumulatively over the remaining period of the MMPA rule, 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 MHI IFKWs in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 MHI IFKW in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of MHI IFKW in the wild.

8.1.11 Green Sea Turtle - Central North Pacific and East Pacific DPSs

On April 6, 2016, NMFS listed eleven DPSs of green sea turtles, including two DPSs listed as threatened that occur within the HCTT action area: Central North Pacific and East Pacific.

Central North Pacific DPS

The Central North Pacific DPS is estimated to have 3,846 nesting females, which nest across 12 sites, with 96% nesting at the largest site of East Island, French Frigate Shoals, Hawaii (Seminoff et al. 2015). Nesting abundance at East Island, French Frigate Shoals has increased at 4.8% annually.

Central North Pacific DPS green sea turtles are expected to experience mortality, non-auditory injury, AINJ, TTS, behavioral and physiological stress responses in the HRC from HCTT sonar, air guns, and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 2,068 instances of behavioral harassment, 1,171 instances of TTS, and 45 instances of AINJ in a maximum year of HCTT sonar, air guns and explosives activities. The model also predicts 11 instances of non-auditory injury and five mortalities to this DPS from HCTT explosive activities every seven years.

As discussed in Section 6.2.1, a non-auditory injury can impact an individual sea turtle's fitness, causing a reduction in foraging success, reproduction, or increased susceptibility to predators. Depending on the type of injury (e.g., severity, location) and other relevant factors (e.g., animal life stage, size, condition prior to exposure), a reduction in fitness from non-auditory injury could either be temporary with recoverable impacts, or result in long-term fitness consequences for some individuals. While long-term fitness consequences could affect population growth, the quantitative analysis estimated a relatively small number (i.e., less than two per year) of non-auditory injuries to this DPS. Considering the size of the population, any such fitness impacts

would not result in an appreciable reduction in the reproductive potential or growth rate of the Central North Pacific DPS of green sea turtles.

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of green sea turtle PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions.

We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Based on our sea turtle vessel strike effects analysis (Section 6.2.2), we estimate there would be three lethal vessel strikes of Central North Pacific DPS green sea turtles annually under the action. Since some of the turtles killed by vessel strike could be females, the action may also result in a reduction in reproduction of this species. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

From our analysis of anticipated life stages and sex ratios of sea turtle vessel strikes (Section 6.2.1), we assume that of the estimated three lethal strikes annually two would be females. We add to the vessel strike mortality one Central North Pacific DPS green sea turtle mortality per year (i.e., model predicted five over seven years) from explosives, to get three total nesting female mortalities per year. This represents an annual loss of 0.08% of the estimated 3,846 nesting females (Seminoff et al. 2015) for the Central North Pacific DPS (note: this is likely an overestimate because it assumes all females killed will be mature adults). We do not consider this to be an appreciable reduction in the numbers of female green sea turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. The anticipated loss of current and future reproductive potential is relatively small compared and would have a negligible effect on the recent increasing trend in nesting abundance reported for this DPS. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population of Central North Pacific DPS green sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the current geographic range of this DPS and no reduction in the distribution of this DPS is expected as a result of the action.

The *Recovery Plan for U.S. Pacific Populations of the Green Turtle* (NMFS and USFWS 1998c) identified the following delisting criteria related to the size of nesting and foraging populations:

1) each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over 6 years; 2) nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period; and 3) Foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region. While we do anticipate mortality of Central North Pacific DPS green sea turtles as a result of Navy vessel strike, the mortality level anticipated will not result in an appreciable reduction in the numbers or reproductive rate of this population of green sea turtles. Thus, we do not expect this level of mortality to impact the survival or recovery of this population.

In summary, 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 Action Proponents will conduct in the HCTT 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 Central North Pacific DPS green sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 Central North Pacific DPS green sea turtles in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of Central North Pacific DPS green sea turtles in the wild.

Green Sea Turtle East Pacific DPS

Green turtles in the East Pacific DPS are found from the California/Oregon border south to central Chile. The largest nesting site is at Colola, Mexico, which hosts 58% of the nesting females for the DPS (Seminoff et al. 2015). The observed increases in nesting abundance at Colola, a stable trend at Galápagos, and record high numbers at sites in Costa Rica suggest that the population is resilient to future perturbations.

Green turtles in the East Pacific DPS are found from the California/Oregon border south to central Chile. There are thirty-nine nesting sites for the East Pacific DPS, with an estimated 20,062 nesting females. Major nesting sites occur at Michoacán, Mexico, and the Galápagos Islands, Ecuador. The largest nesting site is at Colola, Mexico, which hosts 58% of the nesting females for the DPS (Seminoff et al. 2015). There are no estimates of population growth for the DPS as a whole, although some populations, including the one in Southern California, are likely increasing in recent decades (Seminoff et al. 2015; Hanna et al. 2021).

East Pacific DPS green sea turtles are expected to experience non-auditory injury, AINJ, TTS, behavioral and physiological stress responses in the HRC from HCTT sonar, air guns, and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 53 instances of behavioral harassment, 638 instances of TTS, and nine instances of AINJ in a maximum year of HCTT sonar, air guns and explosives activities. The model also predicts one non-auditory injury from HCTT explosive activities every seven years.

As discussed in Section 6.2.1, a non-auditory injury can impact an individual sea turtle's fitness, causing a reduction in foraging success, reproduction, or increased susceptibility to predators. Depending on the type of injury (e.g., severity, location) and other relevant factors (e.g., animal life stage, size, condition prior to exposure), a reduction in fitness from non-auditory injury could either be temporary with recoverable impacts, or result in long-term fitness consequences for some individuals. While long-term fitness consequences could affect population growth, the quantitative analysis estimated a relatively small number (i.e., one every seven years) of non-auditory injuries to this DPS. Considering the size of the population, any such fitness impacts would not result in an appreciable reduction in the reproductive potential or growth rate of the East Pacific DPS of green sea turtles.

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of green sea turtle PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions.

We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Our vessel strike analysis (Section 6.2.2) estimates there will be seven lethal and five nonlethal vessel strikes of East Pacific DPS sea turtles annually under the action. Since some of the turtles killed by vessel strike could be females, the action may also result in a reduction in reproduction of this species. Loss of a sexually mature female would have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

From our analysis of anticipated life stages and sex ratios of sea turtle vessel strikes (Section 6.2.1), we assume that of the estimated seven lethal strikes annually, up to five would be females. There are 39 nesting sites for the East Pacific DPS, with an estimated 20,062 nesting females (Seminoff et al. 2015). The loss of five female green turtles in a given year due to vessel strike would reduce the reproductive potential of this population by less than 0.03% (note: this is likely an overestimate because it assumes all females killed will be mature adults). We do not consider this to be an appreciable reduction in the numbers of female green sea turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population of green sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the

current geographic range of this DPS and no reduction in the distribution of this DPS is expected as a result of the action.

Up to three green sea turtles from this DPS are also expected to experience sub-lethal effects from a vessel strike as a result of the action. Injury from a vessel strike may result in temporary reduced fitness until the injury heals or potentially have longer term consequences for serious injuries. Nonetheless, we do not expect this level of injured sea turtles to result in an appreciable reduction in the reproductive rate of this population. Overall, the impacts expected to occur and affect East Pacific DPS green sea turtles in the action area are not anticipated to result in appreciable reductions in overall reproduction, abundance, or distribution of this species.

The recovery plan for the Eastern Pacific green sea turtle identifies six major actions needed to achieve recovery (NMFS and USFWS 1998b). Of these, the one identified action of relevance to HCTT activities is to minimize boat collision mortalities, particularly within San Diego County. While we do anticipate a small number of East Pacific DPS green sea turtles would be struck by Action Proponent vessels operating in San Diego Bay, the mortality level anticipated would not result in an appreciable reduction in the numbers or reproductive rate of this DPS of green sea turtles. Thus, we do not expect this level of mortality to impact the survival or recovery of this population.

In summary, 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 Action Proponents will conduct in the HCTT 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 East Pacific DPS green sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 East Pacific DPS green sea turtles in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of East Pacific DPS green sea turtles in the wild.

8.1.12 Hawksbill Sea Turtle

The hawksbill turtle has a circumglobal distribution throughout tropical and, to a lesser extent, sub-tropical waters of the Atlantic, Indian, and Pacific Oceans. Based on the best available information, approximately 25,000 female hawksbill sea turtles nest each year throughout these three ocean basins (NMFS 2013b).

Hawksbill sea turtles are expected to experience AINJ, TTS, behavioral and physiological stress responses in the HRC from HCTT sonar and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 19 instances of behavioral harassment and 18 instances of TTS

in a maximum year of HCTT sonar and explosives activities. The model also predicts two AINJ of hawksbills every seven years of the proposed action.

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of hawksbill PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions. We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Based on our sea turtle vessel strike effects analysis (Section 6.2.2), we estimated there would be three lethal vessel strike of hawksbill sea turtles every seven years of the proposed action. Since some of the hawksbills killed by vessel strike could be females, the action may also result in a reduction in reproduction of this species. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

An average of between 11,000 and 12,700 hawksbill nests are estimated to occur each year in the Pacific (USFWS 2012). On average, hawksbill turtles nest every two or three years and lay 4.5 nests each year (USFWS 2012). If we assume that most turtles nest every two years and the lower estimate of the number of nests annually, this equates to a likely total population size of approximately 4,889 females in the Pacific. The loss of up to three adult female hawksbills (i.e., conservatively assuming all three strikes are of adult females) every seven years due to vessel strike would reduce the reproductive potential of the Pacific population by about 0.06% every seven years. We do not consider this to be an appreciable reduction in the numbers of female hawksbill turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population of hawksbill sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the current geographic range of hawksbill sea turtles and no reduction in the distribution of this species is expected as a result of the action.

The *Recovery Plan for U.S. Pacific Populations of the Hawksbill Turtle* (NMFS and USFWS 1998d) identified the following delisting criteria related to the size of nesting and foraging populations: 1) each stock must average 1,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over

6 years; 2) all females estimated to nest annually at "source beaches" are either stable or increasing for 25 years; and 3) foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region. While we do anticipate a small number of hawksbill sea turtles will be struck by Action Proponent vessels within the HRC, the mortality level anticipated will not result in an appreciable reduction in the numbers or reproductive rate of this species. Thus, we do not expect this level of mortality to impact the survival or recovery of hawksbill sea turtles.

In summary, 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 Action Proponents will conduct in the HCTT 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 hawksbill sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 hawksbill sea turtles in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of hawksbill sea turtles in the wild.

8.1.13 Leatherback Sea Turtle

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 HCTT action area. Western Pacific leatherbacks nest in the Indo-Pacific, primarily in Indonesia, Papua New Guinea, and the Solomon Islands. Martin et al. (2020b) provided a median estimate of the total number of nesting females (i.e., over one, 3-year remigration interval) at Jamursba Medi and Wermon beaches of 790 females, as a snapshot of current abundance in 2017. Applying the reasonable, science-based estimate of 75% to the Martin et al. (2020a) estimate of 790 nesting females at Jamursba Medi and Wermon beaches, the total number of nesting females in the West Pacific population would be 1,054 females. The median trend in annual nest counts estimated for Jamursba Medi nesting beaches from data collected from 2001 to 2017 was negative 5.7% annually (NMFS and USFWS 2020a). The median trend in annual nest counts estimated for Wermon nesting beaches from data collected from 2006 to 2017 (excluding 2013 through 2015 due to low or insufficient effort) was negative 2.3% annually (NMFS and USFWS 2020a). In the absence of population trend data on other leatherback life history stages, we consider these trends in annual nest counts an index of the population’s growth rate.

Leatherback sea turtles are expected to experience AINJ, TTS, behavioral and physiological stress responses in the HRC and California portion of the action area from HCTT sonar and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 49 instances of

behavioral harassment and 384 instances of TTS in a maximum year of HCTT sonar and explosives activities. The model also predicts 19 AINJ of leatherbacks every seven years of the proposed action. Based on our sea turtle vessel strike effects analysis (Section 6.2.2), we estimate there would be no vessel strikes of leatherback sea turtles as a result of the proposed action.

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of leatherback PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions. We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

The *Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle* (NMFS and USFWS 1998a) identified the following delisting criteria related to the size of nesting and foraging populations: 1) each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over 6 years; 2) nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period; and 3) foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region. Since we do not anticipate any effects would result in the mortality or appreciably reduce the fitness of individual leatherback sea turtles, any adverse effects from the action would not impact the survival or recovery of this species.

In summary, 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 Action Proponents will conduct in the HCTT 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 leatherback sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 leatherback sea turtles in the wild. Thus, we determine that the action

would not appreciably reduce the likelihood of both survival and recovery of leatherback sea turtles in the wild.

8.1.14 Loggerhead Sea Turtle – North Pacific DPS

On September 22, 2011, the NMFS designated nine DPSs of loggerhead sea turtles, including the North Pacific DPS, which was listed as endangered. The estimated 2015 total nesting abundance for the loggerhead North Pacific DPS is approximately 8,733 nesting females (USFWS 2020). North Pacific DPS loggerhead nesting likely increased between 1999 and 2012 at a rate of around 2.3% annually (Martin et al. 2020a; USFWS 2020).

Loggerhead sea turtles are expected to experience non-auditory injury, AINJ, TTS, behavioral and physiological stress responses in the HRC and California portion of the action area from HCTT sonar and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 135 instances of behavioral harassment, 727 instances of TTS, and nine AINJ in a maximum year of HCTT sonar and explosives activities. The model also predicts seven non-auditory injuries of loggerheads every seven years of the proposed action.

As discussed in Section 6.2.1, a non-auditory injury can impact an individual sea turtle's fitness, causing a reduction in foraging success, reproduction, or increased susceptibility to predators. Depending on the type of injury (e.g., severity, location) and other relevant factors (e.g., animal life stage, size, condition prior to exposure), a reduction in fitness from non-auditory injury could either be temporary with recoverable impacts, or result in long-term fitness consequences for some individuals. While long-term fitness consequences could affect population growth, the quantitative analysis estimated a relatively small number (i.e., one per year, on average) of non-auditory injuries to this DPS. Considering the size of the population, any such fitness impacts would not result in an appreciable reduction in the reproductive potential or growth rate of the North Pacific DPS of loggerhead sea turtles.

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger, such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of loggerhead PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions. We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Based on our sea turtle vessel strike effects analysis (Section 6.2.2), we estimated there would be up to one lethal vessel strike of loggerhead sea turtles annually under the action. Since some of

the loggerheads killed by vessel strike could be females, the action may also result in a reduction in reproduction of this species. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

The annual loss of one adult female loggerhead due to a vessel strike would reduce the reproductive potential of the North Pacific population (estimated at 8,733 nesting females) by about 0.01%. We do not consider this to be an appreciable reduction in the numbers of female loggerhead turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population of loggerhead sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population. In addition, the relatively small loss of reproductive potential resulting from vessel strike will not affect the current geographic range of loggerhead sea turtles and no reduction in the distribution of this species is expected as a result of the action.

The *Recovery Plan for U.S. Pacific Populations of the Loggerhead Turtle* (NMFS and USFWS 1998e) identified the following delisting criteria related to the size of nesting and foraging populations: 1) each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually over 6 years; 2) nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period, and 3) foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region. While we do anticipate a small number of loggerhead sea turtles will be struck by Action Proponent vessels, the mortality level anticipated will not result in an appreciable reduction in the numbers or reproductive rate of this species. Thus, we do not expect this level of mortality to impact the survival or recovery of North Pacific DPS loggerhead sea turtles.

In summary, 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 Action Proponents will conduct in the HCTT 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 North Pacific DPS loggerhead sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT 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 North Pacific DPS loggerhead sea turtles in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of North Pacific DPS loggerhead sea turtles in the wild.

8.1.15 Olive Ridley Sea Turtle

The olive ridley is separated into two listing designations: endangered for breeding populations on the Pacific coast of Mexico, and threatened wherever found except where listed as endangered (i.e., in all other areas throughout its range). The endangered Mexico breeding population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and USFWS 2014). The portion of the threatened population that nests in the East Pacific counts of adult olive ridley sea turtles on eastern Pacific nesting beaches similarly number at more than 1,000,000 nesting females on beaches from Costa Rica and Nicaragua (Valverde et al. 2012; NMFS and USFWS 2014). The estimated minimum population size of olive ridley sea turtles (juveniles and adults) in the Western Pacific is 316,883 (NMFS 2023h).

Olive ridley sea turtles are expected to experience AINJ, TTS, behavioral and physiological stress responses in the HRC from HCTT sonar and explosives. The quantitative acoustic effects analysis (i.e., NAEMO) predicts 37 instances of behavioral harassment and 224 instances of TTS in a maximum year of HCTT sonar and explosives activities. The model also predicts 13 AINJ of olive ridleys every seven years of the proposed action. Based on the area of impact, we expect the large majority of these impacts will be to olive ridley turtles from the threatened population (i.e., not from the Pacific coast of Mexico population).

Auditory injury in the form of PTS could decrease an individual sea turtle's ability to detect danger, such as approaching vessels or predators; and may reduce foraging or breeding opportunities or increase risks of sustaining other harm. While some proportion of olive ridley PTS exposures could result in minor fitness consequences, sea turtles, in general, are not considered to rely extensively on their hearing for important life functions (Popper et al. 2014a; Nelms et al. 2016). As such, the likelihood that the loss of hearing in a sea turtle would impact its fitness (i.e., survival or reproduction) is low when compared to marine mammals, which rely heavily on sound for basic life functions. We have no information to suggest that TTS in sea turtles would result in fitness consequences to individuals or the populations to which they belong. Similarly, behavioral and physiological responses to acoustic stressors are expected to be temporary and episodic, with no reduction in fitness of individual turtles. Any adverse effects to sea turtles resulting from TTS or behavioral responses are expected to be relatively minor and short-term.

Based on our sea turtle vessel strike effects analysis (Section 6.2.2), we estimated there would be up to one lethal vessel strike of olive ridley sea turtles annually under the action from each of the two ESA-listed populations. Since some of the olive ridleys killed by vessel strike could be females, the action may also result in a reduction in reproduction of this species. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

Mexico's Pacific Coast Breeding Population

There are six primary arribada nesting beaches used by Mexico's Pacific Coast breeding population of olive ridleys. The largest of these is La Escobilla, with an increasing abundance trend and an estimated one million nesting females annually (NMFS and USFWS 2014). The endangered Mexico population of olive ridley sea turtles is comprised of more than 1,000,000 nesting females (Abreu-Grobois and Plotkin 2008; Valverde et al. 2012; NMFS and USFWS 2014). At-sea estimates of olive ridleys off of Mexico and Central America also support an increasing population trend. The loss of one female olive ridley in a given year due to a vessel strike would not appreciably impact the numbers of females or the reproductive potential of the Mexico Pacific breeding population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population, we do not expect this level of mortality to impact the survival or recovery of this population. In addition, the relatively small loss of reproductive potential resulting from vessel strikes will not affect the current geographic range of this olive ridely population and no reduction in the distribution of this population is expected as a result of the action.

The *Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle* (NMFS and USFWS 1998f) identified the following delisting criteria related to the size of nesting and foraging populations: 1) all females estimated to nest annually at "source beaches" are either stable or increasing for 10 years; and 2) foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region. While we do anticipate a small number of olive ridley sea turtles will be struck by Action Proponent vessels, the mortality level anticipated will not result in an appreciable reduction in the numbers or reproductive rate of this species. Thus, we do not expect this level of mortality to impact the survival or recovery of the Mexico Pacific breeding population of olive ridley.

In summary, 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 Action Proponents will conduct in the HCTT 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 olive ridley Mexico Pacific breeding population sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT activities and related incidental take continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of olive ridley Mexico Pacific breeding population sea turtles in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of olive ridley Mexico breeding population sea turtles in the wild.

All Other Breeding Populations

The olive ridley is thought to be the most abundant sea turtle in the world. There is no global estimate of olive ridley abundance, and we rely on nest counts and nesting females to estimate

abundance in each of the ocean basins, described below. Eguchi et al. (2007) estimated a weighted average of the yearly abundance estimates as 1.39 million. The portion of the threatened olive ridley sea turtle population that nests in the East Pacific from Costa Rica and Nicaragua is estimated at more than 1,000,000 nesting females (Valverde et al. 2012; NMFS and USFWS 2014). The estimated minimum population size of olive ridley sea turtles (juveniles and adults) in the Western Pacific is 316,883 (Navy 2023). Despite large declines in recent decades, the species' large population size allows some resilience to future perturbation.

As with the Mexico Pacific breeding population, the loss of one female olive ridley per year throughout the remainder of its range would not appreciably impact the numbers of females or the reproductive potential of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population, we do not expect this level of mortality to impact the survival or recovery of this population.

As discussed above for the Mexico Pacific population, the olive ridley recovery plan identified specific delisting criteria related to the size of nesting and foraging populations. While we do anticipate a small number of olive ridley sea turtles would be struck by Action Proponent vessels, we do not expect this level of mortality to impact the survival or recovery of olive ridley sea turtles from all other breeding populations.

In summary, 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 Action Proponents will conduct in the HCTT 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 olive ridley sea turtles from all other breeding populations in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT activities and related incidental take continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of olive ridley sea turtles from all other breeding populations in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery all other breeding populations of olive ridley sea turtles in the wild.

8.1.16 Giant Manta Ray

The giant manta ray, listed as threatened under the ESA (83Fed. Reg. 2916; January 22, 2018), is a large, migratory elasmobranch that inhabits tropical, subtropical, and temperate waters globally. They frequent productive coastlines, oceanic islands, offshore pinnacles, and seamounts, often aggregating in feeding and cleaning stations.

From our fish acoustic effects analysis (Section 6.3.1), we determined that giant manta rays are likely to be adversely affected by HCTT explosives. Lethal impacts may occur when individuals are within close proximity to detonations, particularly in shallow or mid-depth water columns. Even if not fatal, exposure to high-intensity shock waves and pressure variations can cause significant physiological stress, temporary disorientation, and impaired swimming ability. Sub-

lethal injuries such as temporary threshold shifts (TTS) in hearing, minor barotrauma, and reduced foraging efficiency have been observed in similar large pelagic species (Wysocki et al. 2006; Popper et al. 2014a; Casper et al. 2016).

Due to the lack of available density or abundance information for giant manta rays in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of giant manta rays likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a very small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. Due to the relative rarity of this species in the vast majority of the action area, the dispersed nature of HCTT activities involving explosives, and the small range to effects resulting in mortality for most HCTT explosives, the lethal take of giant manta rays is expected to be relatively low. Most giant manta ray exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. While we do anticipate a very small number of giant manta ray mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of giant manta rays within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this species. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect the giant manta ray. The abundance, distribution, and reproduction of giant manta rays are not expected to be appreciably reduced by HCTT activities.

The initial focus of the interim recovery program, as stated in the NMFS giant manta ray recovery outline, is to: 1) stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and 2) gather additional information through research and monitoring on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries, and other potential threats that may contribute to the species' decline (NMFS 2019f). While we do anticipate a small number of giant manta rays would be killed or seriously injured by HCTT explosives, the mortality level anticipated would not result in an appreciable reduction in the numbers or reproductive rate of this species. Thus, we do not expect this level of mortality to impact the survival or recovery of giant manta rays.

In summary, 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 Action Proponent will conduct in the HCTT 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 giant manta rays in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT

activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of giant manta rays in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of giant manta rays in the wild.

8.1.17 Oceanic Whitetip Shark

Although global population abundance for this species is unavailable, best available scientific and commercial data indicate this species has undergone significant historical declines throughout its range. For example, the final rule to list this species as threatened presented data indicating oceanic whitetips comprised 20% of the total shark catch in the tropical tuna purse seine fishery in the early 2000s, but from 2005 to 2009, the proportion of oceanic whitetip sharks was less than 2% (83Fed. Reg. 4153). Continued fishing pressure and associated mortality is the greatest ongoing threat to the recovery of this species. NMFS (2025b) estimated a 2024 population abundance around 2.3 million oceanic whitetip sharks in the Pacific Ocean.

Despite this population's anticipated low density in the action area, we determined that a small number of individual oceanic whitetip sharks are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to exposure of oceanic whitetip sharks to explosives.

Due to the lack of available density or abundance information for oceanic whitetip sharks in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of oceanic whitetip sharks likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. Due to the relative rarity of this species in the vast majority of the action area, the dispersed nature of HCTT activities involving explosives, and the small range to effects resulting in mortality for most HCTT explosives, the lethal take of oceanic whitetip sharks is expected to be relatively low. Most oceanic whitetip shark exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult oceanic whitetip shark not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. While we do anticipate a very small number of oceanic whitetip shark mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of oceanic whitetip sharks within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this species. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect the oceanic whitetip shark. The abundance, distribution, and reproduction of oceanic whitetip sharks are not expected to be appreciably reduced by HCTT activities.

The three recovery objectives from the recovery plan for the oceanic whitetip shark are to: 1) Ensure the oceanic whitetip shark maintains resiliency and geographic representation, and is a functional component of the ecosystem, by increasing overall abundance to achieve viable populations in all ocean basins; 2) Increase oceanic whitetip shark resiliency by managing or eliminating significant anthropogenic threats; and 3) Ensure the continued viability of the oceanic whitetip shark through development and effective implementation of regulatory mechanisms for the long-term protection of the species (NMFS 2024c). NMFS also developed four alternative demographic recovery criteria that can be used to determine when a stable and sustainable population size has been reached. Recovery objectives are not likely to be impacted by this action because we anticipate only a very small number of oceanic whitetip sharks will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 oceanic whitetip sharks in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of oceanic whitetip sharks in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of oceanic whitetip sharks in the wild.

8.1.18 Scalloped Hammerhead Shark – Eastern Pacific DPS

Although overall abundance data for the Eastern Pacific DPS scalloped hammerhead shark is lacking, populations appear to be on the decline, most notably due to both legal and illegal capture in fisheries south of the action area (Miller et al. 2014; Peñaherrera-Palma et al. 2018).

Despite this population's anticipated low abundance in the action area, we determined that a small number of scalloped hammerhead sharks are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to exposure of scalloped hammerhead sharks to explosives.

Due to the lack of available density or abundance information for scalloped hammerhead sharks in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of scalloped hammerheads likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. Due to the relative rarity of this species in the vast majority of the action area, the dispersed nature of HCTT activities involving explosives, and the small range to effects resulting in mortality for most HCTT explosives, the

lethal take of scalloped hammerhead sharks is expected to be relatively low. Most scalloped hammerhead shark exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult scalloped hammerhead shark not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. While we do anticipate a very small number of scalloped hammerhead shark mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect scalloped hammerheads. The abundance, distribution, and reproduction of Eastern Pacific DPS scalloped hammerhead sharks are not expected to be appreciably reduced by HCTT activities.

In summary, 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 Action Proponent will conduct in the HCTT 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 Eastern Pacific DPS scalloped hammerhead sharks in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Eastern Pacific DPS scalloped hammerhead sharks in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Eastern Pacific DPS scalloped hammerhead sharks in the wild.

8.1.19 Steelhead – Southern California DPS

Southern California DPS steelhead occur in very low numbers in their historical range, inclusive of the California portion of the action area. A combination of persistent anthropogenic threats, mostly in freshwater spawning and rearing habitats and including fish passage barriers, continue to hinder this species' recovery.

Despite this population's anticipated low abundance in the action area, we determined that a small number of individual Southern California DPS steelhead are likely to be exposed to and adversely affected by HCTT explosive use. Due to the lack of available density or abundance information for California Central Valley DPS steelhead in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of steelhead likely to be adversely affected, based on the best available information

on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect.

A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to exposure of this DPS to explosives. Most steelhead exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. While we do anticipate a very small number of Southern California DPS steelhead mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Since no spawning or freshwater rearing habitat will be affected by the HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. In addition, it is presumed adult steelhead not harmed or killed could continue to spawn in future years and produce juveniles to replace any individuals lost during HCTT activities. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect this DPS. Therefore, the abundance, distribution, and reproduction of Southern California DPS steelhead is not likely to be appreciably reduced by HCTT activities in the action area.

One of the objectives of the Southern California steelhead recovery plan (NMFS 2012b) is to increase abundance of steelhead to viable population levels, including the expression of all life history forms and strategies. While we do anticipate a small number of Southern California steelhead would be killed or seriously injured by HCTT explosives, the mortality level anticipated would not result in an appreciable reduction in the numbers or reproductive rate of this DPS. Thus, we do not expect this level of mortality to impact the survival or recovery of Southern California steelhead.

In summary, 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 Action Proponents will conduct in the HCTT 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 Southern California DPS steelhead in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Southern California DPS steelhead in the wild. Thus, we determine that the action would not appreciably reduce the likelihood of both survival and recovery of Southern California DPS steelhead in the wild.

8.1.20 Steelhead – California Central Valley DPS

California Central Valley DPS steelhead overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area,

we determined that a small number of California Central Valley DPS steelhead are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to steelhead exposure to explosives.

Due to the lack of available density or abundance information for California Central Valley DPS steelhead in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of steelhead likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of California Central Valley DPS steelhead is expected to be minimal due to the: 1) limited spatial overlap of this DPS' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this DPS in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most steelhead exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult steelhead not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of California Central Valley DPS steelhead mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect California Central Valley DPS steelhead. The abundance, distribution, and reproduction of California Central Valley DPS steelhead are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for California Central Valley DPS steelhead, as identified in the NMFS recovery plan (NMFS 2014b), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this DPS will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 California Central Valley DPS steelhead in the wild by

reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of California Central Valley DPS steelhead in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of California Central Valley DPS steelhead in the wild.

8.1.21 Steelhead – Central California Coast DPS

Central California Coast DPS steelhead overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Central California Coast DPS steelhead are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to steelhead exposure to explosives.

Due to the lack of available density or abundance information for Central California Coast DPS steelhead in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of steelhead likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Central California Coast DPS steelhead is expected to be minimal due to the: 1) limited spatial overlap of this DPS' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this DPS in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most steelhead exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult steelhead not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Central California Coast DPS steelhead mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Central California Coast DPS steelhead. The abundance, distribution, and reproduction of Central California Coast DPS steelhead are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Central California Coast DPS steelhead, as identified in the NMFS recovery plan (NMFS 2014b; NMFS 2016c), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this DPS will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Central California Coast DPS steelhead in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Central California Coast DPS steelhead in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Central California Coast DPS steelhead in the wild.

8.1.22 Steelhead – Northern California DPS

Northern California DPS steelhead overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Northern California DPS steelhead are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to steelhead exposure to explosives.

Due to the lack of available density or abundance information for Northern California DPS steelhead in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of steelhead likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Northern California DPS steelhead is expected to be minimal due to the: 1) limited spatial overlap of this DPS' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this DPS in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most steelhead exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult steelhead not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities.

Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Northern California DPS steelhead mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Northern California DPS steelhead. The abundance, distribution, and reproduction of Northern California DPS steelhead are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Northern California DPS steelhead, as identified in the NMFS recovery plan (NMFS 2016c), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this DPS will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Northern California DPS steelhead in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Northern California DPS steelhead in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Northern California DPS steelhead in the wild.

8.1.23 Steelhead – South-Central California Coast DPS

South-Central California Coast DPS steelhead overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of South-Central California Coast DPS steelhead are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to steelhead exposure to explosives.

Due to the lack of available density or abundance information for South-Central California Coast DPS steelhead in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of steelhead likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is

anticipated to be exposed at levels resulting in an adverse effect. The lethal take of South-Central California Coast DPS steelhead is expected to be minimal due to the: 1) limited spatial overlap of this DPS' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this DPS in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most steelhead exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult steelhead not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of South-Central California Coast DPS steelhead mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect South-Central California Coast DPS steelhead. The abundance, distribution, and reproduction of South-Central California Coast DPS steelhead are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for South-Central California Coast DPS steelhead, as identified in the NMFS recovery plan (NMFS 2013c), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this DPS will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 South-Central California Coast DPS steelhead in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of South-Central California Coast DPS steelhead in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of South-Central California Coast DPS steelhead in the wild.

8.1.24 Chinook Salmon – California Coastal ESU

California Coastal ESU Chinook salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area,

we determined that a small number of California Coastal ESU Chinook salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to Chinook salmon exposure to explosives.

Due to the lack of available density or abundance information for California Coastal ESU Chinook salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of Chinook salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of California Coastal ESU Chinook salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most Chinook salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult Chinook salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of California Coastal ESU Chinook salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect California Coastal ESU Chinook salmon. The abundance, distribution, and reproduction of California Coastal ESU Chinook salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for California Coastal ESU Chinook salmon, as identified in the NMFS recovery plan (NMFS 2014b; NMFS 2016b; NMFS 2016f), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 California Coastal ESU Chinook salmon in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of California Coastal ESU Chinook salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of California Coastal ESU Chinook salmon in the wild.

8.1.25 Chinook Salmon – Central Valley Spring-Run ESU

Central Valley spring-run ESU Chinook salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Central Valley spring-run ESU Chinook salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to Chinook salmon exposure to explosives.

Due to the lack of available density or abundance information for Central Valley spring-run ESU Chinook salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of Chinook salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Central Valley spring-run ESU Chinook salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most Chinook salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult Chinook salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Central Valley spring-run ESU Chinook salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects

analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Central Valley spring-run ESU Chinook salmon. The abundance, distribution, and reproduction of Central Valley spring-run ESU Chinook salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Central Valley spring-run ESU Chinook salmon, as identified in the NMFS recovery plan (NMFS 2014b; NMFS 2014a), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Central Valley spring-run ESU Chinook salmon in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Central Valley spring-run ESU Chinook salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Central Valley spring-run ESU Chinook salmon in the wild.

8.1.26 Chinook Salmon – Sacramento River Winter-Run ESU

Sacramento River winter-run ESU Chinook salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Sacramento River winter-run ESU Chinook salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to Chinook salmon exposure to explosives.

Due to the lack of available density or abundance information for Sacramento River winter-run ESU Chinook salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of Chinook salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Sacramento River winter-run ESU Chinook salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively

small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most Chinook salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult Chinook salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Sacramento River winter-run ESU Chinook salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Sacramento River winter-run ESU Chinook salmon. The abundance, distribution, and reproduction of Sacramento River winter-run ESU Chinook salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Sacramento River winter-run ESU Chinook salmon, as identified in the NMFS recovery plan (NMFS 2014b), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Sacramento River winter-run ESU Chinook salmon in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Sacramento River winter-run ESU Chinook salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Sacramento River winter-run ESU Chinook salmon in the wild.

8.1.27 Coho Salmon - Central California Coast ESU

Central California Coast ESU coho salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Central California Coast ESU coho salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including

mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to coho salmon exposure to explosives.

Due to the lack of available density or abundance information for Central California Coast ESU coho salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of coho salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Central California Coast ESU coho salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most coho salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult coho salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Central California Coast ESU coho salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Central California Coast ESU coho salmon. The abundance, distribution, and reproduction of Central California Coast ESU coho salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Central California Coast ESU coho salmon, as identified in the NMFS recovery plan (NMFS 2012a; NMFS 2014b), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Central California Coast ESU coho salmon in the wild by

reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Central California Coast ESU coho salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Central California Coast ESU coho salmon in the wild.

8.1.28 Coho Salmon - Oregon Coast ESU

Oregon Coast ESU coho salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Oregon Coast ESU coho salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to coho salmon exposure to explosives.

Due to the lack of available density or abundance information for Oregon Coast ESU coho salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of coho salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Oregon Coast ESU coho salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most coho salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult coho salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Oregon Coast ESU coho salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Oregon Coast ESU coho salmon. The abundance, distribution, and reproduction of Oregon Coast ESU coho salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Oregon Coast ESU coho salmon, as identified in the NMFS recovery plan (NMFS 2014a; NMFS 2016f), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Oregon Coast ESU coho salmon in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Oregon Coast ESU coho salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Oregon Coast ESU coho salmon in the wild.

8.1.29 Coho Salmon - Southern Oregon and Northern California Coasts ESU

Southern Oregon and Northern California Coast ESU coho salmon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of Southern Oregon and Northern California Coast ESU coho salmon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to coho salmon exposure to explosives.

Due to the lack of available density or abundance information for Southern Oregon and Northern California Coast ESU coho salmon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information is not available to estimate the exact number of coho salmon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of Southern Oregon and Northern California Coast ESU coho salmon is expected to be minimal due to the: 1) limited spatial overlap of this ESUs' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this ESU in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for explosives used in the NOCAL RC.

Most coho salmon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness

or to the local population. Any adult coho salmon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of Southern Oregon and Northern California Coast ESU coho salmon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this ESU within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this ESU. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect Southern Oregon and Northern California Coast ESU coho salmon. The abundance, distribution, and reproduction of Southern Oregon and Northern California Coast ESU coho salmon are not expected to be appreciably reduced by HCTT activities.

Recovery objectives for Southern Oregon and Northern California Coast ESU coho salmon, as identified in the NMFS recovery plan (NMFS 2012a; NMFS 2014a), are not likely to be impacted by this action because we anticipate only a very small number of steelhead from this ESU will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 Southern Oregon and Northern California Coast ESU coho salmon in the wild by reducing the reproduction, numbers, or distribution of this population. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Southern Oregon and Northern California Coast ESU coho salmon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of Southern Oregon and Northern California Coast ESU coho salmon in the wild.

8.1.30 Eulachon – Southern DPS

Southern DPS eulachon overlap with a very small portion of the HCTT action area in the NOCAL RC. Despite this population's anticipated low abundance in the action area, we determined that a small number of southern DPS eulachon are likely to be exposed to and adversely affected by HCTT explosive use. A range of impacts including mortality, barotrauma (non-auditory injury), hearing loss (TTS), and behavioral responses are likely to occur due to exposure of eulachon to explosives.

Due to the lack of available density or abundance information for southern DPS eulachon in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish that would be exposed to stressors from explosives. While information

is not available to estimate the exact number of eulachon likely to be adversely affected, based on the best available information on species distribution and HCTT explosive use, only a small percentage of the total population within the HCTT action area is anticipated to be exposed at levels resulting in an adverse effect. The lethal take of southern DPS eulachon is expected to be minimal due to the: 1) limited spatial overlap of this DPS' range with the action area (i.e., only in the NOCAL RC); 2) infrequent and isolated use of explosives (particularly in-water explosives) in the NOCAL RC; 3) anticipated low density of this species in areas where explosives would be used; and 4) relatively small NEWs and ranges to effects resulting in fish mortality for HCTT explosives in the NOCAL RC.

Most eulachon exposures to explosives are expected to result in short-term behavioral or physiological stress responses or TTS, with no long-term consequences to the individual's fitness or to the local population. Any adult eulachon not harmed or killed could continue to reproduce in future years and produce juveniles to replace any individuals lost during HCTT activities. Since no spawning or freshwater rearing habitat will be affected by HCTT activities, impacts on spawning survival and survival from egg to juvenile are not expected. While we do anticipate a very small number of southern DPS eulachon mortalities and injuries (some with fitness consequences) from explosives, the mortality and injury levels predicted will not result in an appreciable reduction in the numbers or reproductive rate of this DPS within the action area or range wide. Thus, we do not expect this level of mortality and injury to impact the survival or recovery of this DPS. Besides explosives, based on our effects analysis, none of the other stressors resulting from the proposed action are likely to adversely affect southern DPS eulachon. The abundance, distribution, and reproduction of southern DPS eulachon are not expected to be appreciably reduced by HCTT activities.

The stated goals of the recovery plan (NMFS 2017c) for the southern DPS of eulachon are to: 1) Increase the abundance and productivity of eulachon; 2) Protect and enhance the genetic, life history, and spatial diversity of eulachon throughout its geographical range; and 3) Reduce existing threats to warrant delisting of the species. Included in the criteria for delisting this DPS are that each of the four subpopulations is self-sustaining (i.e., each subpopulation has less than 5% probability of extinction in 100 years) and has a stable or increasing growth rate greater than one across multiple generations. These recovery objectives are not likely to be impacted by this action because we anticipate only a very small number of southern DPS eulachon will experience mortality or reduced fitness as a result.

In summary, 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 Action Proponent will conduct in the HCTT 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 southern DPS eulachon in the wild by reducing the reproduction, numbers, or distribution of that species. Similarly, the effects from ongoing HCTT activities continuing into the reasonably foreseeable future would not be expected, directly or

indirectly, to appreciably reduce the likelihood of recovery of southern DPS eulachon in the wild. Thus, we determine that the proposed action would not appreciably reduce the likelihood of both survival and recovery of southern DPS eulachon in the wild.

8.2 Destruction and Adverse Modification Analysis

8.2.1 Main Hawaiian Islands Insular DPS False Killer Whale Critical Habitat

In Section 6.1.3, we determined that the effects of vessel noise, aircraft noise, weapons noise, physical disturbance and strike, explosives, and air guns on the PBF of MHI IFKW designated critical habitat are either insignificant or discountable. In Section 6.1.3, we determined that the effects of HCTT sonar and other transducer use on the prey resource characteristic of designated critical habitat are insignificant (i.e., so minor that the effect cannot be meaningfully evaluated). In Section 6.1.3, we determined that sonar and other transducers will result in adverse effects to the PBF, specifically to the ability of MHI IFKWs to use or occupy designated critical habitat. The discussion below evaluates whether sound levels resulting in impacts to MHI IFKW use or occupancy would be introduced into the critical habitat such that the effects of the action will appreciably diminish the value of critical habitat for the conservation of the MHI IFKW. When evaluating whether sonar has appreciably decreased the value of critical habitat for conservation, we consider timing, frequency, and duration of the sound and its resulting impacts on the population, as well as the spatial extent of impacts when they occur, relative to the area designated as critical habitat.

Based on our effects analysis presented above (Section 6.1.3), we anticipate that sound from HCTT sonar would be introduced into MHI IFKW designated critical habitat infrequently at levels that would result in impairment of MHI IFKW habitat use or occupancy (i.e., average of 0.23 disruptions per animal annually in critical habitat). In addition to significant behavioral responses and TTS, some exposures to sonar within critical habitat are likely to result in masking of important biological sounds, which could impair MHI IFKW foraging and communication. Consideration of behavioral response to individual animals as well as the population are important to our understanding of whether there will be disruptions of use or occupancy of the area, and if so whether the disruptions will diminish the value of habitat for conservation of the species in a meaningful way.

We do not anticipate that MHI IFKWs will experience long duration or repeat exposures within a short period of time in designated critical habitat due to the following: 1) the large area of habitat available to and utilized by the animals (NMFS 2018c), 2) the relatively small area that will be affected by sonar at any given period of time, (3) that most HCTT activities resulting in significant disruptions of habitat use last less than one day, and (4) that long duration (i.e., more than one day) HCTT 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), largely outside of designated critical habitat. This decreases the likelihood that MHI IFKWs and HCTT activities will co-occur for extended periods of time or repetitively over the duration of an activity (or subsequent activities) while within critical habitat. Based on the available literature that indicates infrequent exposures resulting in behavioral disruptions are unlikely to impact an

individual's overall energy budget (Southall et al. 2007; New et al. 2014; King et al. 2015; Villegas-Amtmann et al. 2015; Harris et al. 2017; U.S. Department of the Navy 2017; Farmer et al. 2018), we do not expect this level of exposure to result in long-term fitness consequences for exposed animals.

As discussed in Section 6.1.1, some limited masking could occur due to the Action Proponent's use of sonar and other transducers when animals are in close enough proximity. Because most HCTT sonar sources typically have low duty cycles, the effects of such masking are expected to be limited. However, our effects analysis predicts impacts to MHI IFKWs within critical habitat from a variety of activities involving multiple sonar bins (including broadband sources) that could potentially include high duty or continuous active sonar sources. While the lower source levels of these systems limit the range of impact (compared to more traditional systems), MHI IFKWs close to the sonar source could experience masking on a longer time scale than those exposed to traditional (i.e., low duty cycle) sonars. However, even with high duty and continuous sonar sources, the duration of masking would be limited since both the sonar source and the animal are moving within the environment. Thus, we expect that HCTT sonar will result in a limited amount of masking that could impair MHI IFKW critical habitat use or occupancy.

As described above, we use our analysis of individual animal responses to HCTT sonar use to indicate whether behavioral disturbances could impair use or occupancy of the habitat at a population level. Based on the preceding analysis considering effects to individual animals, we do not anticipate HCTT sonar use would impair use of occupancy of the habitat at a population level. Given that impairment of use and occupancy will not have population level consequences (and HCTT activities occur throughout the range of MHI IFKWs and throughout all areas of designated critical habitat), we find that this is an indicator that diminishment of the value of critical habitat for conservation of the species should not be appreciable.

The limited duration of the disruption further supports the foregoing finding. Based on the best available information regarding exposure of MHI IFKWs and other marine mammal species to military sonar (Falcone et al. 2009; Tyack et al. 2011a; Falcone and Schorr 2014; e.g., Baird et al. 2016; e.g., Baird et al. 2017a), these short term disruptions of behavior are not anticipated to lead to long duration (e.g., more than one day) abandonment or avoidance of any particular area within critical habitat, or of critical habitat as a whole. Even with decades of military sonar use within areas of recently designated critical habitat, as well as monitoring of MHI IFKWs use of these habitats, we do not have information to suggest that these animals exhibit long duration avoidance of particular areas of critical habitat, or critical habitat as a whole, due to sonar use. Based on the range to effects values described previously and the extent of designated critical habitat, when sonar exposure does occur in portions of critical habitat, only a small portion of the habitat area will be exposed at levels that could result in impacts to use and occupancy, and the vast majority of critical habitat would remain available to the animals for undisrupted use and occupancy. Sounds from sonar used by the Action Proponents are transient, and do not persist in the marine environment at times when the sonar source is not being operated.

Scenarios where HCTT sonar and other transducer use within the action area are likely to result in changes to the animal's use or occupancy of MHI IFKW critical habitat are anticipated to be sporadic and temporary. The Navy has been conducting training and testing activities in the HRC and within recently designated critical habitat for decades. Despite this, monitoring (e.g., Baird et al. 2013; Baird et al. 2014; Baird et al. 2017a) indicates that MHI IFKWs continue to utilize these habitats. Further, the Action Proponents have proposed geographic mitigation for several areas in the HRC (see Section 2.4.2), much of which is encompassed by MHI IFKW designated critical habitat. These areas include some of the habitats most frequently used by MHI IFKWs in the Main Hawaiian Islands. This will minimize exposure of designated critical habitat and MHI IFKWs within critical habitat to sonar that could result in impairment of the use or occupancy of the habitat. For these reasons, while we have determined that MHI IFKW's use of designated critical habitat is likely to be disrupted intermittently in several locations, we do not anticipate that either individuals or the population would avoid or abandon use of the habitat for important biological functions.

In summary, while we have determined that MHI IFKW's use of designated critical habitat is likely to be disrupted intermittently at varying locations both annually and into the reasonably foreseeable future, we do not anticipate that either individuals or the population would avoid or abandon use of the habitat for important biological functions. Evidence of individual responses to noise created by HCTT sonar and other transducers within or affecting critical habitat, as well as consideration of the temporal and spatial extent of impacts, indicates that, while sound from sonar and other transducers may impact the value of critical habitat for the conservation of the species, it is unlikely that the population's use or occupancy of critical habitat would be impaired to an extent that the conservation value of the critical habitat would be appreciably diminished. As a result, we find that the introduction of noise by HCTT sonar and other transducers is unlikely to appreciably diminish the value of critical habitat as a whole for the conservation of MHI IFKWs.

9 CONCLUSION

After reviewing and analyzing the current status of the listed species, the environmental baseline within the action area, the consequences of the proposed action and associated activities, and the cumulative effects, it is NMFS's biological opinion that the proposed action is not likely to jeopardize the continued existence of the following ESA-listed species: blue whale; fin whale; Western North Pacific DPS gray whale; Mexico DPS humpback whale; Central America DPS humpback whale; sei whale; sperm whale; Main Hawaiian Islands Insular DPS false killer whale; Guadalupe fur seal; Hawaiian monk seal; Central North Pacific DPS green sea turtle; East Pacific DPS green sea turtle; olive ridley sea turtle – Mexico's Pacific coast population; olive ridley sea turtle – all other populations; hawksbill sea turtle; leatherback sea turtle; North Pacific DPS loggerhead sea turtle; Southern DPS eulachon; Southern California DPS steelhead; California Central Valley DPS steelhead; Central California Coast DPS steelhead; Northern California DPS steelhead; South-Central California Coast DPS steelhead; California Coastal ESU Chinook; Central Valley Spring-Run ESU Chinook; Sacramento River Winter-Run ESU

Chinook; Central California Coast ESU coho; Oregon Coast ESU coho; Southern Oregon and Northern California Coasts ESU coho; Eastern Pacific DPS scalloped hammerhead shark; oceanic whitetip shark; and giant manta ray.

After reviewing and analyzing the current status of the designated critical habitat, the environmental baseline within the action area, the consequences of the proposed action and associated activities, and the cumulative effects, it is NMFS's biological opinion that the proposed action is not likely to destroy or adversely modify the designated critical habitat of the Main Hawaiian Islands Insular DPS false killer whale.

In Section 4.1.2 we determined the effects of exposure were discountable or a response would be insignificant and, therefore, the effects of the proposed action may affect, but are not likely to adversely affect the following ESA-listed or proposed species: Southern Resident killer whale; North Pacific right whale; Southern DPS green sturgeon; black abalone; white abalone; and sunflower sea star.

In Section 4.1.3 we determined the effects of the proposed action may affect, but is not likely to adversely affect designated critical habitat of the following species: Mexico DPS humpback whale; Central America DPS humpback whale; Southern Resident killer whale; Hawaiian monk seal; leatherback sea turtle; Southern DPS green sturgeon; and black abalone.

10 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. "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 actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering (50 CFR §222.102). NMFS has defined the term "harass" in a policy directive (PD 02-110-19) as to *create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding or sheltering*. "Incidental take" is defined by regulation as takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR §402.02). Section 7(b)(4) and section 7(o)(2) of the ESA, as well as in regulation at 50 CFR §402.14(i)(6) 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.

When an action is reasonably certain to result in incidental take of ESA-listed marine mammals, section 7(b)(4) of the ESA requires that such taking be authorized under section 101(a)(5) of the MMPA in order for the Secretary to issue an ITS for ESA-listed marine mammals and that the ITS specify the measures necessary to comply with section 101(a)(5) of the MMPA. As part of

the proposed action, the Permits Division proposes to issue LOAs. Accordingly, the terms of this ITS and the exemption from Section 9 of the ESA become effective only upon the issuance of the 101(a)(5) authorization.

The marine mammal portion of this ITS is a preliminary statement. It will become final and effective upon the issuance of the authorization under section 101(a)(5) of the Marine Mammal Protection Act and NMFS's written confirmation to the Federal agency, for the duration of the authorization and with relevant revisions per the final authorization.

10.1 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 CFR §402.14(i)(1)(i)). In the opinion, we determined that incidental take of ESA-listed species is reasonably certain to occur as a result of the proposed action. The amount or extent of take that will occur as a result of the proposed action was estimated based on the best information available. In some instances, as noted below, we used a surrogate for the amount or extent of take anticipated. When it is not possible or practicable to specify the amount or extent of take, a surrogate may be used if we: 1) describe the causal link between the surrogate and take of the listed species, 2) 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 3) set a clear standard for determining when the level of anticipated take has been exceeded (50 CFR §402.14(g)(7)).

10.1.1 Marine Mammals and Sea Turtles

Table 62 summarizes the number and types of take of ESA-listed marine mammals and sea turtles anticipated to occur over a maximum year of HCTT activities from acoustic and explosive stressors, and over every seven years from vessel strike. Table 63 summarizes the number and types of take of ESA-listed marine mammals and sea turtles anticipated to occur every seven years of HCTT activities from acoustic and explosive stressors.

As discussed in Section 6.2.1, based on the available sea turtle density information, it is not possible, nor would it be an accurate representation of potential effects, to quantify take (i.e., in the form of the number of turtles suffering mortality, injury, TTS, and behavioral disruption) or to monitor impacts in terms of ESA-listed sea turtles less than 40 cm, affected by acoustic stressors resulting from the proposed action. Because it is not practical to express the amount of anticipated take of smaller sea turtles from other species, we instead use a surrogate measure to express the amount or extent of incidental take. The surrogate for the incidental take of ESA-listed sea turtles less than 40 cm 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., significant behavioral response, TTS, AINJ, non-auditory injury, and mortality) from sonar and explosives in those areas occupied sea turtles (less than 40 cm), and 2) sonar and explosive activity levels, which is

Table 62. Number and Type of ESA Takes of Marine Mammals and Sea Turtles Anticipated to Occur as a Result of HCTT Over a Maximum Year of Activities from Acoustic and Explosive Stressors, and Over Every Seven Years from Vessel Strike

ESA-Listed Species	Acoustic and Explosive Stressors: Annual Take During Maximum Year of Activities				Vessel Strike: Take Every 7-year Period
	Harassment: TTS / Significant Behavioral Response	Harm: Auditory Injury (PTS)	Harm: Non-Auditory Injury	Mortality	Mortality[a]
Marine Mammals					**Navy / Coast Guard**
Blue Whale	3,199 / 1,464	28	0	0	1 / 1
Fin Whale	9,862 / 3,725	55	1	0	5 / 1
Humpback Whale – Central America DPS	1,341 / 547	19	0	0	1/ 1
Humpback Whale – Mexico DPS	3,175 / 1,274	43	1	0	1/ 1
Gray Whale – Western North Pacific DPS	97 / 72	2	0	0	0 / 0
Sei Whale	434 / 121	5	0	0	1 / 1
Sperm Whale	1,304 / 4,236	4	0	0	1 / 0
False Killer Whale – Main Hawaiian Islands Insular DPS	64 / 105	0	0	0	0 / 0
Hawaiian Monk Seal	198 / 1,051 [b]	5	1	0	0 / 0
Guadalupe Fur Seal	81,354 / 266,199	51	3	1	0 / 0
Sea Turtles					**Navy and Coast Guard Combined**
Green [c] – East Pacific DPS	638 / 53	9	1	0	47
Green [c] - Central North Pacific DPS	1,171 / 2,068	45	3	1	21
Hawksbill [c]	18 / 19	1	0	0	3
Loggerhead [c] – North Pacific Ocean DPS	727 / 135	9	2	0	2
Olive Ridley [c]	224 / 37	5	0	0	7
Leatherback [c]	384 / 49	6	0	0	0

[a] No more than six total Navy vessel or USV strikes from all ESA-listed species (or DPS's) combined are exempted over each 7-year period of the proposed action. No more than one total Coast Guard vessel strike from all ESA-listed species (or DPS's) combined is exempted over each 7-year period of the proposed action.

[b] Estimated Hawaiian monk seal harassment takes include up to 360 annual instances of significant behavioral response due to in-air acoustic effects from PMRF weapons firing noise.

[c] A surrogate was used for take of sea turtles less than 40 cm from acoustic and explosive stressors (see discussion above).

the total number of hours or count (by sonar or explosive bin type) that would be used in the action area as part of the proposed action.

Predicted ranges to effects for ESA-listed sea turtles are calculated using NAEMO based upon sound exposure criteria, which are used to predict the onset of sea turtle mortality, injury (auditory and non-auditory), or TTS. Ranges to effects to sea turtles from exposure to sonar and other transducers, airguns, and explosives are provided in the *Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area* (Navy 2024h), and are incorporated into this opinion by reference (Section 3.4, pp. 3-30 through 3-39). Sea turtles within these ranges could be exposed above the effect threshold levels, and a portion may experience those effects. Ranges to effects, in combination with sonar and explosive activity levels (i.e., the number of explosives used or sonar hours, by bin/sonar type, in portions of the action area), can be used to estimate the area within the water column within which ESA-listed sea turtles less than 40 cm would be exposed to a sound source at levels resulting in the anticipated effects (i.e., mortality, injury, AINJ, TTS). Thus, there is a causal link between the surrogate factors above (i.e., range to effects and activity level) and take of hawksbill, green (Central North Pacific and East Pacific DPSs), olive ridley (Mexico Pacific Coast Breeding Colonies and All Other Areas), loggerhead (North Pacific DPS), and leatherback sea turtles less than 40 cm, 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 sea turtles to impulsive and non-impulsive sound sources, or updated sound exposure criteria levels based on new information. Any increase in the predicted ranges to effects on sea turtles from explosives used in the action area would result in a larger area within the water column within which sea turtles 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.

Table 63. Number and Type of ESA Takes of Marine Mammals and Sea Turtles Anticipated to Occur as a Result of HCTT Over Every Seven Years from Acoustic and Explosive Stressors (Navy 2025i; Navy 2025h)

ESA-Listed Species	Acoustic and Explosive Stressors Take Every 7-Year Period			
	Harassment: TTS / Significant Behavioral Response	Harm: Auditory Injury (PTS)	Harm: Non-Auditory Injury	Mortality
Blue Whale	16,727 / 8,605	152	0	0
Fin Whale	47,566 / 21,479	300	1	0
Gray Whale – Western North Pacific DPS	418 / 434	5	0	0
Humpback Whale – Mexico DPS	15,669 / 7,701	219	1	0
Humpback Whale – Central America DPS	6,593 / 3,305	96	0	0
Sei Whale	2,334 / 714	14	0	0
Sperm Whale	6,608 / 23,617	6	0	0
Guadalupe Fur Seal	409,620 / 1,491,214	284	16	1
Hawaiian Monk Seal	1,231 / 7,164 [a]	24	1	0
False Killer Whale – Main Hawaiian Islands Insular DPS	372 / 637	0	0	0
Sea Turtles				
Green [b] – East Pacific DPS	3,941 / 350	56	1	0
Green [b] - Central North Pacific DPS	7,967 / 14,384	303	11	5
Hawksbill [b]	112 / 125	2	0	0
Loggerhead [b] – North Pacific Ocean DPS	4,334 / 830	50	7	0
Olive Ridley [b]	1,385 / 180	13	0	0
Leatherback [b]	2,360 / 245	19	0	0

[a] Estimated Hawaiian monk seal harassment takes over every seven years include up to 2,520 instances of significant behavioral response due to in-air acoustic effects from PMRF weapons firing noise.
[b] A surrogate was used for take of sea turtles less than 40 cm from acoustic and explosive stressors (see discussion above).

10.1.2 Fishes

As described in Section 6.3.1, due to the lack of available density or abundance information for ESA-listed fish species in the portion of the action area where HCTT explosives would be used, we cannot quantify the total number of individual fish from each species that would be exposed to stressors from 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 the number of fish suffering mortality, injury, TTS, and behavioral disruption) or to monitor take-related impacts in terms of individuals of the following ESA-listed fish species: giant manta ray; oceanic whitetip shark; Eastern Pacific DPS scalloped hammerhead shark; Southern California DPS steelhead; California Central Valley DPS steelhead; Central California Coast DPS steelhead; Northern California DPS steelhead; South-Central California Coast DPS steelhead; California Coastal ESU Chinook; Central Valley Spring-Run ESU Chinook; Sacramento River Winter-Run ESU Chinook; Central California Coast ESU coho; Oregon Coast ESU coho; Southern Oregon and Northern California Coasts ESU coho; and southern DPS eulachon.

Because it is not practical to express the amount of anticipated take of these fish species, we instead use a surrogate measure to express the amount or extent of incidental take. The surrogate for the incidental take of the ESA-listed fish listed above 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 adults of these species, and 2) explosive activity level, which is the total number of explosives (by bin type) that would be detonated in the action area as part of the proposed action (see Section 6.3.1 for details on range to effects).

Predicted ranges to effects for ESA-listed fish species are calculated using NAEMO based upon sound exposure criteria which are used to predict the onset of mortality, injury, or TTS within fishes within the HCTT action area. As discussed above for sea turtles, 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 fish species as a result of the proposed action. The surrogate for take of ESA-listed fish would be exceeded if new information indicates larger ranges to effects than predicted for our effects analysis, or activity

levels involving the use of explosives are greater than the maximum levels proposed by the Action Proponents.

10.1.3 Activity Levels as Indicators of Take Resulting from Acoustic Stressors

As discussed in this opinion, the estimated take of ESA-listed marine mammals, sea turtles, and fish[30] from acoustic stressors is based on the NAEMO modeling, which represents the best available means of numerically quantifying take. As the level of modeled sonar, explosive or air gun use increases, the level of take is likely to increase as well. For take from these acoustic stressors, feasible monitoring techniques for detecting and calculating actual take at the scale of HCTT 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 the NAEMO methodology, and the link between sonar, explosive, and air gun 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 Action Proponents to report to NMFS any exceedance of activity levels (for any activity associated with take) 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 Action Proponents to reinitiate consultation.

10.2 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 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 such impacts, reasonable and prudent measures, and term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any 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" refer to those actions that NMFS and USFWS consider necessary or appropriate to minimize the impact of the incidental take on the species (50 CFR §402.02). Reasonable and prudent measures, along with the terms and conditions that implement them, cannot alter the basic design, location, scope, duration, or timing of the action, may involve only minor changes, and may include measures implemented inside or outside of the action area that avoid, reduce, or offset the impact of incidental take (50 CFR 402.14(i)(2)).

[30] For estimating take of ESA-listed fish, and hawksbill, green, olive ridley, loggerhead, and leatherback sea turtles less than 40 cm, NMFS only used the ranges to effects modeled by the Navy's NAEMO model.

We believe the following RPMs are necessary and appropriate:

1. The Action Proponents (i.e., Navy, Coast Guard, Army, and Air Force) and NMFS's Permits Division shall minimize effects to ESA-listed species from the use of active sonar, explosives, air guns, and vessels during training and testing activities.
2. The Action Proponents and NMFS's Permits Division shall monitor and report to NMFS's OPR ESA Interagency Cooperation Division on impacts to ESA-listed species from the use of sonar and other transducers, air guns, explosives, and vessels during training and testing activities. This includes adherence to the monitoring and reporting measures specified in the final MMPA rule and LOAs.
3. The Action Proponents shall work with NMFS to improve on the available density information for the MHI IFKW DPS.

We have not identified any other measures that are available to implement at this time that meet the parameters of 402.14(i)(2) and that would avoid, reduce or offset the impact of incidental take.

10.3 Terms and Conditions

To be exempt from the prohibitions of section 9 and any 4(d) rule that extends the protections of section 9 to threatened species, the Action Proponents and NMFS Permits Division must comply with the terms and conditions described below. The Action Proponents and NMFS Permits Division have a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR §402.14(i)(4)). Any incidental take that is identified in this ITS and is in compliance with the terms and conditions identified in this section is not a prohibited taking under the ESA (16 U.S.C. § 1536(o)), and no other authorization or permit under the ESA is required (50 CFR §402.14(i)(6)).

The following terms and conditions implement reasonable and prudent measure #1:

a) The NMFS Permits Division shall ensure that all mitigation measures as prescribed in the updated final MMPA rule and LOAs are implemented by the Action Proponents.

b) The Action Proponents shall continue to request technical assistance and implement adaptive management efforts in collaboration with NMFS to help inform future consultations on training and testing in the action area. Adaptive management discussions may include reviewing the feasibility of potential new measures to increase mitigation effectiveness.

The following terms and conditions implement reasonable and prudent measure #2:

a) The Navy, on behalf of the Action Proponents, shall monitor training and testing activities and submit reports (classified/unclassified) annually to the NMFS Permits Division and the 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 of whether 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 rule and LOAs.

b) The Action Proponents 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 Action Proponents shall report when enough data are available to determine if the dead or seriously injured ESA-listed species may be attributable to HCTT activities, including but not limited to, the use of explosives and vessel strike.

c) In the event that Action Proponent personnel (uniformed military, civilian, or contractors while conducting Action Proponent work) discover a live or dead stranded marine mammal or sea turtle within the action area or on Action Proponents' property, they shall report the incident to NMFS immediately or as soon as operational security considerations allow.

d) If NMFS personnel determine that the circumstances of any of the strandings reported suggest investigation of the associated Action Proponent 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 Action Proponents 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. The Action Proponents will submit this information as soon as possible, but no later than 7 business days after the request.

e) In the event of a cetacean live stranding (or near-shore atypical milling) event within the HCTT action area or within 50 km (27 nmi) of the boundary of the action area, where the NMFS Marine Mammal Stranding Network is engaged in herding or other interventions to return animals to the water, NMFS OPR will advise the Action Proponents of the need to implement shutdown procedures for all active acoustic sources or explosive devices within 50 km of the stranding. Following this initial shutdown, NMFS will communicate with the Action Proponents to determine whether circumstances support modification of the shutdown zone. The Action Proponents may decline to implement all or part of the shutdown if the holder of the LOA, or his/her designee, determines that it is necessary for national security. Shutdown procedures for live stranding or milling cetaceans include the following:

   (1) If at any time, the marine mammal(s) die or are euthanized, or if herding/intervention efforts are stopped, NMFS will immediately advise that the shutdown around the animals' location is no longer needed;

(2) Otherwise, shutdown procedures will remain in effect until NMFS determines and advises that all live animals involved have left the area (either of their own volition or following an intervention); and

(3) If further observations of the marine mammals indicate the potential for re-stranding, additional coordination will be required to determine what measures are necessary to minimize that likelihood (e.g., extending the shutdown or moving operations farther away) and to implement those measures as appropriate.

f) The Action Proponent annual HCTT reports shall include a confirmation that foreign military use of sonar and explosives, when such activities are planned by and under the substantial control and responsibility of the U.S. Navy and, therefore, included in the proposed action (see Section 2.3), combined with the U.S. Navy's use of sonar and explosives, did not cause exceedance of the analyzed levels (within each NAEMO modeled sonar and explosive bin) used for estimating predicted impacts, which formed the basis of our acoustic impacts effects analysis that was used to determine authorized levels of take under the MMPA and exempted take in this ITS. Only foreign military use of sonar and explosives during activities that are planned by and under the substantial control and responsibility of the U.S. Navy fall within this requirement.

The following terms and conditions implement reasonable and prudent measure #3:

a) The Action Proponents shall work with NMFS through the consultation adaptive management process to improve on the uniform density layer that was used to model HCTT Phase IV acoustic and explosive impacts to MHI IFKWs. Satellite tagging data and sightings information suggest that this DPS's use and occupancy of their range is concentrated in particular "high-use" or high conservation value habitat areas (see Figure 30). The goal of this effort should be to develop a more spatially explicit density layer for HCTT Phase V NAEMO estimation of impacts that accounts for the variability in MHI IFKW use and occupancy of their range through space and time.

11 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 an action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 CFR §402.02). The following conservation recommendations should be considered by the Action Proponents to minimize or avoid effects to threatened and endangered species associated with this action:

1. The Action Proponents should continue developing autonomous marine mammal detection technologies to reduce the risk of vessel strike.

2. The Navy should continue to conduct behavioral response studies to obtain 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 military readiness activities.

3. As practicable, the Action Proponents should supplement the proposed visual monitoring mitigation measures with passive acoustic monitoring for additional activities (i.e., beyond those where this type of mitigation is already conducted) that could cause marine mammal injury or mortality.

4. The Navy should research the abundance and distribution of ESA-listed fish species in the action area to incorporate into Navy density models for estimating impacts to fish from acoustic and explosive stressors.

5. The Navy should continue to investigate the use of thermal imaging systems as a potential large whale vessel strike mitigation measure and the compatibility of such systems with HCTT activities with manned and unmanned vessels.

6. The Action Proponents should explore the use of new protocols and/or communication tools to improve on the collection of post-strike information for species identification following a large whale strike in the HCTT action area.

In order for NMFS's OPR Endangered Species Act 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 Action Proponents and NMFS's Permits Division should notify the Endangered Species Act Interagency Cooperation Division of any conservation recommendations they implement in their final action. Notice can be provided to nmfs.hq.esa.consultations@noaa.gov with the ECO number for this consultation (from the signature page) in the subject line.

12 REINITIATION NOTICE

This concludes formal consultation on the Action Proponents' proposed HCTT Phase IV activities and the Permits Division's promulgation of regulations and issuance of incidental take authorizations pursuant to the MMPA. Consistent with 50 CFR §402.16(a), reinitiation of consultation is required and shall be requested by the Federal agency, where discretionary Federal agency involvement or control over the action has been retained or is authorized by law and:

1. If the amount or extent of incidental taking specified in the ITS is exceeded;
2. If new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not previously considered;
3. If the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the opinion; or

4. If a new species is listed or critical habitat designated that may be affected by the identified action.

13 REFERENCES

59 FR 440. 1994. 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 50394. 1999. 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.

70 FR 37160. 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. N. O. A. A. National Marine Fisheries Service, Commerce, editor.

70 FR 52488. 2005. 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.

84 FR 66652. 2019. Endangered and Threatened Species; Determination on the Designation of Critical Habitat for Giant Manta Ray. NOAA, editor.

Abrahms, B., and coauthors. 2019. Memory and resource tracking drive blue whale migrations. Proceedings of the National Academy of Sciences (Online version before inclusion in an issue).

Abreu-Grobois, A., and P. Plotkin. 2008. IUCN SSC Marine Turtle Specialist Group. *Lepidochelys olivacea*.

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-Barrios, R., C. Sabater-Marco, and J. Olivero-Verbel. 2018. Ecotoxicological assessment of perchlorate using in vitro and in vivo assays. Environmental Science and Pollution Research 25(14):13697-13708.

Acevedo-Whitehouse, K., and A. L. J. Duffus. 2009. Effects of environmental change on wildlife health. Philosophical Transactions of the Royal Society of London B Biological Sciences 364(1534):3429-3438.

Adams, D. H., and E. Amesbury. 1998. Occurrence of the manta ray, *Manta birostris*, in the Indian River Lagoon, Florida. Florida Scientist:7-9.

Adams, D. H., and R. Paperno. 2007. Preliminary assessment of a nearshore nursery ground for the scalloped hammerhead off the Atlantic coast of Florida. American Fisheries Society Symposium 50:165-174.

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.

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.

Alava, J. J., and coauthors. 2006. Loggerhead sea turtle (*Caretta caretta*) egg yolk concentrations of persistent organic pollutants and lipid increase during the last stage of embryonic development. Science of The Total Environment 367(1):170-181.

Albert, D. J. 2011. What's on the mind of a jellyfish? A review of behavioural observations on Aurelia sp. jellyfish. Neurosci Biobehav Rev 35(3):474-82.

Allen, C. D., and coauthors. 2017. How many males are enough? Feminization of green sea turtle foraging aggregations in the Pacific. 37th Annual Symposium on Sea Turtle Biology and Conservation, Las Vegas, Nevada.

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.

Anderson, D. M., and coauthors. 2008. Harmful algal blooms and eutrophication: examining linkages from selected coastal regions of the United States. Harmful Algae 8(1):39-53.

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., and coauthors. 2017. Sperm whale long-range echolocation sounds revealed by ANTARES, a deep-sea neutrino telescope. Scientific Reports 7(1):45517.

Andrzejaczek, S., and coauthors. 2018. Temperature and the vertical movements of oceanic whitetip sharks, Carcharhinus longimanus. Scientific Reports 8(1):8351.

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.

Arauz, R., Y. Cohen, J. Ballestero, A. Bolaños, and M. Pérez. 2004. Decline of shark populations in the Exclusive Economic Zone of Costa Rica. International Symposium on Marine Biological Indicators for Fisheries Management. UNESCO, FAO. Paris, France.

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.

Armstrong, A. O., and coauthors. 2016. Prey density threshold and tidal influence on reef manta ray foraging at an aggregation site on the Great Barrier Reef. PLoS ONE 11(5):e0153393.

Arnould, J. P. Y. 2009. Southern fur seals, *Arctocephalus* spp. Pages 1079-1084 *in* B. W. W.F. Perrin, & J. G. M. Thewissen, editor. Encyclopedia of Marine Mammals, 2 edition. Academic Press, Cambridge, Massachusetts.

Associated Press. 2021. Whales found dead under Australian navy ship in San Diego.

Associates, M. 2020. San Diego Bay eelgrass inventory. Prepared for US Navy Region Southwest Naval Facilities Engineering Command and San Diego Unified Port District.

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.

Ataman, A., and coauthors. 2021. Evaluating prevalence of external injuries on nesting loggerhead sea turtles Caretta caretta in southeastern Florida, USA. Endangered Species Research 46:137-146.

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.

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, 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., J. L. Pawloski, P. E. Nachtigall, M. Blonz, and R. C. Gisner. 1995. Echolocation signals and transmission beam pattern of a false killer whale (Pseudorca crassidens). The Journal of the Acoustical Society of America 98(1):51-59.

Au, W. W. L., R. W. Floyd, R. H. Penner, and A. E. Murchison. 1974. Measurement of echolocation signals of the Atlantic bottlenose dolphin, Tursiops truncatus Montagu in open waters. Journal of the Acoustical Society of America 56(4):1280-1290.

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. 2006. Acoustic properties of humpback whale songs. Journal of Acoustical Society of America 120(August 2006):1103-1110.

Au, W. W. L., J. L. Pawloski, T. W. Cranford, R. C. Gisner, and P. E. Nachtigall. 1993. Transmission beam pattern of a false killer whale. (Pseudorca crassidens). Journal of the Acoustical Society of America 93(4 Pt.2):2358-2359. the 125th Meeting of the Acoustical Society of American. Ottawa, Canada. 17-21 May.

Au, W. W. L., A. N. Popper, and R. R. Fay. 2000. Hearing by whales and dolphins. Springer-Verlag, New York.

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., 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).

Austin, B., and X. H. Zhang. 2006. Vibrio harveyi: a significant pathogen of marine vertebrates and invertebrates. Letters in applied microbiology 43(2):119-124.

Avens, L., and coauthors. 2020. Regional comparison of leatherback sea turtle maturation attributes and reproductive longevity. Marine Biology 167(1):1-12.

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., and coauthors. 2021. Hawksbill sea turtle life-stage durations, somatic growth patterns, and age at maturation. Endangered Species Research 45:127-145.

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.

Azanza-Ricardo, J., and coauthors. 2017. Possible Effect of Global Climate Change on Caretta caretta (Testudines, Cheloniidae) Nesting Ecology at Guanahacabibes Peninsula, Cuba. Chelonian Conservation and Biology.

Azzara, A. J., W. M. V. Zharen, and J. J. Newcomb. 2013. Mixed-methods analytic approach for determining potential impacts of vessel noise on sperm whale click behavior. Journal of the Acoustical Society of America 134(6):4566-4574.

Babushina, E. S., G. L. Zaslavskii, and L. I. Yurkevich. 1991. Air and underwater hearing of the northern fur seal: Audiograms and auditory frequency discrimination. Biofizika 36(5):904-907.

Bachman, M. J., J. M. Keller, K. L. West, and B. A. Jensen. 2014. Persistent organic pollutant concentrations in blubber of 16 species of cetaceans stranded in the Pacific Islands from 1997 through 2011. Science of The Total Environment 488–489:115–123.

Badger, J., and coauthors. 2025. Accounting for spatiotemporal sampling bias in a long-term dataset establishes a decline in abundance of endangered false killer whales Pseudorca crassidens in the main Hawaiian Islands. Endangered Species Research.

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.

Baird, R. 2013. Geographic and temporal patterns in the acoustic detection of sperm whales *Physeter macrocephalus* in the central and western North Pacific Ocean. Aquatic Mammals 39(3):253-269.

Baird, R. W. 2016. The lives of Hawai ‘i’s dolphins and whales: natural history and conservation. University of Hawaii press.

Baird, R. W., and coauthors. 2015. Biologically Important Areas for Cetaceans within U.S. Waters - Hawai'i Region. Aquatic Mammals 41(1):54-64.

Baird, R. W., and coauthors. 2023. Field studies and analyses from 2020 through 2022 to support the cooperative conservation and long-term management of main Hawaiian Islands insular false killer whales. Funded by NOAA Section 6 Award NA19NMF4720111, Olympia, WA.

Baird, R. W., and A. M. Gorgone. 2005. False killer whale dorsal fin disfigurements as a possible indicator of long-line fishery interactions in Hawaiian waters. Pacific Science 59(4):593 - 601.

Baird, R. W., and coauthors. 2012. Range and primary habitats of Hawaiian insular false killer whales: An assessment to inform determination of "critical habitat". Endangered Species Research.

Baird, R. W., and coauthors. 2014. Odontocete Studies on the Pacific Missile Range Facility in July/August 2013: Satellite-Tagging, Photo Identification, and Passive Acoustic Monitoring U.S. Navy Pacific Fleet.

Baird, R. W., S. W. Martin, R. Manzano-Roth, D. L. Webster, and B. L. Southall. 2017a. Assessing Exposure and Response of Three Species of Odontocetes to Mid-frequency Active Sonar during Submarine Commanders Courses at the Pacific Missile Range Facility: August 2013 through February 2015. Draft Report, Honolulu, HI.

Baird, R. W., and coauthors. 2008. Movements of satellite-tagged false killer whales around the main Hawaiian Islands. Pacific Scientific Review Group, Kihei, Hawaii.

Baird, R. W., and coauthors. 2010. Movements and habitat use of Cuvier's and Blainville's beaked whales in Hawaii: Results from satellite tagging in 2009/2010. National Marine Fisheries Service.

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. 2017b. 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. Odontocete Studies on the Pacific Missile Range Facility in February 2015: Satellite-Tagging, Photo-Identification, and Passive Acoustic Monitoring. Final Report (Prepared for Naval Facilities Engineering Command Pacific under HDR Environmentl, Operations and Construction, Inc. Contract No. N62470-10-D-3011, CTO KB28). Honolulu, HI: 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. 1987. Alternative population estimates of humpback whales (*Megaptera novaeangliae*) in Hawaiian waters. Canadian Journal of Zoology 65(11):2818-2821.

Baker, C. S., L. M. Herman, B. G. Bays, and G. B. Bauer. 1983a. The impact of vessel traffic on the behavior of humpback whales. Pages 5 *in* Fifth Biennial Conference on the Biology of Marine Mammals, New England Aquarium, Boston, Massachusetts.

Baker, C. S., L. M. Herman, B. G. Bays, and G. B. Bauer. 1983b. The impact of vessel traffic on the behavior of humpback whales in southeast Alaska: 1982 season. National Marine Fisheries Service, 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.

Balazs, G. H. 1991. Research Plan for Marine Turtle Fibropapilloma: Results of a December 1990 Workshop. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service [Southwest Fisheries Science Center].

Balazs, G. H., K. S. Van Houtan, S. A. Hargrove, S. M. Brunson, and S. K. Murakawa. 2015. A review of the demographic features of Hawaiian green turtles (Chelonia mydas). Chelonian Conservation and Biology 14(2):119-129.

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.

Barbieri, M. M., and coauthors. 2016. Protozoal-related mortalities in endangered Hawaiian monk seals Neomonachus schauinslandi. Diseases of aquatic organisms 121(2):85-95.

Barlow, D. R., K. S. Bernard, P. Escobar-Flores, D. M. Palacios, and L. G. Torres. 2020. Links in the trophic chain: Modeling functional relationships between in situ oceanography, krill, and blue whale distribution under different oceanographic regimes. Marine Ecology Progress Series 642:207–225.

Barlow, J. 1994. Recent information on the status of large whales in California waters, volume 203. US Department of Commerce, National Oceanic and Atmospheric Administration ….

Barlow, J. 1995. The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. Oceanographic Literature Review 9(42):784.

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. 2016. Cetacean Abundance in the California Current Estimated from Ship-based Line-transect Surveys in 1991–2014. (NOAA Administrative Report NMFS-SWFSC-LJ-1601). La Jolla, CA: Southwest Fisheries Science Center.

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.

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.

Barraza, A. D., and coauthors. 2020. Persistent organic pollutants in green sea turtles (Chelonia mydas) inhabiting two urbanized Southern California habitats. Marine Pollution Bulletin 153:110979.

Bartol, S. M., and D. R. Ketten. 2006. Turtle and tuna hearing. Pages 98-105 *in* Y. Swimmer, and R. Brill, editors. Sea turtle and pelagic fish sensory biology: Developing techniques to reduce sea turtle bycatch in longline fisheries, NOAA Technical Memo.

Bartol, S. M., J. A. Musick, and M. Lenhardt. 1999. Auditory Evoked Potentials of the Loggerhead Sea Turtle (*Caretta caretta*). Copeia 3:836-840.

Bauer, G. B. 1986. The behavior of humpback whales in Hawaii and modifications of behavior induced by human interventions. University of Hawaii.

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. 2013. Examining Explosions in Southern California and Their Potential Impact on Cetacean Acoustic Behavior. Southwest Fisheries Science Center, La Jolla, California.

Beale, C. M., and P. Monaghan. 2004. Behavioural responses to human disturbance: A matter of choice? Animal Behaviour 68(5):1065-1069.

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.

Becker, E. A., and coauthors. 2020. Habitat-Based Density Estimates for Cetaceans in the California Current Ecosystem Based on 1991-2018 Survey Data. U.S. Department of Commerce.

Becker, E. A., and coauthors. 2021. Habitat-based density estimates for cetaceans within the waters of the U.S. Exclusive Economic Zone around the Hawaiian Archipelago.

Begall, S., E. P. Malkemper, J. Červený, P. Němec, and H. Burda. 2013. Magnetic alignment in mammals and other animals. Mammalian Biology 78:10-20.

Bejder, L., A. Samuels, H. Whitehead, and N. Gales. 2006. Interpreting short-term behavioural responses to disturbance within a longitudinal perspective. Animal Behaviour 72(5):1149-1158.

Belcher, R. L., and T. E. Lee Jr. 2002. Arctocephalus townsendi. Mammalian Species 2002(700):1-5.

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. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere 2(7):1-27.

Benson, S. R., and coauthors. 2020. A long-term decline in the abundance of endangered leatherback turtles, Dermochelys coriacea, at a foraging ground in the California Current Ecosystem. Global Ecology and Conservation 24:e01371.

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.

Bergmann, M., L. Gutow, and M. Klages. 2015. Marine anthropogenic litter. Springer.

Bermúdez, R., and coauthors. 2016. Effect of ocean acidification on the structure and fatty acid composition of a natural plankton community in the Baltic Sea. Biogeosciences 13(24):6625.

Bernasconi, M., R. Patel, and L. Nøttestad. 2012. Behavioral observations of baleen whales in proximity of a modern fishing vessel. Pages 335-338 *in* The Effects of Noise on Aquatic Life. Springer.

Berzin, A. A. 1971. The sperm whale. (*Physeter macrocephalus*). Pishchevaya Promyshlennost Moscow, NTIS No. TT-71-50152.

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. 2015. Status review of the humpback whale (Megaptera novaeangliae) under the Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Bigelow, K., J. Rice, and F. Carvalho. 2022. Future Stock Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean (Update on Project 101).

Bignami, S., S. Sponaugle, and R. K. Cowen. 2013. Response to ocean acidification in larvae of a large tropical marine fish, *Rachycentron canadum*. Global change biology 19(4):996–1006.

Birney, K., and coauthors. 2015. Report on the 2014 Vessel Speed Reduction Incentive Trial in the Santa Barbara Channel.

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.

Bjorndal, K. A., and A. B. Bolten. 2010. Hawksbill sea turtles in seagrass pastures: success in a peripheral habitat. Marine Biology 157:135-145.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. 2005. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15(1):304-314.

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 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 29:E342–E365.

Blanchard, A. L., and coauthors. 2019. Prey biomass dynamics in gray whale feeding areas adjacent to northeastern Sakhalin (the Sea of Okhotsk), Russia, 2001–2015. Marine Environmental Research 145:123-136.

Bland, A. 2017. Why California Fishermen Are Throwing Deafening "Seal Bombs" at Sea Lions and Why No One is Stopping Them.

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.

Blevins, C., and coauthors. 2022. Sex-and age-specific migratory strategies of blue whales in the northeast Pacific Ocean. Frontiers in Marine Science 9:944918.

Blondin, H., B. Abrahms, L. B. Crowder, and E. L. Hazen. 2020. Combining high temporal resolution whale distribution and vessel tracking data improves estimates of ship strike risk. Biological Conservation 250:108757.

Blondin, H. E., and coauthors. 2022. Land-dependent marine species face climate-driven impacts on land and at sea. Marine Ecology Progress Series 699:181-198.

Blumenthal, J. M., and coauthors. 2009. Ecology of hawksbill turtles, *Eretmochelys imbricata*, on a western Caribbean foraging ground. Chelonian Conservation and Biology 8(1):1-10.

Boerger, C. M., G. L. Lattin, S. L. Moore, and C. J. Moore. 2010. Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. Marine Pollution Bulletin 60(12):2275–2278.

Bonaviri, C., M. Graham, P. Gianguzza, and N. T. Shears. 2017. Warmer temperatures reduce the influence of an important keystone predator. Journal of Animal Ecology 86(3):490-500.

Bonfil, R., S. Clarke, and H. Nakano. 2008. The biology and ecology of the oceanic whitetip shark, *Carcharhinus longimanus*. Pages 128-139 *in* M. D. Camhi, E. K. Pikitch, and E. A. Babcock, editors. Sharks of the open ocean: Biology, Fisheries and Conservation. Blackwell Publishing Ltd, Oxford, United Kingdom.

Boughton, D. A. 2022. South-Central/Southern California Coast Domain, In: Southwest Fisheries Science Center. 2022. Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest. Report to National Marine Fisheries Service - West Coast Region from Southwest Fisheries Science Center, Fisheries Ecology Division, Santa Cruz, CA.

Bousman, W. G., and R. M. Kufeld. 2005. UH-60A Airloads Catalog. National Aeronautics and Space Administration, Moffett Field, CA.

Bowman, V., and coauthors. 2024. Injuries to Pacific mackerel (Scomber japonicus) from underwater explosions. ICES Journal of Marine Science 81(8):1685-1695.

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. 2012. Line-transect abundance estimates of false killer whales (Pseudorca crassidens) in the Pelagic Region of the Hawaiian Exclusive Economic Zone and in the insular waters of the northwestern Hawaiian Islands.

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. 2017. Abundance estimates of cetaceans from a line-transect survey within the U.S. Hawaiian Islands Exclusive Economic Zone. Fishery Bulletin 115(2):129-142.

Bradford, A. L., E. M. Oleson, K. A. Forney, J. E. Moore, and J. Barlow. 2021. Line-transect abundance estimates of cetaceans in U.S. waters around the Hawaiian Islands in 2002, 2010, and 2017. NOAA, NMFS.

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, S. 1987. Age, growth and reproductive biology of the silky shark, Carcharhinus falciformis, and the scalloped hammerhead, Sphyrna lewini, from the northwestern Gulf of Mexico. Environmental Biology of Fishes 19(3):161-173.

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.

Briscoe, D. K., and coauthors. 2025. Multi-decade northward shift of loggerhead sea turtle pelagic habitat as the eastern North Pacific Transition Zone becomes more oligotrophic. Frontiers in Marine Science 11:1513162.

Briscoe, D. K., and coauthors. 2021. Dynamic thermal corridor may connect endangered loggerhead sea turtles across the Pacific Ocean. Frontiers in Marine Science 8:630590.

Britton-Simmons, K. H. 2004. Direct and indirect effects of the introduced alga Sargassum muticum on benthic, subtidal communities of Washington State, USA. Marine Ecology-Progress Series 277:61-78.

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.

Brumm, H., and H. Slabbekoorn. 2005. Acoustic communication in noise. Advances in the Study of Behavior 35:151-209.

Brüniche-Olsen, A., and coauthors. 2018. The inference of gray whale (Eschrichtius robustus) historical population attributes from whole-genome sequences. BMC Evolutionary Biology 18(1):1-12.

Brunson, S., and coauthors. 2022. Three decades of stranding data reveal insights into endangered hawksbill sea turtles in Hawai ‘i. Endangered Species Research 47:109-118.

Buck, J. R., and P. L. Tyack. 2000. Response of gray whales to low-frequency sounds. Journal of the Acoustical Society of America 107(5):2774.

Budelmann, B. U. 1992a. Hearing by Crustacea. Pages 131–139 *in* R. R. F. A. N. P. E. D. B. Webster, editor. Evolutionary Biology of Hearing. Springer-Verlag, New York, NY.

Budelmann, B. U. 1992b. Hearing in nonarthropod invertebrates. Pages 141-155 *in* The evolutionary biology of hearing. Springer.

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.

Burdin, A. M., O. A. Sychenko, and M. M. Sidorenko. 2013. Status of western gray whales off northeastern Sakhalin Island, Russia in 2012. IWC Scientific Committee, Jeju, Korea.

Burgess, K. B., and coauthors. 2016. Manta birostris, predator of the deep? Insight into the diet of the giant manta ray through stable isotope analysis. Royal Society Open Science 3(11):160717.

Burkhardt-Holm, P., and A. N'Guyen. 2019. Ingestion of microplastics by fish and other prey organisms of cetaceans, exemplified for two large baleen whale species. Marine Pollution Bulletin 144:224-234.

Burkhardt, K., and coauthors. 2025. Horizontal and vertical movement ecology of the oceanic whitetip shark (Carcharhinus Longimanus) in French Polynesia. Marine Biology 172(2):1-17.

Burrows, J. A., J. T. Harvey, K. M. Newton, D. A. Croll, and S. R. Benson. 2012. Marine mammal response to interannual variability in Monterey Bay, California. Marine Ecology Progress Series 461:257–271.

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.

Busse, L., and coauthors. 2006. Domoic acid in phytoplankton and fish in San Diego, CA, USA. Harmful Algae 5(1):91-101.

Butler, J., and coauthors. 2009. Status review report for black abalone (*Haliotis cracherodiiI* Leach, 1814). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Region.

Butler, J., M. Neuman, D. Pinkard, R. Kvitek, and G. Cochrane. 2006. The use of multibeam sonar mapping techniques to refine population estimates of the endangered white abalone (Haliotis sorenseni). Fishery Bulletin 104(4):521.

Cai, W., and coauthors. 2014. Increasing frequency of extreme El Niño events due to greenhouse warming. Nature Climate Change 4(2):111-116.

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. Final Report for contract AB-133F-10-RP-0106. Cascadia Research, PSRG-2013-13R, Olympia, Washington.

Calambokidis, J., and J. Barlow. 2020. Updated abundance estimates for blue and humpback whales along the U.S. West Coast using data through 2018. U.S. Department of Commerce, NOAA, NMFS.

Calambokidis, J., J. Barlow, K. Flynn, E. Dobson, and G. H. Steiger. 2017. Update on abundance, trends, and migrations of humpback whales along the US West Coast. IWC Report SC A 17:18-21.

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., E. Falcone, A. Douglas, L. Schlender, and J. Jessie Huggins. 2009b. Photographic identification of humpback and blue whales off the U.S. West Coast: Results and updated abundance estimates from 2008 field season. Final Report for Contract AB133F08SE2786. Cascadia Research, Olympia, Washington.

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. 2015. Biologically Important Areas for Selected Cetaceans Within U.S. Waters – West Coast Region. Aquatic Mammals (Special Issue) 41(1):39–53.

Caldwell, M. C., D. K. Caldwell, and P. L. Tyack. 1990. Review of the signature-whistle hypothesis for the Atlantic bottlenose dolphin. Pages 199-234 *in* S. L. R. R. Reeves, editor. The Bottlenose Dolphin. Academic Press, San Diego.

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.

Camargo, S. M., and coauthors. 2016. Structure and Genetic Variability of the Oceanic Whitetip Shark, *Carcharhinus longimanus*, Determined Using Mitochondrial DNA. PLoS ONE 11(5):e0155623.

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, J. G., and L. R. Goodman. 2004. Acute sensitivity of juvenile shortnose sturgeon to low dissolved oxygen concentrations. Transactions of the American Fisheries Society 133(3):772-776.

Campbell, L. M. 2003. Contemporary Culture, Use, and Conservation of Sea Turtles. P. L. Lutz, Musick, j. A., Wyneken, J., editor. The Biology of Sea Turtles volume Volume II. CRC Press, Boca Raton, FL.

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.

Cardeñosa, D., and coauthors. 2018. CITES-listed sharks remain among the top species in the contemporary fin trade. Conservation letters 11(4):e12457.

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.

Carillo, M., and F. Ritter. 2008. Increasing numbers of ship strikes in the Canary Islands: Proposals for immediate action to reduce risk of vessel-whale collisions. International Whaling Commission Scientific Committee, Santiago, Chile.

Carlson, J. K., and S. Gulak. 2012. Habitat use and movement patterns of oceanic whitetip, bigeye thresher and dusky sharks based on archival satellite tags. Collect. Vol. Sci. Pap. ICCAT 68(5):1922-1932.

Carlton, J. T., and L. G. Eldredge. 2015. Update and revisions of the marine bioinvasions of Hawai'i: The introduced and cryptogenic marine and estuarine animals and plants of the Hawaiian archipelago. Bish. Mus. Bull. Zool 9:25-47.

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. V. 2021a. Estimates of marine mammal, sea turtle, and seabird bycatch in the California large-mesh drift gillnet fishery: 1990-2019. U.S. Department of Commerce.

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., and coauthors. 2017. U. S. Pacific marine mammal stock assessments, 2016. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., and coauthors. 2018. 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. 2021. Sources of Human-related Injury and Mortality for US Pacific West Coast Marine Mammal Stock Assessments, 2015-2019. U.S. Department of Commerce, NOAA Technical Memorandum.

Carretta, J. V., and coauthors. 2014. U. S. Pacific marine mammal stock assessments, 2013. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., and coauthors. 2024. U.S. Pacific Marine Mammal Stock Assessments: 2023.

Carretta, J. V., and coauthors. 2025. Draft U.S. Pacific Marine Mammal Stock Assessments: 2024. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center

Carretta, J. V., and coauthors. 2023a. U.S. Pacific Marine Mammal Stock Assessments: 2023. NOAA-TM-NMFS-SWFSC-704. U.S. Department of Commerce, National Oceanographic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., and coauthors. 2023b. U.S. Pacific marine mammal stock assessments: 2022.

Carretta, J. V., Oleson, E.M., Forney, K.A., Muto, M.M., Weller, D.W., Lang, A.R., Baker, J., Hanson, B., Orr, A.J., Barlow, J., Moore, J.E. 2022. U.S. Pacific Marine Mammal Stock Assessments: 2021. U.S. Department of Commerce, NOAA, NMFS.

Carretta, J. V., Oleson, E.M., Forney, K.A., Muto, M.M., Weller, D.W., Lang, A.R., Baker, J., Hanson, B., Orr, A.J., Barlow, J., Moore, J.E., Brownell, R. L. Jr. 2021b. U.S. Pacific Marine Mammal Stock Assessments: 2020. U.S. Department of Commerce, NOAA, NMFS.

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.

Carroll, G. M., J. C. George, L. M. Philo, and C. W. Clark. 1989. Ice entrapped gray whales near Point Barrow, Alaska: Behavior, respiration patterns, and sounds. Pages 10 *in* Eighth Biennial Conference on the Biology of Marine Mammals, Asilomar Conference Center, Pacific Grove, California.

Cartwright, R., and coauthors. 2025. Tending the sick: Observations of epimeletic behavior in humpback whales towards conspecifics during entanglement events. PLoS ONE 20(4):e0321284.

Cartwright, R., and coauthors. 2019. Fluctuating reproductive rates in Hawaii's humpback whales, Megaptera novaeangliae, reflect recent climate anomalies in the North Pacific. Royal Society Open Science 6(3):181463.

Casal, A., and J. Orós. 2009. Plasma biochemistry and haematology values in juvenile loggerhead sea turtles undergoing rehabilitation. The Veterinary Record 164(21):663.

Casper, B., M. Halvorsen, F. Mattews, T. Carlson, and A. Popper. 2013. Recovery of barotrauma injuries resulting from exposure to pile driving sound in two sizes of hybrid striped bass. PLoS ONE, 8(9), e73844. .

Casper, B. M., T. J. Carlson, M. B. Halvorsen, and A. N. Popper. 2016. Effects of impulsive pile-driving exposure on fishes. Pages 125-132 *in* The effects of noise on aquatic life II. Springer.

Casper, B. M., M. B. Halvorsen, and A. N. Popper. 2012. Are sharks even bothered by a noisy environment? Adv Exp Med Biol 730:93-7.

Casper, B. M., P. S. Lobel, and H. Y. Yan. 2003. The hearing sensitivity of the little skate, *Raja erinacea*: A comparison of two methods. Environmental Biology of Fishes 68(4):371-379.

Casper, B. M., and D. A. Mann. 2006. Evoked potential audiograms of the nurse shark (*Ginglymostoma cirratum*) and the yellow stingray (*Urobatis jamaicensis*). Environmental Biology of Fishes 76:101-108.

Casper, B. M., and D. A. Mann. 2009. Field hearing measurements of the Atlantic sharpnose shark *Rhizoprionodon terraenovae*. J Fish Biol 75(10):2768-76.

Cassoff, R. M., and coauthors. 2011. Lethal entanglement in baleen whales. Diseases of aquatic organisms 96(3):175-185.

Castellote, M., C. W. Clark, and M. O. Lammers. 2012. Acoustic compensation to shipping and air gun noise by Mediterranean fin whales (Balaenoptera physalus). Pages 1 *in* A. N. P. A. Hawkings, editor. The Effects of Noise on Aquatic Life. Springer Science.

Caut, S., E. Guirlet, and M. Girondot. 2009. Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. Marine Environmental Research 69(4):254-261.

CDFW. 2000. Lower American River Pilot Salmon and Steelhead Spawning Habitat Improvement Project. Quarterly Status Report July 1999-March 2000.

Center for Ocean Solutions. 2009. Pacific Ocean Synthesis: Scientific Literature Review of Coastal and Ocean Threats, Impacts, and Solutions. Stanford University, San Jose, California.

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., T. M. Work, G. H. Balazs, S. K. Murakawa, and R. Morris. 2008. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). Marine Biology 154(5):887-898.

Chan, F., and coauthors. 2017. Persistent spatial structuring of coastal ocean acidification in the California Current System. Scientific Reports 7(1):2526.

Chapman, C. J., and A. D. Hawkins. 1973. Field study of hearing in cod, *Gadus morhua*-l. Journal of Comparative Physiology 85(2):147–167.

Chapman, R., and J. A. Seminoff. 2016. Status of Loggerhead Turtles (Caretta caretta) within Nations of the Inter-American Convention for the Protection and Conservation of Sea Turtles. Pages 46 in Inter-American Convention for the protection and conservation of sea turtles (IAC) , Technical Document CIT-CC13-2016-Tec.

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.

Cheung, W. W., 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.

Cheung, W. W., and coauthors. 2009. Projecting global marine biodiversity impacts under climate change scenarios. Fish and Fisheries 10(3):235-251.

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.

Chin, A., P. M. Kyne, T. I. Walker, and R. McAuley. 2010. An integrated risk assessment for climate change: analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. Global change biology 16(7):1936-1953.

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.

Christensen-Dalsgaard, J., and coauthors. 2012. Specialization for underwater hearing by the tympanic middle ear of the turtle, Trachemys scripta elegans. Proceedings of the Royal Society B: Biological Sciences 279(1739):2816-2824.

Christiansen, F., and coauthors. 2016. Spatial variation in directional swimming enables juvenile sea turtles to reach and remain in productive waters. Marine Ecology Progress Series, 557, 247–259. .

CITES. 2013. Consideration of Proposals for Amendment of Appendices I and II: Proposal Summary--Manta Rays *Manta* spp.

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.

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 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. JNCC Report No. 281.

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 coauthors. 2009. Acoustic masking in marine ecosystems: Intuitions, analysis, and implication. Marine Ecology Progress Series 395:201–222.

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.

Clark, T. B. 2010. Abundance, home range, and movement patterns of manta rays (Manta alfredi, M. birostris) in Hawai‘i. [Honolulu]:[University of Hawaii at Manoa],[December 2010].

Clarke, C. R., and coauthors. 2015. Global mitochondrial DNA phylogeography and population structure of the silky shark, *Carcharhinus falciformis*. Marine Biology 162(5):945-955.

Clarke, D., C. Dickerson, and K. Reine. 2003. Characterization of underwater sounds produced by dredges. Third Specialty Conference on Dredging and Dredged Material Disposal, Orlando, Florida.

Clarke, M. R. 1976. Observations on sperm whale diving. Journal of the Marine Biological Association of the United Kingdom 56(3):809-810.

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.

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.

Coles, S., and L. G. Eldredge. 2002. Nonindigenous species introductions on coral reefs: a need for information. Pacific Science 56(2):191-209.

Compagno, J. V. L. 1984. FAO species catalogue Vol.4, part 2 sharks of the world: An annotated and illustrated catalogue of shark species known to date. Food and Agriculture Organization of the United Nations.

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):art43.

Cook, S., and T. Forrest. 2005. Sounds produced by nesting leatherback sea turtles (Dermochelys coriacea). Herpetological Review 36(4):387-389.

Cooke, J. G., and coauthors. 2019. Population Assessment Update for Sakhalin Gray Whales. Western Gray Whale Advisory Panel. International Whaling Commission Document Cambridge, UK.

Cooke, J. G., and coauthors. 2017. Population assessment update for Sakhalin gray whales, with reference to stock identity. Paper SC/67a/NH/11 presented to the International Whaling Commission.

Cooper, H. L., D. C. Potts, and A. Paytan. 2016. Metabolic responses of the North Pacific krill, Euphausia pacifica, to short-and long-term pCO2 exposure. Marine Biology 163(10):207.

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.

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 Acoustical Society of America 113(2):1155–1165.

Costa, D. P., and coauthors. 2016. Assessing the exposure of animals to acoustic disturbance: towards an understanding of the population consequences of disturbance. Proceedings of Meetings on Acoustics. AIP Publishing.

Costante, D. 2021. Endangered Species Act: Quantifying Threats Impacting Listing. William & Mary, East Petersburg, PA.

Council, N. R. 2004. Endangered and threatened fishes in the Klamath River Basin: causes of decline and strategies for recovery. National Academies Press.

Council, N. R. 2010. Ocean acidification: a national strategy to meet the challenges of a changing ocean. National Academies Press.

Couturier, L. I., and coauthors. 2013. Stable isotope and signature fatty acid analyses suggest reef manta rays feed on demersal zooplankton. PLoS ONE 8(10):e77152.

Cox, B. S., A. M. Dux, M. C. Quist, and C. S. Guy. 2012. Use of a seismic air gun to reduce survival of nonnative lake trout embryos: a tool for conservation? North American Journal of Fisheries Management 32(2):292-298.

Cox, K. W. 1960. Review of the Abalone of California. California Fish and Game 46(4):381-406.

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.

Craig, J. K., and coauthors. 2001. Ecological effects of hypoxia on fish, sea turtles, and marine mammals in the northwestern Gulf of Mexico. American Geophysical Union, Washington, D.C.

Crain, C. M., B. S. Halpern, M. W. Beck, and C. V. Kappel. 2009. Understanding and Managing Human Threats to the Coastal Marine Environment. Pages 39–62 *in* R. S. Ostfeld, and W. H. Schlesinger, editors. The Year in Ecology and Conservation Biology, 2009. Blackwell Publishing, Oxford, UK.

Crane, N. L., and K. Lashkari. 1996. Sound production of gray whales, Eschrichtius robustus, along their migration route: A new approach to signal analysis. Journal of the Acoustical Society of America 100(3):1878-1886.

Cranford, T. W., and P. Krysl. 2015. Fin whale sound reception mechanisms: Skull vibration enables low-frequency hearing. PLoS ONE 10(1):e116222.

Croll, D., and coauthors. 1999. 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.

Crum, N., T. Gowan, A. Krzystan, and J. Martin. 2019. Quantifying risk of whale–vessel collisions across space, time, and management policies. Ecosphere 10(4):e02713.

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.

Cummings, W. C., P. O. Thompson, and R. Cook. 1968. Underwater sounds of migrating gray whales, Eschrichtius glaucus (Cope). Journal of the Acoustical Society of America 44(5):1278-1281.

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.

Curé, C., and coauthors. 2025. Severity Scoring of Sperm Whale Behavioral Responses to an Operational Sonar Source Reveals Importance of Received Level and Source-Receiver Distance. Aquatic Mammals 51(5):8-27.

Curtis, K. A., and coauthors. 2025. Harnessing the power of photo-ID data for apportionment to migratory whale herds: U.S. West Coast humpback whale stock proportions by latitude for the period 2019-2024.

Curtis, K. A., and coauthors. 2022. Abundance of humpback whales (*Megaptera novaeangliae*) wintering in Central America and southern Mexico from a one-dimensional spatial capture-recapture model.

Czapanskiy, M. F., and coauthors. 2021. Modelling short-term energetic costs of sonar disturbance to cetaceans using high-resolution foraging data. Journal of Applied Ecology.

D'Agnese, E., and coauthors. 2020. Reemergence of Guadalupe fur seals in the US Pacific Northwest: The epidemiology of stranding events during 2005–2016. Marine Mammal Science 36(3):828-845.

D'Amico, A. 2013. Analysis of Monk Seal Behavior Relative to Navy Activities. San Diego, CA: SPAWAR Systems Center Pacific.

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.

Dagit, R., and coauthors. 2020. Occurrences of Steelhead Trout (*Oncorhynchus mykiss*) in southern California, 1994-2018. California Fish and Game 106(1):38-56.

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., H. D. Fisher, and J. D. Schempp. 1984. Sound production by the gray whale and ambient noise levels in Laguna San Ignacio, Baja California Sur, Mexico. Pages 511-542 *in* M. L. J. S. L. S. S. Leatherwood, editor. The Gray Whale, Eschrichtius robustus. Academic Press, New York.

Dahlheim, M. E., and D. K. Ljungblad. 1990. Preliminary hearing study on gray whales (Eschrichtius robustus) in the field. Pages 335-346 *in* J. A. T. R. A. Kastelein, editor. Sensory Abilities of Cetaceans: Laboratory and Field Evidence. Plenum Press, New York.

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.

Danil, K., and coauthors. 2021. Marine algal toxins and their vectors in southern California cetaceans. Harmful Algae 103:102000.

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.

Danner, G. R., J. Chacko, and F. Brautigam. 2009. Voluntary ingestion of soft plastic fishing lures affects brook trout growth in the laboratory. North American Journal of Fisheries Management 29(2):352–360.

Dantas, D. V., M. Barletta, and M. F. da Costa. 2012. The seasonal and spatial patterns of ingestion of polyfilament nylon fragments by estuarine drums (Sciaenidae). Environmental Science and Pollution Research 19(2):600–606.

Darling, J. D., and coauthors. 2022. Humpback whales (*Megaptera novaeangliae*) attend both Mexico and Hawaii breeding grounds in the same winter: mixing in the northeast Pacific. Biology Letters 18(2):20210547.

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.

Davenport, J., J. Wrench, J. McEvoy, and V. Carnacho-Ibar. 1990. Metal and PCB concentrations in the "Harlech" leatherback. Marine Turtle Newsletter 48:1-6.

Davis, G. E., P. L. Haaker, and D. V. Richards. 1996. Status and trends of white abalone at the California Channel Islands. Transactions of the American Fisheries Society 125(1):42-48.

Davison, P., and R. G. Asch. 2011. Plastic ingestion by mesopelagic fishes in the North Pacific Subtropical Gyre. Marine Ecological Progress Series 432:173–180.

Day, R. D., A. L. Segars, M. D. Arendt, A. M. Lee, and M. M. Peden-Adams. 2007. Relationship of blood mercury levels to health parameters in the loggerhead sea turtle (*Caretta caretta*). Environmental Health Perspectives 115(10):1421.

De Boer, M., J. Saulino, T. Lewis, and G. Notarbartolo-Di-Sciara. 2015. New records of whale shark (Rhincodon typus), giant manta ray (Manta birostris) and Chilean devil ray (Mobula tarapacana) for Suriname. Marine Biodiversity Records 8.

de Stephanis, R., J. Giménez, E. Carpinelli, C. Gutierrez-Exposito, and A. Cañadas. 2013. As main meal for sperm whales: Plastics debris. Marine Pollution Bulletin 69(1-2):206-214.

De Weerdt Joelle, P. A., and coauthors. 2023. Spatial structuring and migratory destination of humpback whales, *Megaptera novaeangliae*, from Nicaragua. Book of Abstracts VLIZ Marine Science Day 2023.

Deakos, M. H., J. D. Baker, and L. Bejder. 2011. Characteristics of a manta ray *Manta alfredi* population off Maui, Hawaii, and implications for management. Marine Ecology Progress Series 429:245–260.

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.

Dearden, E. K. 2025. Singing Activity of Humpback Whales (Breeding Stock G) at Two Locations with Varied Boat Activity along the Pacific Coast of Costa Rica. UVM Patrick Leahy Honors College Senior Theses.

Deem, S. L., and coauthors. 2006. Blood values in free-ranging nesting leatherback sea turtles (*Dermochelys coriacea*) on the coast of the Republic of Gabon. Journal of Zoo and Wildlife Medicine 37(4):464-471.

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.

Delaney, D. G., and coauthors. 2017. Patterns in artisanal coral reef fisheries revealed through local monitoring efforts. PeerJ 5:e4089.

DeMaster, D. P., C. W. Fowler, S. L. Perry, and M. F. Richlen. 2001. Predation and competition: the impact of fisheries on marine-mammal populations over the next one hundred years. Journal of Mammalogy 82(3):641-651.

Denkinger, J., and coauthors. 2013. Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? Ocean & coastal management 80:29-35.

DeRuiter, S. L., and K. L. Doukara. 2012. Loggerhead turtles dive in response to airgun sound exposure. Endangered Species Research 16:55-63.

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.

Desforges, J. P., and coauthors. 2016. Immunotoxic effects of environmental pollutants in marine mammals. Environment International 86:126–139.

Dewar, H., and coauthors. 2008. Movements and site fidelity of the giant manta ray, Manta birostris, in the Komodo Marine Park, Indonesia. Marine Biology 155(2):121-133.

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.

Díaz-Delgado, E., O. Crespo-Neto, and R. O. Martínez-Rincón. 2021. Environmental preferences of sharks bycaught by the tuna purse-seine fishery in the Eastern Pacific Ocean. Fisheries Research 243:106076.

Díaz, M. P., H. P. Kunc, and J. D. Houghton. 2024. Anthropogenic noise predicts sea turtle behavioural responses. Marine Pollution Bulletin 198:115907.

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,.

DiMatteo, A., and coauthors. 2024. Sea turtle density surface models along the United States Atlantic coast. Endangered Species Research 53:227-245.

DiNardo, J., K. L. Stierhoff, and B. X. Semmens. 2021. Modeling the past, present, and future distributions of endangered white abalone (Haliotis sorenseni) to inform recovery efforts in California. PLoS ONE 16(11):e0259716.

Division of Aquatic Resources. 2021. Rapid Response to three species of Invasive Corals in Kāneʻohe Bay, Oʻahu, Hawaiʻi. Hawai'i Division of Aquatic Resources, Aquatic Invasive Species Team, Honolulu, HI.

do Sul, J. A. I., and M. F. Costa. 2014. The present and future of microplastic pollution in the marine environment. Environmental Pollution 185:352-364.

Dobroski, N., C. Brown, R. Nedelcheva, C. Scianni, and J. Thompson. 2015. 2015 Biennial Report on the California Marine Invasive Species Program. California State Lands Commission Marine Facilities Division, Sacramento, CA.

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.

Doney, S. C., and coauthors. 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science 4(1):11–37.

Donohoe, C. J., D. E. Rundio, D. E. Pearse, and T. H. Williams. 2021. Straying and life history of adult steelhead in a small California coastal stream revealed by otolith natural tags and genetic stock identification. North American Journal of Fisheries Management 41(3):711-723.

Dow Piniak, W. E., S. A. Eckert, C. A. Harms, and E. M. Stringer. 2012. Underwater hearing sensitivity of the leatherback sea turtle (*Dermochelys coriacea*): Assessing the potential effect of anthropogenic noise. U.S. Dept of the Interior, Bureau of Ocean Energy Management, Headquarters, Herndon, VA.

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-Jan):33-46.

Drake, J., and coauthors. 2008. Summary of scientific conclusions of the review of the status of eulachon (Thaleichthys pacificus) in Washington, Oregon and California. U. S. Department of Commerce, Northwest Fisheries Science Center, Seattle, WA.

Dransfield, A., and coauthors. 2014. Where the whales are: Using habitat modeling to support changes in shipping regulations within National Marine Sanctuaries in Central California. Endangered Species Research 26(1):39–57.

DuBois, M. J., N. F. Putman, and S. E. Piacenza. 2021. A global assessment of the potential for ocean-driven transport in hatchling sea turtles. Water 13(6):757.

Dueñas, M.-A., and coauthors. 2018. The role played by invasive species in interactions with endangered and threatened species in the United States: a systematic review. Biodiversity and Conservation 2018(27):3171-3183.

Duffy, C., and D. Abbott. 2003. Sightings of mobulid rays from northern New Zealand, with confirmation of the occurrence of Manta birostris in New Zealand waters.

Dulvy, N. K., S. A. Pardo, C. A. Simpfendorfer, and J. K. Carlson. 2014. Diagnosing the dangerous demography of manta rays using life history theory. PeerJ 2:e400.

Duncan, K. M., and K. N. Holland. 2006. Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks Sphyrna lewini in a nursery habitat. Marine Ecology Progress Series 312:211-221.

Dunlop, R. A. 2016. The effect of vessel noise on humpback whale, Megaptera novaeangliae, communication behaviour. Animal Behaviour 111:13-21.

Dunlop, R. A., J. Braithwaite, L. O. Mortensen, and C. M. Harris. 2021. Assessing population-level effects of anthropogenic disturbance on a marine mammal population. Frontiers in Marine Science 8:624981.

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., 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.

Durbach, I. N., and coauthors. 2021. Changes in the Movement and Calling Behavior of Minke Whales (*Balaenoptera acutorostrata*) in Response to Navy Training. Frontiers in Marine Science 8.

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.

Eckert, G. L. 2007. Spatial patchiness in the sea cucumber Pachythyone rubra in the California Channel Islands. Journal of Experimental Marine Biology and Ecology 348(1-2):121-132.

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. Fish and Wildlife Service.

Eckert, S. A., K. L. Eckert, P. Ponganis, and G. L. Kooyman. 1989. Diving and foraging behavior of leatherback sea turtles (Dermochelys coriacea). Canadian Journal of Zoology 67:2834-2840.

Eckert, S. A., D. W. Nellis, K. L. Eckert, and G. L. Kooyman. 1986. Diving patterns of two leatherback sea turtles (Dermochelys coriacea) during internesting intervals at Sandy Point, St. Croix, USVI. Herpetologica 42(3):381-388.

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, P. L. 1988. Characteristics of finback *Balaenoptera physalus* vocalizations in the St. Lawrence estuary. Bioacoustics 1:131-149.

Edmonds, N. J., C. J. Firmin, D. Goldsmith, R. C. Faulkner, and D. T. Wood. 2016. A review of crustacean sensitivity to high amplitude underwater noise: Data needs for effective risk assessment in relation to UK commercial species. Marine Pollution Bulletin 108:5-11.

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.

Eguchi, T., and coauthors. 2020. Effects of a power plant closure on home ranges of green turtles in an urban foraging area. Endangered Species Research 41:265-277.

Eguchi, T., T. Gerrodette, R. L. Pitman, J. A. Seminoff, and P. H. Dutton. 2007. At-sea density and abundance estimates of the olive ridley turtle Lepidochelys olivacea in the eastern tropical Pacific. Endangered Species Research 3(2):191-203.

Eguchi, T., A. R. Lang, and D. W. Weller. 2023. Abundance of eastern North Pacific gray whales 2022/2023.

Eguchi, T., and coauthors. 2018. Loggerhead turtles (Caretta caretta) in the California Current: abundance, distribution, and anomalous warming of the North Pacific. Frontiers in Marine Science 5:452.

Eguchi, T., and J. Seminoff. 2011. Final report on the aerial survey of the Southern California Bight 2011. National Marine Fisheries Service, Southwest Fisheries Science Center.

Eisaguirre, J. H., and coauthors. 2020. Trophic redundancy and predator size class structure drive differences in kelp forest ecosystem dynamics. Ecology 101(5):e02993.

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.

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.

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.

EPA. 2010. Climate Change Indicators in the United States: Weather and Climate. Pages 14 *in*. Evironmental Protection Agency.

EPA. 2014. Technical Fact Sheet - Perchlorate. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response.

Epperly, S. P., J. Braun, and A. J. Chester. 1995. Aerial surveys for sea turtles in North Carolina inshore waters. Fishery Bulletin 93(2):254-261.

Epperly, S. P., and coauthors. 1996. Beach strandings as an indicator of at-sea mortality of sea turtles. Bulletin of Marine Science 59(2):289-297.

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., 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., 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.

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.

Estupiñán-Montaño, C., E. Tamburin, and A. Delgado-Huertas. 2021. New insights into the trophic ecology of the scalloped hammerhead shark, Sphyrna lewini, in the eastern tropical Pacific Ocean. Environmental Biology of Fishes 104(12):1611-1627.

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, P. G. H., and A. Bjørge. 2013. Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership: 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., J. L. Crespo-Picazo, B. Sterba-Boatwright, B. A. Stacy, and D. Garcia-Parraga. 2017. Defining risk variables causing gas embolism in loggerhead sea turtles (Caretta caretta) caught in trawls and gillnets. Scientific Reports 7.

Fair, P. A., and coauthors. 2010. Contaminant blubber burdens in Atlantic bottlenose dolphins (*Tursiops truncatus*) from two southeastern U.S. estuarine areas: concentrations and patterns of PCBs, pesticides, PBDEs, PFCs, and PAHs. The Science of the Total Environment 408(7):1577–97.

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 coauthors. 2022. Movements and residency of fin whales (Balaenoptera physalus) in the California Current System. Mammalian Biology:1-18.

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. Movements of Cuvier's beaked whales in a region of frequent naval activity: Insights from sighting, photoidentification, and satellite tag data. Pages 78-79 *in* Eighteenth Biennial Conference on the Biology of Marine Mammals, Quebec City, Canada.

FAO. 2011. Food and Agriculture Organization of the United Nations Fisheries and Aquaculture Department Statistics.

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.

Fedewa, E. J., T. M. Jackson, J. J. Richar, J. L. Gardner, and M. A. Litzow. 2020. Recent shifts in northern Bering Sea snow crab (*Chionoecetes opilio*) size structure and the potential role of climate-mediated range contraction. Deep Sea Research Part II.

Feist, B. E., J. F. Samhouri, K. A. Forney, and L. E. Saez. 2021. Footprints of fixed-gear fisheries in relation to rising whale entanglements on the US West Coast. Fisheries Management and Ecology 28(3):283-294.

Felix, A., M. E. Stevens, and R. L. Wallace. 1995. Unpalatability of a colonial rotifer, *Sinantherina socialis,* to small zooplanktivorous fishes. Invertebrate Biology 114(2):139–144.

Ferrara, C. R., and coauthors. 2014. First evidence of leatherback turtle (Dermochelys coriacea) embryos and hatchlings emitting sounds. Chelonian Conservation and Biology 13(1):110-114.

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.

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. J. 2015. 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. 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., and A. K. Jenkins. 2012. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis, San Diego, CA.

Finneran, J. J., and C. E. Schlundt. 2013. Effects of fatiguing tone frequency on temporary threshold shift in bottlenose dolphins (Tursiops truncatus). The 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. The Journal of the Acoustical Society of America 122(2):1249-1264.

Fisher, F. W. 1994. Past and present status of Central Valley chinook salmon. Conservation Biology:870-873.

Fisheries and Oceans Canada. 2019. Necropsy results: Southern Resident Killer Whale J34.

Fleischer, L. A. 1978. The distribution, abundance, and population characteristics of the Guadalupe fur seal, Arctocephalus townsendi (Merriam 1897). University of Washington, Seattle, Washington.

Fletcher, N., L.,, L. Anderson, K. Melanson, and P. Raimondi. 2021. Investigations into recruitment enhancement and recovery of black abalone (*Haliotis cracherodii*) on San Clemente Island.

Flores-Cascante, L., J. Gómez-Gutiérrez, M. d. C. Gómez del Prado-Rosas, and D. Gendron. 2019. Helminth load in feces of free-ranging blue and fin whales from the gulf of California. Acta Parasitologica 64(3):625-637.

Foley, A. M., B. A. Schroeder, A. E. Redlow, K. J. Fick-Child, and W. G. Teas. 2005. Fibropapillomatosis in stranded green turtles (Chelonia mydas) from the eastern United States (1980–98): trends and associations with environmental factors. Journal of Wildlife Diseases 41(1):29-41.

Foley, A. M., and coauthors. 2019. Characterizing watercraft-related mortality of sea turtles in Florida. The Journal of Wildlife Management 83(5):1057-1072.

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.

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. K., and R. R. Reeves. 2008. Fight or flight: antipredator strategies of baleen whales. Mammal Review 38(1):50-86.

Ford, M. J. 2022. Biological Viability Assessment Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest.

Formicki, K., A. Korzelecka-Orkisz, and A. Tański. 2019. Magnetoreception in fish. Journal of Fish Biology 95(1):73-91.

Forney, K. A., J. Barlow, and J. Carretta. 1995. The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. Oceanographic Literature Review 9(42):784-785.

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 D. R. Kobayashi. 2007. Updated estimates of mortality and injury of cetaceans in the Hawaii-based longline fishery, 1994-2005. NOAA Technical Memorandum NMFS-SWFSC-412. 33p.

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.

Fossi, M. C., and coauthors. 2012. Are baleen whales exposed to the threat of microplastics? A case study of the Mediterranean fin whale (Balaenoptera physalus). Marine Pollution Bulletin 64(11):2374-2379.

Francis, C. D. J. R. B. 2013. A framework for understanding noise impacts on wildlife: An urgent conservation priority. Frontiers in Ecology and the Environment 11(6):305-313.

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.

Frazer, L. N., and E. Mercado Iii. 2000. A sonar model for humpback whale song. IEEE Journal of Oceanic Engineering 25(1):160-182.

Freedman, R., and S. S. Roy. 2012. Spatial patterning of Manta birostris in United States east coast offshore habitat. Applied Geography 32(2):652-659.

Frisk, G. V. 2012. Noiseonomics: The relationship between ambient noise levels in the sea and global economic trends. Scientific Reports 2(437):1–4.

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.

Fuentes, M., M. Hamann, and C. J. Limpus. 2009a. Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. Journal of Experimental Marine Biology and Ecology 383(1):56-64.

Fuentes, M. M. P. B., C. J. Limpus, and M. Hamann. 2011. Vulnerability of sea turtle nesting grounds to climate change. Global change biology 17:140-153.

Fuentes, M. M. P. B., and coauthors. 2009b. Proxy indicators of sand temperature help project impacts of global warming on sea turtles in northern Australia. Endangered Species Research 9:33-40.

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. .

Fuxjager, M. J., K. R. Davidoff, L. A. Mangiamele, and K. J. Lohmann. 2014. The geomagnetic environment in which sea turtle eggs incubate affects subsequent magnetic navigation behaviour of hatchlings. Proceedings of the Royal Society B: Biological Sciences 281(1791):20141218.

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.

Gagnon, C. 2016. Western gray whale activity in the East China Sea from acoustic data: Memorandum for Dr. Brandon Southall.

Gagnon, K., S. J. Peacock, Y. Jin, and M. A. Lewis. 2015. Modelling the spread of the invasive alga Codium fragile driven by long-distance dispersal of buoyant propagules. Ecological Modelling 316:111–121.

Gailey, G., and coauthors. 2020. Effects of sea ice on growth rates of an endangered population of gray whales. Scientific Reports 10(1):1-8.

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.

Gailey, G., and coauthors. 2022. Gray whale density during seismic surveys near their Sakhalin feeding ground. Environmental Monitoring and Assessment 194(Suppl 1):739.

Gallagher, C. A., V. Grimm, L. A. Kyhn, C. C. Kinze, and J. Nabe-Nielsen. 2021. Movement and seasonal energetics mediate vulnerability to disturbance in marine mammal populations. The American Naturalist 197(3):296-311.

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.

Gallo-Reynoso, J. P., A. L. Figueroa-Carranza, and B. J. Le Boeuf. 2008. Foraging behavior of lactating Guadalupe fur seal females. Pages 595-614 *in* Avances en el Estudio de los Mamíferos de México (Publicaciones Especiales ed.), volume 3. Asociación Mexicana de Mastozoologí, La Paz, Mexico.

Gálvez, C., M. A. Pardo, and F. R. Elorriaga-Verplancken. 2020. Impacts of extreme ocean warming on the early development of a marine top predator: The Guadalupe fur seal. Progress in Oceanography 180.

Gálvez, C., S. Raverty, F. Galván-Magaña, C. L. Field, and F. R. Elorriaga-Verplancken. 2023. Mortality in an off-shore predator critical habitat during the Northeast Pacific marine heatwave. Frontiers in Marine Science.

Gambell, R. 1985. 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. 1999. The International Whaling Commission and the contemporary whaling debate. Pages 179-198 *in* J. R. Twiss Jr., and R. R. Reeves, editors. Conservation and Management of Marine Mammals. Smithsonian Institution Press, Washington.

Gannier, A., and E. Praca. 2007. SST fronts and the summer sperm whale distribution in the north-west Mediterranean Sea. Journal of the Marine Biological Association of the United Kingdom 87(1):187.

Gaos, A. R., and coauthors. 2021. Hawksbill Nesting in Hawai'i: 30-Year Dataset Reveals Recent Positive Trend for a Small, Yet Vital Population. Frontiers in Marine Science 8.

Gaos, A. R., and coauthors. 2020. Hawaiian hawksbills: a distinct and isolated nesting colony in the Central North Pacific Ocean revealed by mitochondrial DNA. Conservation Genetics 21:771-783.

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-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., and coauthors. 2014. Decompression sickness ('the bends') in sea turtles. Diseases of aquatic organisms 111.3:191-205.

Gardner, S. C., M. D. Pier, R. Wesselman, and J. A. Juarez. 2003. Organochlorine contaminants in sea turtles from the Eastern Pacific. Marine Pollution Bulletin 46(9):1082-1089.

Geijer, C. K., and A. J. Read. 2013. Mitigation of marine mammal bycatch in US fisheries since 1994. Biological Conservation 159:54-60.

Gelsleichter, J., G. Sparkman, L. A. Howey, E. J. Brooks, and O. N. Shipley. 2020. Elevated accumulation of the toxic metal mercury in the Critically Endangered oceanic whitetip shark Carcharhinus longimanus from the northwestern Atlantic Ocean. Endangered Species Research 43:267-279.

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.

Gende, S. M., and coauthors. 2019. Active whale avoidance by large ships: components and constraints of a complementary approach to reducing ship strike risk. Frontiers in Marine Science 6:592.

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.

Gennip, S. J., and coauthors. 2017. Going with the flow: the role of ocean circulation in global marine ecosystems under a changing climate. Global change biology 23(7):2602-2617.

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 D. J. St. Aubin. 1990. Sea mammals and oil: confronting the risks. Academic Press, New York.

Gerber, J. A., J. Roletto, L. E. Morgan, D. M. Smith, and L. J. Gage. 1993. Findings in pinnipeds stranded along the central and northern California coast, 1984–1990. Journal of Wildlife Diseases 29(3):423-433.

Germanov, E. S., A. D. Marshall, L. Bejder, M. C. Fossi, and N. R. Loneragan. 2018. Microplastics: no small problem for filter-feeding megafauna. Trends in ecology & evolution 33(4):227-232.

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.

Gilman, E., and coauthors. 2010. Mitigating sea turtle by-catch in coastal passive net fisheries. Fish and Fisheries 11(1):57-88.

Gilman, E., and coauthors. 2007. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation 139(1):19-28.

Girondot, M., and coauthors. 2015. Spatio-temporal distribution of Manta birostris in French Guiana waters. Journal of the Marine Biological Association of the United Kingdom 95(1):153-160.

Gjertz, I., and A. Børset. 1992. Pupping in the most northerly harbor seal (*Phoca vitulina*). Marine Mammal Science 8(2):103–109.

Gobush, K., T. Wurth, J. Henderson, B. Becker, and C. Littnan. 2017. Prevalence of interactions between Hawaiian monk seals (Nemonachus schauinslandi) and nearshore fisheries in the main Hawaiian Islands. Pacific Conservation Biology 23(1):25-31.

Godard-Codding, C. A. J., and coauthors. 2011. Pacific Ocean–Wide Profile of CYP1A1 Expression, Stable Carbon and Nitrogen Isotope Ratios, and Organic Contaminant Burden in Sperm Whale Skin Biopsies. Environmental Health Perspectives 119(3):337–343.

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.

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.

Goldstein, T., and coauthors. 2008. Novel symptomatology and changing epidemiology of domoic acid toxicosis in California sea lions (Zalophus californianus): an increasing risk to marine mammal health. Proceedings of the Royal Society B: Biological Sciences 275(1632):267-276.

Good, C., and D. Johnston. 2010. Spatial modeling of optimal North Pacific right whale (Eubalaena japonica) calving habitats. Pages 71 *in* Alaska Marine Science Symposium, Hotel Captain Cook, Anchorage, Alaska.

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.

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.

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.

Graham, R. T., and coauthors. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLoS ONE 7(5):e36834.

Gransier, R., and R. A. Kastelein. 2024. Similar susceptibility to temporary hearing threshold shifts despite different audiograms in harbor porpoises and harbor seals. The Journal of the Acoustical Society of America 155(1):396-404.

Grant, S. C. H., and P. S. Ross. 2002. Southern Resident Killer Whales at Risk: Toxic Chemicals in the British Columbia and Washington Environment.

Gravem, S., and coauthors. 2021. *Pycnopodia helianthoides*. IUCN Red List of Threatened Species https://doi. org/10.2305/iucn. uk 1.

Green, M. E., and coauthors. 2022. Updated connectivity assessment for the scalloped hammerhead (Sphyrna lewini) in Pacific and Indian Oceans using a multi-marker genetic approach. Fisheries Research 251.

Greene, C. R., and W. J. Richardson. 1988. Characteristics of marine seismic survey sounds in the Beaufort Sea. The Journal of the Acoustical Society of America 83(6):2246-2254.

Greenman, J. 2024. California Vessel Collisions. Personal communicatio to Chip Johnson. H. D. Johnson, editor. National Marine Fisheries Service, Silver Spring, MD.

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.

Gribsholt, B., and E. Kristensen. 2002. Effects of bioturbation and plant roots on salt marsh biogeochemistry: a mesocosm study. Marine Ecology-Progress Series 241:71–87.

Grimes, C. B. 2007. ecology of juvenile salmon in the northeast Pacific Ocean. Symposium Ocean Ecology of Salmon in Western North America: Regional Comparisons (2005: Anchorage, Alaska). American Fisheries Society.

Grosholz, E. D., and G. M. Ruiz. 1996. Predicting the impact of introduced marine species: lessons from the multiple invasions of the European green crab Carcinus maenas. Biological Conservation 78(1-2):59-66.

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.

Grout, J. A., C. D. Levings, and J. S. Richardson. 1997. Decomposition rates of purple loosestrife (Lythrum salicaria) and Lyngbyei's sedge (Carexlyngbyei) in the Fraser River Estuary. Estuaries 20(1):96-102.

Gudger, E. W. 1922. The most northerly record of the capture in Atlantic waters of the United States of the giant ray, Manta birostris. Science 55(1422):338-340.

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.

Gulko, D., and K. L. Eckert. 2003. Sea Turtles: An Ecological Guide. Mutual Publishing, Honolulu, Hawaii.

Gulland, F. M., 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., K. E. Richerson, K. A. Somers, V. J. Tuttle, and J. E. Jannot. 2022. Information in Support of a Five-Year Status Review of Eulachon (*Thaleichthys pacificus*) Listed under the Endangered Species Act: Southern

Distinct Population Segment.

Gustafson, R., K. E. Richerson, K. A. Somers, V. J. Tuttle, and J. T. McVeigh. 2023. Observed and Estimated Bycatch of Eulachon in 2002–2021 U.S. West Coast Groundfish Fisheries. Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, WA.

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., 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.

Haigh, R., D. Ianson, C. A. Holt, H. E. Neate, and A. M. Edwards. 2015. Effects of ocean acidification on temperate coastal marine ecosystems and fisheries in the Northeast Pacific. PLoS ONE 10(2):e0117533.

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. 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.

Hakamada, T., and K. Matsuoka. 2016. The number of western North Pacific common minke, Bryde's and sei whales distributed in JARPNII Offshore survey area. International Whaling Commission document SC F 16.

Hakamada, T., K. Matsuoka, H. Murase, and T. Kitakado. 2017. Estimation of the abundance of the sei whale Balaenoptera borealis in the central and eastern North Pacific in summer using sighting data from 2010 to 2012. Fisheries science 83(6):887-895.

Hall, M. R., M. 2013. Bycatch and non-tuna catch in the tropical tuna purse seine fisheries of the world. . Pages 249 *in* FAO Fisheries and Aquaculture Technical Paper No. 568., Rome.

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. California Department of Fish and Game.

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, S. L., and coauthors. 2021. Disease-driven mass mortality event leads to widespread extirpation and variable recovery potential of a marine predator across the eastern Pacific. Proceedings of the Royal Society B 288(1957):20211195.

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.

Hanlon, R. T., and J. B. Messenger. 1996. Cephalopod Behaviour. Cambridge University Press, Cambridge, New York.

Hanna, M. E., and coauthors. 2021. Citizen-Sourced Sightings and Underwater Photography Reveal Novel Insights About Green Sea Turtle Distribution and Ecology in Southern California. Frontiers in Marine Science 8:671061.

Hanni, K. D., D. J. Long, R. E. Jones, P. Pyle, and L. E. Morgan. 1997a. Sightings and strandings of Guadalupe fur seals in Central and Northern California, 1988-1995. Journal of Mammalogy 78(2):684-690.

Hanni, K. D., D. J. Long, R. E. Jones, P. Pyle, and L. E. Morgan. 1997b. Sightings and strandings of Guadalupe fur seals in central and northern California, 1988–1995. Journal of Mammalogy 78(2):684-690.

Hanson, B. M., C. K. Emmons, E. J. Ward, J. A. Nystuen, and M. O. Lammers. 2013. Assessing the coastal occurrence of endangered killer whales using autonomous passive acoustic recorders. The Journal of the Acoustical Society of America 134(5):3486-3495.

Hanson, M. B., E. J. Ward, C. K. Emmons, M. M. Holt, and D. M. Holzer. 2017. Assessing the movements and occurrence of southern resident killer whales relative to the US Navy's Northwest Training Range Complex in the Pacific Northwest. Prepared for: US Navy, US Pacific Fleet, Pearl Harbor, HI. Prepared by: National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center under MIPR (00070).

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.

Harned, S. P., and coauthors. 2022. Genetic population dynamics of the critically endangered scalloped hammerhead shark (Sphyrna lewini) in the Eastern Tropical Pacific. Ecol Evol 12(12):e9642.

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 coauthors. 2017. Marine mammals and sonar: dose-response studies, the risk-disturbance hypothesis and the role of exposure context. Journal of Applied Ecology:1-9.

Harry, A., W. Macbeth, A. Gutteridge, and C. Simpfendorfer. 2011. The life histories of endangered hammerhead sharks (Carcharhiniformes, Sphyrnidae) from the east coast of Australia. Journal of Fish Biology 78(7):2026-2051.

Hart, C. E., C. Ley-QuiÑonez, A. Maldonado-Gasca, A. Zavala-Norzagaray, and F. A. A. Grobois. 2014. Nesting characteristics of olive ridley turtles (Lepidochelys olivacea) on El Naranjo Beach, Nayarit, Mexico. Herpetological Conservation and Biology 9(2):524-534.

Hart, K. M., P. Mooreside, and L. B. Crowder. 2006. Interpreting the spatio-temporal patterns of sea turtle strandings: going with the flow. Biological Conservation 129(2):283-290.

Harting, A. L., and coauthors. 2021. Population-level impacts of natural and anthropogenic causes-of-death for Hawaiian monk seals in the main Hawaiian Islands. Marine Mammal Science 37(1):235-250.

Hassler, T. J. 1987. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest). Coho Salmon. DTIC Document.

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.

Hastings, M. C., and A. N. Popper. 2005. Effects of sound on fish. California Department of Transportation, Sacramento, California.

Hatch, L., and A. J. Wright. 2007. A brief review of anthropogenic sound in the oceans. International Journal of Comparative Psychology 20:12.

Haver, S. M., and coauthors. 2021. Large vessel activity and low-frequency underwater sound benchmarks in United States waters. Marine Science 8.

Hawkes, L. A., A. C. Broderick, H. Godfrey, B. Godley, and M. J. Witt. 2014. The impacts of climate change on marine turtle reproductoin success. Pages 287-310 *in* B. Maslo, and L. Lockwood, editors. Coastal Conservation. Cambridge University Press, Cambridge.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2009. Climate change and marine turtles. Endangered Species Research 7(2):137-154.

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.

Hawkins, A. D., A. E. Pembroke, and A. N. Popper. 2015. Information gaps in understanding the effects of noise on fishes and invertebrates. Reviews in Fish Biology and Fisheries 25(1):39-64.

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. H., and coauthors, editors. 2023. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments 2022. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-304, Woods Hole, MA.

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.

Hays, G. C., A. D. Mazaris, and G. Schofield. 2014. Different male vs. female breeding periodicity helps mitigate offspring sex ratio skews in sea turtles. Frontiers in Marine Science 1:43.

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.

HDOA. 2018. Aquaculture in Hawaii. Hawaii Department of Agriculture, Animal Industry Division, http://hdoa.hawaii.gov/ai/aquaculture-and-livestock-support-services-branch/aquaculture-in-hawaii/.

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.

Heady, W., and coauthors. 2022. Roadmap to recovery for the sunflower sea star (*Pycnopodia helianthoides*) along the west coast of North America. The Nature Conservancy, Sacramento, CA, US.

Healey, M., C. Groot, and L. Margolis. 1991. Life history of chinook salmon (Oncorhynchus tshawytscha). Pacific salmon life histories:313-393.

Heinrichs, S., M. O'Malley, H. Medd, and P. Hilton. 2011. Manta ray of hope: global threat to manta and mobula rays. Manta Ray of Hope Project (www. mantarayofhope. com).

Helfman, G., B. B. Collette, D. E. Facey, and B. W. Bowen. 2009. The Diversity of Fishes: Biology, Evolution, and Ecology. Wiley.

Helm, J. E., S. M. Gende, and P. M. Lukacs. 2023. Quantifying behavior and collision risk of humpback whales surfacing near large ships: implications for detection and avoidance. Endangered Species Research 51:115-126.

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.

Hemery, L. G., S. R. Marion, C. G. Romsos, A. L. Kurapov, and S. K. Henkel. 2016. Ecological niche and species distribution modelling of sea stars along the Pacific Northwest continental shelf. Diversity and Distributions 22(12):1314-1327.

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.

Henderson, J. R. 2001. A pre-and post-MARPOL Annex V summary of Hawaiian monk seal entanglements and marine debris accumulation in the Northwestern Hawaiian Islands, 1982–1998. Marine Pollution Bulletin 42(7):584-589.

Hewitt, R. P. 1985. Reaction of dolphins to a survey vessel: Effects on census data. Fishery Bulletin 83(2):187-194.

Hildebrand, J. 2004. Impacts of anthropogenic sound on cetaceans. Unpublished paper submitted to the International Whaling Commission Scientific Committee SC/56 E 13.

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. 2009a. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series 395:5–20.

Hildebrand, J. A. 2009b. Metrics for characterizing the sources of ocean anthropogenic noise. Journal of the Acoustical Society of America 125(4):2517.

Hiruki, L. M., and T. J. Ragen. 1992. A compilation of historical monk seal, Monachus schauinslandi, counts. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Hitipeuw, C., and J. Maturbongs. 2002. Marine turtle conservation program, Jamursba-Medi nesting beach, north coast of the Bird's Head Peninsula, Papua. Pages 161-175 *in* Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. Western Pacific Regional Fishery Management Council Honolulu, Hawaii.

Hobday, A. J., and coauthors. 2016. A hierarchical approach to defining marine heatwaves. Progress in Oceanography 141:227-238.

Hobday, A. J., and M. J. Tegner. 2000. Status review of white abalone (Haliotis sorenseni) throughout its range in California and Mexico.

Hobday, A. J., M. J. Tegner, and P. L. Haaker. 2000. Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. Reviews in Fish Biology and Fisheries 10(4):493-514.

Hoegh-Guldberg, O., and J. F. Bruno. 2010. The impact of climate change on the world's marine ecosystems. Science 328(5985):1523-1528.

Høiberg, M. A., and coauthors. 2025. Marine debris impacts on Hawaiian green sea turtles (Chelonia mydas): High prevalence of hook-and-line fishing gear in strandings. Marine Pollution Bulletin 215:117869.

Holden, J., A. Hansen, A. Furman, R. Kharshan, and E. Austin. 2010. Vapor Corrosion Inhibitors in Hydro-Testing and Long Term Storage Applications. Presented at the NACE International Corrosion 2010 Conference & Expo. , San Antonio, TX.

Holst, M., and coauthors. 2011. Responses of Pinnipeds to Navy Missile Launches at San Nicolas Island, California. Aquatic Animals 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. U.S. Department of Commerce, NMFS-NWFSC-89.

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.

Honebrink, R., and P. Hendricks. 2016. Fishing in Hawai'i: a student manual. Division of Aquatic Resources, Department of Land and Natural Resources, Honolulu, HI.

Honnold, S. P., R. Braun, D. P. Scott, C. Sreekumar, and J. Dubey. 2005. Toxoplasmosis in a Hawaiian monk seal (Monachus schauinslandi). Journal of Parasitology 91(3):695-697.

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.

Horrocks, J. A., and coauthors. 2001. Migration routes and destination characteristics of post-nesting hawksbill turtles satellite-tracked from Barbados, West Indies. Chelonian Conservation and Biology 4(1):107-114.

Hoss, D. E., and L. R. Settle. 1990. Ingestion of plastics by teleost fishes. National Oceanic and Atmospheric Administration, NFMS-SWFSC-154, Honolulu, HI.

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. S., and coauthors. 2024. Direct hearing measurements in a baleen whale suggest ultrasonic sensitivity. Science 386(6724):902-906.

Houser, D. S., S. W. Martin, and J. J. Finneran. 2013. Behavioral responses of California sea lions to mid-frequency (3250-3450 Hz) sonar signals. Marine Environmental Researcgh 92:268–278.

Houston, A. I., P. A. Stephens, I. L. Boyd, K. C. Harding, and J. M. McNamara. 2007. Capital or income breeding? A theoretical model of female reproductive strategies. Behavioral Ecology 18(1):241-250.

Howey-Jordan, L. A., and coauthors. 2013. Complex Movements, Philopatry and Expanded Depth Range of a Severely Threatened Pelagic Shark, the Oceanic Whitetip (*Carcharhinus longimanus*) in the Western North Atlantic. PLoS ONE 8(2):e56588.

Howey, L. A., and coauthors. 2016. Into the deep: the functionality of mesopelagic excursions by an oceanic apex predator. Ecology and Evolution 6(15):5290-5304.

Hoyt, E. 2001. Whale Watching 2001: Worldwide Tourism, Numbers, Expenditures, and Expanding Socioeconomic Benefits, Yarmouth Port,MA.

Hughes, K. A., and coauthors. 2020. Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. Global change biology 2020(26):2702-2716.

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 19(4):1007–1024.

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.

Humber, F., B. J. Godley, and A. C. Broderick. 2014. So excellent a fishe: a global overview of legal marine turtle fisheries. Diversity and Distributions 20(5):579-590.

Illingworth and Rodkin Inc. 2007. Compendium of Pile Driving Sound Data. California Department of Transportation, Sacramento, CA.

Illingworth, R., and R. Rodkin. 2001. Noise and vibration measurements associated with the pile installation demonstration project for the San Francisco-Oakland Bay Bridge east span. Technical Report of Illingworth & Rodkin, Inc., Petaluma, CA.

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.

Ingman, K., and coauthors. 2021. Modeling changes in baleen whale seasonal abundance, timing of migration, and environmental variables to explain the sudden rise in entanglements in California. PLoS ONE 16(4):e0248557.

Intergovernmental Panel on Climate Change. 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change, New York, NY.

IOTC. 2014. Report of the Seventeenth Session of the IOTC Scientific Committee, IOTC–2014–SC17–R[E].

IPCC. 2014. Summary for policymakers. In: Climate Change 2014: Impacts,Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L.White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1-32.

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 P. J. O. Miller. 2015. Sperm whale response to tag boat presence: Biologically informed hidden state models quantify lost feeding opportunities. Ecosphere 6(1):Article 6.

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.

Issac, J. L. 2009. Effects of climate change on life history: Implications for extinction risk in mammals. Endangered Species Research 7(2):115-123.

IUCN. 2017. The IUCN Red List of Threatened Species. Version 2017-1. .

IWC. 2007. Whale population estimates. International Whaling Commission.

IWC. 2016. Report of the Scientific Committee. Journal of Cetacean Research and Management (Supplement) 17.

IWC. 2017. Strategic Plan to Mitigate the Impacts of Ship Strikes on Cetacean Populations: 2017-2020. IWC.

Jackson, G. D., N. G. Buxton, and M. J. A. George. 2000. Diet of the southern opah, *Lampris immaculatus,* on the Patagonian Shelf; the significance of the squid, *Moroteuthis ingens,* and anthropogenic plastic. Marine Ecology Progress Series 206:261–271.

Jackson, J. B., and coauthors. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293(5530):629-637.

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.

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.

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.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. 2015. Marine Mammals of the World: A Comprehensive Guide to Their Identification (2nd ed.). Academic Press, Cambridge, Massachusetts.

Jenkins, A. K., and coauthors. 2022. Physical effects of sound exposure from underwater explosions on Pacific mackerel (Scomber japonicus): Effects on non-auditory tissues. The Journal of the Acoustical Society of America 151(6):3947-3956.

Jenkins, A. K., and coauthors. 2023. Physical Effects from Underwater Explosions on Two Fish Species. Pages 1-9 *in* The Effects of Noise on Aquatic Life: Principles and Practical Considerations. Springer.

Jenkins, W. E., T. I. J. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations. Pages 476-484 *in* Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Jensen, A. S., and G. K. Silber. 2004. Large Whale Ship Strike Database. U.S. Department of Commerce, NMFS-OPR-25.

Jensen, M. P., and coauthors. 2018. Environmental warming and feminization of one of the largest sea turtle populations in the world. Current Biology 28(1):154-159. e4.

Jino, N., and coauthors. 2018. Community-based conservation of leatherback turtles in Solomon Islands: Local responses to global pressures. Conservation and Society 16(4):459-466.

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, C. 2025. Email - Subject: HCTT vessel traffic report. R. Salz, editor, U.S. Navy, Pacific Fleet, Environmental Readiness Division N465CJ.

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, M. L., and S. L. Swartz. 2002. Gray whale, Eschrichtius robustus. Pages 524-536 *in* W. F. P. B. W. J. G. M. Thewissen, editor. Encyclopedia of Marine Mammals. Academic Press, San Diego, California.

Jones, T. T., and coauthors. 2012. Resource requirements of the Pacific leatherback turtle population. PLoS ONE 7(10)): e45447.

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.

Jorgensen, S., A. Klimley, and A. Muhlia-Melo. 2009. Scalloped hammerhead shark Sphyrna lewini, utilizes deep-water, hypoxic zone in the Gulf of California. Journal of Fish Biology 74(7):1682-1687.

Joung, S.-J., N.-F. Chen, H.-H. Hsu, and K.-M. Liu. 2016. Estimates of life history parameters of the oceanic whitetip shark, *Carcharhinus longimanus*, in the Western North Pacific Ocean. Marine Biology Research 12(7):758-768.

Juárez-Ruiz, A., and coauthors. 2022. Guadalupe fur seal pup production predicted from annual variations of sea surface temperature in the southern California Current Ecosystem. ICES Journal of Marine Science 79(5):1637-1648.

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.

Juutilainen, J. 2005. Developmental effects of electromagnetic fields. Bioelectromagnetics 26(S7):S107-S115.

Kahane-Rapport, S., and coauthors. 2022. Field measurements reveal exposure risk to microplastic ingestion by filter-feeding megafauna. Nature Communications 13(1):1-11.

Kahng, S. E. 2006. Ecology and Ecological Impact of an Alien Octocoral, *Carijoa riisei*, in Hawai'i. University of Hawaii at Manoa, Honolulu, HI.

Kahng, S. E., and R. W. Grigg. 2005. Impact of an alien octocoral, *Carijoa riisei*, on black corals in Hawaii. Coral Reefs 24:556–562.

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.

Kannan, K., A. L. Blankenship, P. D. Jones, and J. P. Giesy. 2000. Toxicity Reference Values for the Toxic Effects of Polychlorinated Biphenyls to Aquatic Mammals. Human and Ecological Risk Assessment: An International Journal 6(1):181-201.

Kaplanis, N. J., J. Harris, and J. Smith. 2016. Distribution patterns of the non-native seaweeds *Sargassum horneri* (Turner) C. Agardh and *Undaria pinnatifida* (Harvey) Suringar on the San Diego and Pacific coast of North America. Aquatic Invasions 11(2):111–124.

Kapur, M., and A. Yau. 2018. Size Compositions and Sex Ratios of Oceanic Whitetip Sharks and Giant Manta Rays for Longline Fisheries in the Pacific Islands Region. NOAA, National Marine Fisheries Service, Pacific Islands Fisheries Science Center, Honolulu, HI.

Karl, T. R. 2009. Global climate change impacts in the United States. Cambridge University Press.

Kashiwagi, T., T. Ito, and F. Sato. 2010. Occurrences of reef manta ray, Manta alfredi, and giant manta ray, M. birostris. Japan, examined by photographic records. Report of Japanese Society for Elasmobranch Studies 46:20-27.

Kashiwagi, T., A. D. Marshall, M. B. Bennett, and J. R. Ovenden. 2011. Habitat segregation and mosaic sympatry of the two species of manta ray in the Indian and Pacific Oceans: Manta alfredi and M. birostris. Marine Biodiversity Records 4.

Kastak, D., J. Mulsow, A. Ghoul, and C. Reichmuth. 2008. Noise-induced permanent threshold shift in a harbor seal. Journal of the Acoustical Society of America 123(5 Part 2):2986.

Kastak, D. A., and R. J. Schusterman. 1998. Low-frequency amphibious hearing in pinnipeds: Methods, measurements, noise, and ecology. Journal of the Acoustical Society of America 103(4):2216-2228.

Kastelein, R. A., R. Gransier, L. Hoek, A. Macleod, and J. M. Terhune. 2012. Hearing threshold shifts and recovery in harbor seals (*Phoca vitulina*) after octave-band noise exposure at 4 kHz. The Journal of the Acoustical Society of America 132(4):2745–61.

Kastelein, R. A., and coauthors. 2020. Temporary hearing threshold shift in harbor seals (Phoca vitulina) due to one-sixth-octave noise bands centered at 0.5, 1, and 2 kHz. The Journal of the Acoustical Society of America 148(6):3873-3885.

Kastelein, R. A., and coauthors. 2021. Temporary Hearing Threshold Shift in California Sea Lions (Zalophus californianus) Due to One-Sixth-Octave Noise Bands Centered at 2 and 4 kHz: Effect of Duty Cycle and Testing the Equal-Energy Hypothesis. Aquatic Mammals 47(4).

Kastelein, R. A., L. Helder-Hoek, G. Janssens, R. Gransier, and T. Johansson. 2015. Behavioral responses of harbor seals (*Phoca vitulina*) to sonar signals in the 25-kHz range. Aquatic Mammals 41(4):388–399.

Kastelein, R. A., A. Smink, and N. Jennings. 2024. Atlantic Green Turtles and Hawksbill Turtles: Behavioral Responses to Sound. Pages 1243-1261 *in* The Effects of Noise on Aquatic Life: Principles and Practical Considerations. Springer.

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.

Keen, E. M., E. A. Falcone, R. D. Andrews, and G. S. Schorr. 2019. Diel Dive Behavior of Fin Whales (Balaenoptera physalus) in the Southern California Bight. Aquatic Mammals 45(2).

Keen, K. A., R. S. Beltran, E. Pirotta, and D. P. Costa. 2021. Emerging themes in population consequences of disturbance models. Proceedings of the Royal Society B 288(1957):20210325.

Keller, J. M., and coauthors. 2005. Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States Environmental Science and Technology 39(23):9101-9108.

Keller, J. M., J. R. Kucklick, M. A. Stamper, C. A. Harms, and P. D. McClellan-Green. 2004. Associations between organochlorine contaminant concentrations and clinical health parameters in loggerhead sea turtles from North Carolina, USA. Environmental Health Perspectives 112(10):1074.

Keller, J. M., P. D. McClellan-Green, J. R. Kucklick, D. E. Keil, and M. M. Peden-Adams. 2006. Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and *in vitro* exposure experiments. Environmental Health Perspectives 114(1):70-76.

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. .

Kelley, D. E., J. P. Vlasic, and S. W. Brillant. 2021. Assessing the lethality of ship strikes on whales using simple biophysical models. Marine Mammal Science 37(1):251-267.

Kelly, I. K. 2020. A Review of vessel collision threat to sea turtles in Hawai'i. NOAA, NMFS, Pacific Islands Regional Office, Protected Resources Division.

Kenner, M. C., and J. L. Yee. 2022. Black Abalone Surveys at Naval Base Ventura County, San Nicolas Island, California: 2021, Annual Report. U.S. Department of the Interior, U.S. Geological Survey Open-File Report 2022–1107, Reston, Virginia.

Kenner, M. C., and J. L. Yee. 2025. Black abalone surveys at Naval Base Ventura County, San Nicolas Island, California—2022 annual report.

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. 1992a. The cetacean ear: Form, frequency, and evolution. Pages 53–75 *in* R. A. Kastelein, A. Y. Supin, and J. A. Thomas, editors. Marine Mammal Sensory Systems. Plenum Press, New York.

Ketten, D. R. 1992b. The marine mammal ear: Specializations for aquatic audition and echolocation. The Evolutionary Biology of Hearing. D. B. Webster, R. R. Fay and A. N. Popper (eds.). Springer-Verlag, New York, NY. p.717-750.

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 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. U.S. Department of Commerce, NOAA-TM-NMFS-SWFSC-256.

Ketten, D. R. 2008. Underwater ears and the physiology of impacts: Comparative liability for hearing loss in sea turtles, birds, and mammals. Bioacoustics 17(1-3):312-315.

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.

Kiessling, I. 2003. Finding solutions: derelict fishing gear and other marine debris in northern Australia. National Oceans Office.

Kim, K. H., and R. G. Norman. 2023. Acoustic Measurements of PMRF Missile Launch: March 2023 Greeneridge Sciences, Inc., Santa Barbara, CA.

Kincaid, A. L., J. A. Lasala, J. L. Blackburn, B. A. Stacy, and G. N. Lovewell. (in press). Analysis of Anatomical Location and Severity of Watercraft Injuries in Different Species of Sea Turtles. Endangered Species Research.

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.

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. 1997. Homing in on vertebrates. Nature 390(6658):339-340.

Kitchen-Wheeler, A.-M. 2013. The behaviour and ecology of Alfred mantas (Manta alfredi) in the Maldives. Newcastle University.

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.

Klimley, A. 1993. Highly directional swimming by scalloped hammerhead sharks, Sphyrna lewini, and subsurface irradiance, temperature, bathymetry, and geomagnetic field. Marine Biology 117(1):1-22.

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.

Kloepper, L., P. Nachtigall, R. Gisiner, and M. Breese. 2010. Decreased echolocation performance following high-frequency hearing loss in the false killer whale (Pseudorca crassidens). Journal of Experimental Biology 213(21):3717-3722.

Kloepper, L. N., P. E. Nachtigall, M. J. Donahue, and M. Breese. 2012. Active echolocation beam focusing in the false killer whale, Pseudorca crassidens. Journal of Experimental Biology 215(8):1306-1312.

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.

Koch, V., H. Peckham, A. Mancini, and T. Eguchi. 2013. Estimating at-sea mortality of marine turtles from stranding frequencies and drifter experiments. PLoS ONE 8(2):e56776.

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. .

Konar, B., and coauthors. 2019. Wasting disease and static environmental variables drive sea star assemblages in the Northern Gulf of Alaska. Journal of Experimental Marine Biology and Ecology 520:151209.

Koopman, H. N. 2007. Phylogenetic, ecological, and ontogenetic factors influencing the biochemical structure of the blubber of odontocetes. Marine Biology 151(1):277-291.

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.

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.

Kratofil, M. A., G. M. Ylitalo, S. D. Mahaffy, K. L. West, and R. W. Baird. 2020. Life history and social structure as drivers of persistent organic pollutant levels and stable isotopes in Hawaiian false killer whales (Pseudorca crassidens). Science of The Total Environment 733:138880.

Kraus, S. D., and coauthors. 2005. North Atlantic right whales in crisis. Science 309(5734):561-562.

Kretzmann, M., N. J. Gemmell, and A. Meyer. 2001. Microsatellite analysis of population structure in the endangered Hawaiian monk seal. Conservation Biology 15:457-466.

Kretzmann, M., and coauthors. 1997. Low genetic variability in the Hawaiian monk seal. Conservation Biology 11:482-490.

Krumpel, A., and coauthors. 2021. Long-term patterns of noise from underwater explosions and their relation to fisheries in Southern California. Frontiers in Marine Science 8:796849.

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.

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, D. Scheie, L. P. Folkow, and A. S. Blix. 2010. Effects of naval sonar on seals. Norwegian Defense Research Establishment (FFI).

LaCasella, E. 2025. Sea Turtle Standings Data 2018 to March 2025 Excel Spreadsheet. R. Driskell, editor. NMFS Southwest Fisheries Science Center.

Ladich, F., and R. R. Fay. 2013. Auditory evoked potential audiometry in fish. 23(3):317-364.

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.

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.

Laloë, J. O., G. Schofield, and G. C. Hays. 2024. Climate warming and sea turtle sex ratios across the globe. Global change biology 30(1):e17004.

Lambert, P. 2000. Sea Stars of British Columbia, Southeast Alaska, and Puget Sound. UBC press.

Lambertsen, R. 1992. 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., 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.

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. 2022. Population structure of North Pacific gray whales in light of trans-Pacific movements. Marine Mammal Science 38(2):433-468.

Langan, R. 2004. Balancing marine aquaculture inputs and extraction: combined culture of finfish and bivalve molluscs in the open ocean. BULLETIN-FISHERIES RESEARCH AGENCY JAPAN:51-58.

Lange, C., F. Reid, and M. Vernet. 1994. Temporal distribution of the potentially toxic diatom Pseudonitzschia australis at a coastal site in Southern California. Marine Ecology Progress Series:309-312.

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.

Lari, E., A. Kasumyan, F. Falahat, K. Døving, and V. Jafari. 2013. Palatability of food animals for stellate sturgeon A cipenser stellatus P allas, 1771. Journal of Applied Ichthyology 29(6):1222-1224.

Laufkötter, C., J. Zscheischler, and T. L. Frölicher. 2020. High-impact marine heatwaves attributable to human-induced global warming. Science 369(6511):1621-1625.

Laughlin, J. 2006. Underwater sound levels associated with pile driving at the Cape Dissapointment boat launch facility, wave barrier project. Washington State Parks Wave Barrier Project Underwater Technical Report.

Lavender, A. L., S. M. Bartol, and I. K. Bartol. 2014. Ontogenetic investigation of underwater hearing capabilities in loggerhead sea turtles (*Caretta caretta*) using a dual testing approach. Journal of Experimental Biology 217(14):2580-2589.

Lawson, J. M., and coauthors. 2017. Sympathy for the devil: a conservation strategy for devil and manta rays. PeerJ 5:e3027.

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. 1982. 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.

Lebreton, L., and coauthors. 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. Scientific Reports 8(1):4666.

Leduc, A. O., P. L. Munday, G. E. Brown, and M. C. Ferrari. 2013. Effects of acidification on olfactory-mediated behaviour in freshwater and marine ecosystems: a synthesis. Philosophical Transactions of the Royal Society of London B: Biological Sciences 368(1627):20120447.

Lee, H., and coauthors. 2023. Synthesis report of the IPCC Sixth Assessment Report (AR6), Longer report. IPCC.

Lefebvre, K. A., and coauthors. 2016. Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. Harmful Algae 55:13-24.

Leighton, D. 2005. Status review for the black abalone, Haliotis cracherodii Leach 1814. Unpublished document produced for the Black Abalone Status Review Team, Office of Protected Resources, Southwest Region, National Marine Fisheries Service, Long Beach, CA, USA.

Lenhardt, M. L. 1982. Bone conduction hearing in turtles. Journal of Auditory Research.

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. C. Bjorndal, A. B. C. Bolten, D. A. C. Johnson, and P. J. C. Eliazar, editors. Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Lenhardt, M. L. 2002. Sea turtle auditory behavior. Journal of the Acoustical Society of America 112(5 Part 2):2314.

Leroux, R. A., and coauthors. 2012. Re-examination of population structure and phylogeography of hawksbill turtles in the wider Caribbean using longer mtDNA sequences. Journal of Heredity 103(6):806-820.

Lessa, R., F. M. Santana, and R. Paglerani. 1999. Age, growth and stock structure of the oceanic whitetip shark, *Carcharhinus longimanus*, from the southwestern equatorial Atlantic. Fisheries Research 42(1-2):21-30.

Lettrich, M. D., and coauthors. 2023. Vulnerability to climate change of United States marine mammal stocks in the western North Atlantic, Gulf of Mexico, and Caribbean. PLoS ONE 18(9):e0290643.

Levi, F., and P. Francour. 2004. Behavioural response of Mullus surmuletus to habitat modification by the invasive macroalga Caulerpa taxifolia. Journal of Fish Biology 64:55–64.

Lewison, R., and coauthors. 2013. Fisheries bycatch of marine turtles lessons learned from decades of research and conservation. Pages 329-351 *in* The biology of sea turtles, volume III.

Lewison, R. L., and coauthors. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proceedings of the National Academy of Sciences 111(14):5271-5276.

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(3):221-231.

Lewitus, A. J., and coauthors. 2012. Harmful algal blooms along the North American west coast region: History, trends, causes, and impacts. Harmful Algae 19:133-159.

Liboff, A. 2016. Is the geomagnetic map imprinted in pre-emergent egg? Electromagnetic biology and medicine 35(2):167-169.

Liboff, A. R. 2014. Why are living things sensitive to weak magnetic fields? Electromagnetic biology and medicine 33(3):241-245.

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. 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.

Litzow, M. A., F. J. Mueter, and A. J. Hobday. 2014. Reassessing regime shifts in the North Pacific: incremental climate change and commercial fishing are necessary for explaining decadal-scale biological variability. Global change biology 20(1):38-50.

Liu, K.-M., and C.-T. Chen. 1999. Demographic analysis of the scalloped hammerhead, Sphyrna lewini, in the northwestern Pacific. Fisheries science 65(2):218-223.

Llyod, B. D. 2003. Potential effects of mussel farming on New Zealand's marine mammals and seabirds: a discussion paper. Department of Conservation, Wellington, New Zealand.

Lockwood, B. L., and G. N. Somero. 2011. Invasive and native blue mussels (genus Mytilus) on the California coast: The role of physiology in a biological invasion☆. Journal of Experimental Marine Biology and Ecology.

Lohmann, K. J., S. D. Cain, S. Dodge, and C. M. F. Lohmann. 2000. Magnetic navigation in hatchling loggerheads. Pages 6 *in* H. J. Kalb, and T. Wibbels, editors. Nineteenth Annual Symposium on Sea Turtle Biology and Conservation.

Lohmann, K. J., and C. M. Lohmann. 1996. Detection of magnetic field intensity by sea turtles. Nature 380(6569):59.

Lohmann, K. J., and C. M. Lohmann. 1998. Sea turtle navigation and the detection of geomagnetic field features. The Journal of Navigation 51(1):10-22.

Lonhart, S. I., R. Jeppesen, R. Beas-Luna, J. A. Crooks, and J. Lorda. 2019. Shifts in the distribution and abundance of coastal marine species along the eastern Pacific Ocean during marine heatwaves from 2013 to 2018. Marine Biodiversity 2019.

Lontoh, D. N. 2014. Variation in tissue stable isotopes, body size, and reproduction of western Pacific leatherback turtles. San José State University.

Lopez, J., D. Boyd, G. M. Ylitalo, C. Littnan, and R. Pearce. 2012. Persistent organic pollutants in the endangered Hawaiian monk seal (Monachus schauinslandi) from the main Hawaiian Islands. Marine Pollution Bulletin 64(11):2588-2598.

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.

Lotze, H. K., M. Coll, A. M. Magera, C. Ward-Paige, and L. Airoldi. 2011. Recovery of marine animal populations and ecosystems. Trends in ecology & evolution 26(11):595-605.

Lowry, D., and coauthors. 2022. Endangered Species Act Status Review Report: Sunflower Sea Star (*Pycnopodia helianthoides*). Final Report to the National Marine Fisheries Service, Office of Protected Resources.

Luksenburg, J. A., and E. C. Parsons. 2009. The effects of aircraft on cetaceans: Implications for aerial whalewatching.

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.

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.

Lyman, E. 2014. 2013-2014 Hawaii Large Whale Entanglements and Response Efforts around the Main Hawaiian Islands. NOAA Hawaiian Islands Humpback Whale National Marine Sanctuary.

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.

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.

Mackie, G., and C. Singla. 2003. The capsular organ of Chelyosoma productum (Ascidiacea: Corellidae): a new tunicate hydrodynamic sense organ. Brain, behavior and evolution 61(1):45-58.

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., I. Kerr, and R. Payne. 2004. Echolocation clicks of two free-ranging, oceanic delphinids with different food preferences: false killer whales Pseudorca crassidens and Risso's dolphins Grampus griseus. Journal of Experimental Biology 207(11):1811-1823.

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.

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.

Maguire, J.-J. 2006. The state of world highly migratory, straddling and other high seas fishery resources and associated species. Food & Agriculture Org.

Mahaffy, S. D., and coauthors. 2023. Identifying social clusters of endangered main Hawaiian Islands false killer whales. Endangered Species Research 51:249-268.

Malme, C. I., P. R. Miles, C. W. Clark, P. Tyack, and J. E. Bird. 1983. Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray

whale behavior. Final report for the period of 7 June 1982 - 31 July 1983. Report No. 5366. For U.S. Department of the Interior, Minerals Management Service, Alaska OCS Office, Anchorage, AK 99510. 64pp.

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.

Malme, C. I., B. Wursig, J. E. Bird, and P. Tyack. 1986. Behavioral responses of gray whales to industrial noise: Feeding observations and predictive modeling. Final Report for the Outer Continental Shelf Environmental Assessment Program, Research Unit 675. 207pgs.

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.

Mancini, A., V. Koch, J. A. Seminoff, and B. Madon. 2012. Small-scale gill-net fisheries cause massive green turtle Chelonia mydas mortality in Baja California Sur, Mexico. Oryx 46(1):69-77.

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. .

Markowitz, T. M., A. D. Harlin, B. Würsig, and C. J. McFadden. 2004. Dusky dolphin foraging habitat: overlap with aquaculture in New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems 14(2):133-149.

Marks, L. M., and coauthors. 2015. Range expansion of a non-native, invasive macroalga *Sargassum horneri* (Turner) C. Agardh, 1820 in the eastern Pacific. BioInvasions Records 4(4):243–248.

Marshall, A., and M. Bennett. 2010. Reproductive ecology of the reef manta ray Manta alfredi in southern Mozambique. Journal of Fish Biology 77(1):169-190.

Marshall, A., and coauthors. 2016. Manta birostris. The IUCN Red List of Threatened Species 2011: e. T198921A9108067.

Marshall, A. D., L. J. Compagno, and M. B. Bennett. 2009. Redescription of the genus Manta with resurrection of Manta alfredi (Krefft, 1868)(Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa 2301(1):1-28.

Marshall, A. D., C. L. Dudgeon, and M. B. Bennett. 2011. Size and structure of a photographically identified population of manta rays Manta alfredi in southern Mozambique. Marine Biology 158(5):1111-1124.

Martien, K. 2025. Genetic data provide insights into social structure, population structure, and ancestry of false killer whales. Whalewatcher, 2025, False Killer Whales (The Pseudorca Issue):66-70.

Martien, K. K., and coauthors. 2021. Evaluation of Mexico Distinct Population Segment of Humpback Whales as units under the Marine Mammal Protection Act. NOAA, NMFS.

Martien, K. K., and coauthors. 2019. Fidelity to natal social groups and mating within and between social groups in an endangered false killer whale population. Endangered Species Research 40:219-230.

Martin, K. J., and coauthors. 2012. Underwater hearing in the loggerhead sea turtles (*Caretta caretta*): a comparison of behvioral and auditory evoked potential audiograms. Journal of Experimental Biology 215:3001-3009.

Martin, S., and coauthors. 2020a. Update to assessing the population-level impacts of North Pacific loggerhead and western Pacific leatherback turtle interactions: inclusion of the Hawaii-based deep-set and American Samoabased longline fisheries. U.S. Dept. of Commerce, NOAA Technical Memorandum.

Martin, S. L., and coauthors. 2020b. Assessing the population level impacts of North Pacific loggerhead and western Pacific leatherback interactions in the Hawaii-based shallow set longline fishery.

Martin, S. L., S. M. Stohs, and J. E. Moore. 2015a. Bayesian inference and assessment for rare-event bycatch in marine fisheries: a drift gillnet fishery case study. Ecological Applications 25(2):416-429.

Martin, S. W., C. R. Martin, B. M. Matsuyama, and E. E. Henderson. 2015b. Minke whales (Balaenoptera acutorostrata) respond to navy training. Journal of the Acoustical Society of America 137(5):2533-2541.

Martínez-Loustalot, P., and coauthors. 2022. Towards the definition of the humpback whale population units along the Mexican and Central American coasts in the Pacific Ocean. Marine Mammal Science.

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.

Massey, L. M., S. Penna, E. Zahn, D. Lawson, and C. M. Davis. 2023. Monitoring green sea turtles in the San Gabriel River of Southern California. Animals 13(3):434.

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.

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.

Matsuzawa, Y., and coauthors. 2016. Fine-scale genetic population structure of loggerhead turtles in the Northwest Pacific. Endangered Species Research 30:83-93.

Matthews, K. E., J. L. Mohay, J. W. Todd, and R. M. Starr. 2022. Bycatch in the California halibut (*Paralichthys californicus*) trawl fishery. Bulletin of the Southern California Academy of Sciences 121(8):88–109.

Maurer, A. S., and coauthors. 2024. Resource selection by a megaomnivore in a marine foraging habitat. Ecology and Evolution 14(11):e70132.

Mauzey, K. P., C. Birkeland, and P. K. Dayton. 1968. Feeding behavior of asteroids and escape responses of their prey in the Puget Sound region. Ecology 49(4):603-619.

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.

Maybaum, H. L. 1990. Effects of a 3.3 kHz sonar system on humpback whales, *Megaptera novaeangliae*, in Hawaiian waters. EOS 71:92.

Mayne, B., A. D. Tucker, O. Berry, and S. Jarman. 2020. Lifespan estimation in marine turtles using genomic promoter CpG density. PLoS ONE 15(7):e0236888.

Mazaris, A. D., A. S. Kallimanis, S. P. Sgardelis, and J. D. Pantis. 2008. Do long-term changes in sea surface temperature at the breeding areas affect the breeding dates and reproduction performance of Mediterranean loggerhead turtles? Implications for climate change. Journal of Experimental Marine Biology and Ecology.

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. 2017. Widely used marine seismic survey air gun operations negatively impact zooplankton. Nature Ecology and Evolution 1(7):195.

McCauley, R. D., and coauthors. 2000. 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.

McClellan, C. M., J. Braun-McNeill, L. Avens, B. P. Wallace, and A. J. Read. 2010. Stable isotopes confirm a foraging dichotomy in juvenile loggerhead sea turtles. Journal of Experimental Marine Biology and Ecology 387:44-51.

McCorkle, S. L. 2016. Fibropapillomatosis in Marine Turtles: A Comprehensive Review. Nova Southeastern University, Fort Lauderdale, FL.

McCracken, M. L. 2000. Estimation of Sea Turtle Take and Mortality in the Hawaiian Longline Fisheries. National Marine Fisheries Service, Southwest Fisheries Science Center.

McCracken, M. L., and K. A. Forney. 2010. Preliminary assessment of incidental interactions with marine mammals in the hawaii longline deep and shallow set fisheries. NOAA, NMFS.

McCue, L. M., Fahy, C.C., Greenman, J., Wilkinson, K. 2021. Status Review of the Guadalupe Fur Seal (*Arctocephalus townsendi*). National Marine Fisheries Service, Protected Resources Division, West Coast Region, 501 West Ocean Blvd., Long Beach, California, 90802.

McDonald, M., J. Hildebrand, S. Wiggins, and D. Ross. 2008. A 50 year comparison of ambient ocean noise near San Clemente Island: A bathymetrically complex coastal region off Southern California. The Journal of the Acoustical Society of America 124(4):1985–1992.

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. C. Webb. 1995. 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. M. Wiggins. 2006. Increases in deep ocean ambient noise in the Northeast Pacific west of San Nicolas Island, California. Journal of the Acoustical Society of America 120(2):711-718.

McDonald, M. A., and coauthors. 2005. Sei whale sounds recorded in the Antarctic. Journal of the Acoustical Society of America 118(6):3941–3945.

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. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

McEwan, D. R. 2001. Central Valley steelhead. California Department of Fish and Game, Sacramento, California.

McGowan, J. A., and coauthors. 2017. Predicting coastal algal blooms in southern California. Ecology 98(5):1419-1433.

McKenna, L. N., F. V. Paladino, P. S. Tomillo, and N. J. Robinson. 2019. Do sea turtles vocalize to synchronize hatching or nest emergence? Copeia 107(1):120-123.

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.

McLaskey, A. K., and coauthors. 2016. Development of *Euphausia pacifica* (krill) larvae is impaired under pCO_2 levels currently observed in the Northeast Pacific. Marine Ecological Progress Series 555:65–78.

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.

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.

Medeiros, A., O. Luiz, and C. Domit. 2015. Occurrence and use of an estuarine habitat by giant manta ray Manta birostris. Journal of Fish Biology 86(6):1830-1838.

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.

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.

Mercado III, E. 2018. The sonar model for humpback whale song revised. Frontiers in Psychology 9:20.

Merkens, K., A. Simonis, and E. Oleson. 2019. Geographic and temporal patterns in the acoustic detection of sperm whales *Physeter macrocephalus* in the central and western North Pacific Ocean. Endangered Species Research 39:115-133.

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.

Milessi, A. C., and M. C. Oddone. 2003. Primer registro de Manta birostris (Donndorff 1798)(Batoidea: Mobulidae) en el Rio de La Plata, Uruguay. Gayana (Concepción) 67(1):126-129.

Miller, J., and coauthors. 2023. Aquatic Invasive Species Report, Phase 2, Marine Mitigation in Support of P-209 Dry Dock 3 Replacement Joint Base Pearl Harbor-Hickam, Hawaii. Naval Facilities Engineering Systems Command Pacific for Commander, Navy Region Hawaii, Joint Base Pearl Harbor-Hickam, HI.

Miller, J. D., K. A. Dobbs, C. J. Limpus, N. Mattocks, and A. M. Landry Jr. 1998. Long-distance migrations by the hawksbill turtle, Eretmochelys imbricata, from north-eastern Australia. Wildlife Research 25(1):89-95.

Miller, M. H., and C. Klimovich. 2016. Endangered Species Act Status Review Report: Giant Manta (*Manta birostris*) and Reef Manta Ray (*Manta alfredi*). Draft Report to National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD. Pages 127 *in*.

Miller, M. H., and coauthors. 2014. Status review report: scalloped hammerhead shark (*Sphyrna lewini*). Final Report to NMFS, Office of Protected Resources.

Miller, M. H., and C. Klimovich. 2017. Endangered species act status review report: Giant manta ray (Manta biostris), reef manta ray (Manta alfredi). National Marine Fisheries Service.

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.

Millien, V., N. Stafiej, F. Félix, and H. M. Guzman. 2025. Migratory behaviour of humpback whales in the southeastern Pacific under climate change. Scientific Reports 15(1):3989.

Milliken, T., and H. Tokunaga. 1987. The Japanese sea turtle trade, 1970-1986. Center for Environmental Education.

Mintz, J. D. 2012. Vessel traffic in the Hawaii-Southern California and Atlantic fleet tresting and training study areas. Defense Technical Information Center.

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, Environment and Energy Team, Resource Analysis Division.

Mintz, J. D., and R. J. Filadelfo. 2011. 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.

MMC. 2007. Marine mammals and noise: A sound approach to research and management. Marine Mammal Commission.

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 Jr, J. R., S. S. Spitz, K. A. Forney, R. Grotefendt, and P. H. Forestell. 2000. Distribution and abundance of ondontocete species in Hawaiian waters: preliminary results of 1993-98 aerial surveys. Southwest Fisheries Science Center, National Marine Fisheries Service, P.O. Box 271, La Jolla, CA 92038.

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.

Moein, S. E., and coauthors. 1995. Evaluation of seismic sources for repelling sea turtles from hopper dredges. Pages 75-78 *in* L. Z. Hales, editor. Sea Turtle Research Program: Summary report. Prepared for U.S. Army Corps of Engineers, South Atlantic, Atlanta GA and U.S. Naval Submarine Base, Kings Bay, GA.

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.

Monnahan, C. C., T. A. Branch, K. M. Stafford, Y. V. Ivashchenko, and E. M. Oleson. 2014. Estimating historical eastern North Pacific blue whale catches using spatial calling patterns. PLoS ONE 9(6):e98974.

Monteiro, C. C., H. M. Carmo, A. J. Santos, G. Corso, and R. S. Sousa-Lima. 2019. First record of bioacoustic emission in embryos and hatchlings of hawksbill sea turtles (Eretmochelys imbricata). Chelonian Conservation and Biology: Celebrating 25 Years as the World's Turtle and Tortoise Journal 18(2):273-278.

Monzón-Argüello, C., C. Rico, A. Marco, P. López, and L. F. López-Jurado. 2010. Genetic characterization of eastern Atlantic hawksbill turtles at a foraging group indicates major undiscovered nesting populations in the region. Journal of Experimental Marine Biology and Ecology.

Moon, H. B., and coauthors. 2010. Concentrations and accumulation features of PCDDs, PCDFs and dioxin-like PCBs in cetaceans from Korean coastal waters. Chemosphere 79(7):733–739.

Mooney, T. A., and coauthors. 2010. Sound detection by the longfin squid (Loligo pealeii) studied with auditory evoked potentials: Sensitivity to low-frequency particle motion and not pressure. The Journal of Experimental Biology, 213, 3748–3759.

Mooney, T. A., J. E. Samson, A. D. Schlunk, and S. Zacarias. 2016. Loudness-dependent behavioral responses and habituation to sound by the longfin squid (Doryteuthis pealeii). Journal of Comparative Physiology A 202(7):489-501.

Moore, A. 2012. Records of poorly known batoid fishes from the north-western Indian Ocean (Chondrichthyes: Rhynchobatidae, Rhinobatidae, Dasyatidae, Mobulidae). African Journal of Marine Science 34(2):297-301.

Moore, J. E. 2021. Final report of the California Current Ecosystem Survey (CCES) 2018: a PacMAPPS study. U.S. Department of the Interior, Bureau of Ocean Energy Management, Camarillo, California.

Moore, J. E., and coauthors. 2009. A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. Marine Policy 33(3):435-451.

Moore, M. J., and J. M. Van der Hoop. 2012. The painful side of trap and fixed net fisheries: chronic entanglement of large whales. Journal of Marine Biology 2012.

Moore, M. M. 1980. Factors influencing survival of juvenile steelhead rainbow trout (Salmo gairdnerii gairdnerii) in the Ventura River. M.S. thesis. Humboldt State University, Arcata, California.

Moore, P. W. B., and R. J. Schusterman. 1987. Audiometric assessment of northern fur seals, Callorhinus ursinus. Marine Mammal Science 3(1):31-53.

Moore, S. E., and J. T. Clark. 2002. Potential impact of offshore human activities on gray whales (*Eschrichtius robustus*). Journal of Cetacean Research and Management 4(1):19-25.

Moore, S. K., N. J. Mantua, and E. P. Salathé. 2011. Past trends and future scenarios for environmental conditions favoring the accumulation of paralytic shellfish toxins in Puget Sound shellfish. Harmful Algae 10(5):521-529.

Moore, T. J., and coauthors. 2018. Exploring ship traffic variability off California. Ocean and Coastal Management 163:515-527.

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.

Mosca Angelucci, D., and M. C. Tomei. 2022. Uptake/release of organic contaminants by microplastics: A critical review of influencing factors, mechanistic modeling, and thermodynamic prediction methods. Critical Reviews in Environmental Science and Technology 52(8):1356-1400.

Mourier, J. 2012. Manta rays in the Marquesas Islands: first records of Manta birostris in French Polynesia and most easterly location of Manta alfredi in the Pacific Ocean, with notes on their distribution. Journal of Fish Biology 81(6):2053-2058.

Moyle, P. B. 2002. Inland fishes of California. Univ of California Press.

Mrosovsky, N. 1972. Spectrographs of the sounds of leatherback turtles. Herpetologica 28(3): 256-258.

Munday, P. L., and coauthors. 2009. Ocean acidification impairs olfactory discrimination and homing ability of a marine fish. Proceedings of the National Academy of Sciences of the United States of America 106(6):1848–52.

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.

Murray, K. T. 2009. Characteristics and magnitude of sea turtle bycatch in US mid-Atlantic gillnet gear. Endangered Species Research 8(3):211-224.

Murray, S. O., E. Mercado, and H. L. Roitblat. 1998. Characterizing the graded structure of false killer whale (Pseudorca crassidens) vocalizations. The Journal of the Acoustical Society of America 104(3):1679-1688.

Musick, J. A., and C. J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-163 *in* P. Lutz, and J. A. Musick, editors. The biology of sea turtles. CRC Press, Boca Raton, Florida.

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. M., and coauthors. 1998. Status review of chinook salmon from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC 35:443.

Myers, R. A., and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. Nature 423(6937):280.

Myrberg, A. A. 2001. The acoustical biology of elasmobranchs. Environmental Biology of Fishes 60(31-45).

Myrberg, A. A., C. R. Gordon, and A. P. Klimley. 1975a. Attraction of Free-Ranging Sharks by Acoustic Signals in the Near-Subsonic Range. Rosenstiel School of Marine and Atmospheric Science, TR75-4, University of Miami, Tech. Rept. to Office of Nav. Res., Contract No. N00014-67-A-0201-0008, Coral Gables, Florida.

Myrberg, A. A., C. R. Gordon, and A. P. Klimley. 1975b. Rapid Withdrawal from a Sound Source by Sharks under Open-Ocean and Captive Conditions. Rosenstiel School of Marine and Atmospheric Science, TR75-5 University of Miami, Tech. Rept. to Office of Nav. Res., Contract No. N00014-67-A-0201-0008.

Myrberg, A. A., C. R. Gordon, and A. P. Klimley. 1976. Attraction of free ranging sharks by low frequency sound, with comments on its biological significance. A. S. A. D. Hawkins, editor. Sound Reception in Fish. Elsevier, Amsterdam.

Myrberg, A. A., C. R. Gordon, and A. P. Klimley. 1978. Rapid withdrawal from a sound source by open-ocean sharks. The Journal of the Acoustical Society of America 64:1289-1297.

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, 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.

Nachtigall, P. E., and A. Y. Supin. 2008. A false killer whale adjusts its hearing when it echolocates. (Pseudorca crassidens). Journal of Experimental Biology 211(11):1714-1718.

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.

Nalesso, E., and coauthors. 2019. Movements of scalloped hammerhead sharks (Sphyrna lewini) at Cocos Island, Costa Rica and between oceanic islands in the Eastern Tropical Pacific. PLoS ONE 14(3):e0213741.

Nance, H. A., P. Klimley, F. Galvan-Magana, J. Martinez-Ortiz, and P. B. Marko. 2011. Demographic processes underlying subtle patterns of population structure in the scalloped hammerhead shark, Sphyrna lewini. PLoS ONE 6(7):e21459.

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.

National Academies of Sciences, E., and Medicine. 2016. Approaches to understanding the cumulative effects of stressors on marine mammals. National Academies Press.

National Marine Fisheries Service. 2011. Final Recovery Plan for the Sei Whale (*Balaenoptera borealis*). Pages 107 *in*. National Marine Fisheries Service Office of Protected Resources, Silver Spring, MD.

National Marine Fisheries Service. 2022. California Set Gillnet Observer Program Observed Catch Summary, January 1, 2007, through December 31, 2017. National Marine Fisheries Service, Silver Spring, MD.

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.

National Oceanic and Atmospheric Administration. 2016. Testing Detects Algal Toxins in Alaska Marine Mammals.

NAVFAC. 2024. Integrated Natural Resources Management Plan for Joint Base Pearl HarborHickam, O'ahu September 2024: Final. Naval Facilities Engineering Systems Command, Hawaii

Navy. 2013a. Hawaii-Southern California Training and Testing Activities Final Environmental Impact Statement/Overseas Environmental Impact Statement.

Navy. 2013b. San Diego Bay Integrated Natural Resources Management Plan, Final March 2013. U.S. Department of the Navy, Naval Facilities Engineering Command Southwest and Port of San Diego, San Diego, California.

Navy. 2013c. Water Range Sustainability Environmental Program Assessment, Potomac River Test Range Complex. Dahlgren, VA.

Navy. 2017. Atlantic Fleet Training and Testing Biological Assessment. Naval Facilities Engineering Command, Atlantic; Space and Naval Warfare Systems Command.

Navy. 2018. 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. 2021. Biological Assessment for Reinitiation of Formal Consultation Under Section 7 of the Endangered Species Act for Ongoing Military Readiness Activities in the Hawaii-Southern California Training and Testing Study Area. Commander, United States Pacific Fleet and Commander, Naval Sea Systems Command, Department of the Navy

Navy. 2022a. Navy Reply to the National Marine Fisheries Service 11 October 2022 Emailed Request for Information. Pages 5 *in* E. R. D. N. U.S. Pacific Fleet, editor Navy Memo.

Navy. 2022b. NMFS Follow Up Questions and Requests for Additional Information On the Navy's Biological Assessment for Re-Initiation of HSTT. Pages 8 *in*. Department of the Navy.

Navy. 2022c. Supplement NMFS Follow Up Questions and Requests for Additional Information On the Navy's Biological Assessment for Re-Initiation of HSTT. Pages 6 *in*. OPNAV N4154 Environmental Planning & Conservation Branch, Department of the Navy.

Navy. 2023. Biological Assessment for Sunflower Sea Star: Addendum to BA Submitted to NMFS on 2 December 2021 in Support of HSTT Reinitiation Commander, United States Pacific Fleet, Department of the Navy, Environmental Readiness Division and Navy System Commands.

Navy. 2024a. Dive Distribution and Group Size Parameters for Marine Species Occurring in the U.S. Navy's Atlantic and Hawaii-California Training and Testing Study Areas Naval Undersea Warfare Center Division Newport, Rhode Island

Navy. 2024b. Draft Environmental Impact Statement/Overseas Environmental Impact Statement Hawaii-California Training and Testing.

Navy. 2024c. Email from G. Thompson (Navy U.S. Fleet Forces) to R. Salz (NMFS) with title: *RE: NMFS Questions and Comments on Navy BA for Training and Testing of XLUUVs and USVs*.

Navy. 2024d. Hawaii-California Military Readiness Activities Biological Assessment to Support Endangered Species Act Section 7 Consultation with the National Marine Fisheries

Service (Final). United States Pacific Fleet, Navy Systems Commands, and U.S. Army, U.S. Marine Corps, U.S. Coast Guard, U.S. Air Force.

Navy. 2024e. Hawaii-California Training and Testing Draft Environmental Impact Statement/Overseas Environmental Impact Statement (Volume 3): Appendix C Biological Resources Supplemental Information. U.S. Department of the Navy, U.S. Marine Corps, U.S. Coast Guard, U.S. Department of the Army, and U.S. Air Force.

Navy. 2024f. Memo for the Record: Additional Information Supporting the US Navy's Biological Assessment for the Hawaii-California Training and Testing Study Area. U.S. Pacific Fleet N465CJ.

Navy. 2024g. Memo for the Record: Response to Email in Regard to US Navy Biological Assessment for the Hawaii-California Training and Testing Study Area. U.S. Pacific Fleet N465.

Navy. 2024h. Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training and Testing, Naval Undersea Warfare Center Division, Newport, Rhode Island.

Navy. 2025a. Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area: April 2025 (revised). Prepared By: Bioacoustic Analysis and Applied Research Team Naval Information Warfare Center Pacific.

Navy. 2025b. Additional Information Supporting the U.S. Navy's Biological Assessment for the Hawaii-California Training and Testing Study Area: Revised Ship Strike Probability Language. Department of the Navy, Commander, U.S. Pacific Fleet, Pearl Harbor, HI.

Navy. 2025c. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase 4). Naval Information Warfare Center Pacific, San Diego, CA.

Navy. 2025d. Final Environmental Impact Statement/Overseas Environmental Impact Statement Hawaii-California Training and Testing. United States Department of the Navy HCTT EIS/OEIS Project Manager Naval Facilities Engineering Command, Pacific/EV21 258 Makalapa Dr., Suite 100 Pearl Harbor, HI 96860-3134

Navy. 2025e. Hawaii-California Training and Testing Biological Assessment to Support Endangered Species Act Section 7 Consultation with the National Marine Fisheries Service: Addendum to Biological Assessment of 7 October 2024. United States Pacific Fleet, Navy Systems Commands, U.S. Army, U.S. Marine Corps, U.S. Coast Guard, and U.S. Air Force.

Navy. 2025f. Hawaii-California Training and Testing EIS/OEIS Final Version 2: Appendix E Acoustic and Explosive Effects Analysis for Marine Mammals, Reptiles, and Fishes in the Hawaii - California Training and Testing Study Area. Prepared By: Bioacoustic Analysis and Applied Research Team Naval Information Warfare Center Pacific.

Navy. 2025g. Hawaii-California Training and Testing Environmental Impact Statement/Overseas Environmental Impact Statement (Final Version 2): Appendix E.1 In-Air Acoustic Effects on Pinnipeds from Weapons Firing Noise. U.S. Department of the Navy, U.S. Marine Corps, U.S. Coast Guard, U.S. Department of the Army, and U.S. Air Force.

Navy. 2025h. HCTT_BA ITS Summ_ST_2025-05-08. U.S. Pacific Fleet, Environmental Readiness Division N465CJ.

Navy. 2025i. HCTT_EAR_ST_Takes_2025-05-08. U.S. Pacific Fleet, Environmental Readiness Division N465CJ.

Navy. 2025j. Memo for the Record: Additional Consultation Information Supporting the U.S. Navy's Hawaii-California Training and Testing Study Area. U.S. Pacific Fleet N465CJ.

Navy. 2025k. Memorandum for the Record: Additional Consultation Information Supporting the U.S. Navy's Hawaii-California Training and Testing Study Area. U.S. Navy Pacific Fleet N465CJ.

Navy. 2025l. Memorandum for the Record: Additional Consultation Information Supporting the U.S. Navy's Hawaii-California Training and Testing Study Area - Enclosure 1. Navy Reply to NMFS Geographic Mitigation Request for Main Hawaiian Island Insular False Killer Whale. U.S. Navy Pacific Fleet N465CJ.

Navy. 2025m. Memorandum for the Record: Additional Consultation Information Supporting the U.S. Navy's Hawaii-California Training and Testing Study Area - Enclosure 1. Navy Response to NMFS Email of 09 Jul 2025. U.S. Navy Pacific Fleet N465CJ.

Navy. 2025n. U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area. Technical Report September 2025, U.S. Pacific Fleet Environmental Readiness Division, Pearl Harbor, HI.

Nelms, S. E., W. E. D. Piniak, C. R. Weir, and B. J. Godley. 2016. Seismic surveys and marine turtles: An underestimated global threat? Biological Conservation 193:49-65.

Nemoto, T. 1964. School of baleen whales in the feeding areas. Scientific Reports of the Whales Research Institute Tokyo 18:89-110.

Neuman, M., K. L. Stierhoff, and D. Witting. 2017. Preliminary report on studies of white abalone (Haliotis sorenseni) survey methods, demographics, and behavior at San Clemente Island and Point Loma, CA. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Regional Office, Long Beach, CA.

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. 2015. The modelling and assessment of whale-watching impacts. Ocean & coastal management 115:10-16.

Newson, S. E., and coauthors. 2009. Indicators of the impact of climate change on migratory species. Endangered Species Research 7(2):101-113.

Nieukirk, S. L., and coauthors. 2012. Sounds from airguns and fin whales recorded in the mid-Atlantic Ocean, 1999–2009. The Journal of the Acoustical Society of America 131(2):1102–1112.

Nishizawa, H., and coauthors. 2014. Genetic composition of loggerhead turtle feeding aggregations: migration patterns in the North Pacific. Endangered Species Research 24(1):85-93.

Nitta, E., and J. R. Henderson. 1993. A reveiw of interactions between Hawaii's fisheries and protected species. Marine Fisheries Review 55(2):83-92.

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*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, Maryland.

NMFS. 2006. Biological opinion on the issuance of section lO(a)(l)(A) permits to conduct scientific research on the southern resident killer whale (Orcinus orca) distinct population

segment and other endangered or threatened species. National Marine Fisheries Service, Northwest Region, Seattle, Washington.

NMFS. 2008. 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. 2009. 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. 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*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS. 2011. Reducing Vessel Strikes of Large Whales in California. NOAA, NMFS, Southeast Regional Office.

NMFS. 2012a. Final Recovery Plan for Central California Coast coho salmon Evolutionarily Significant Unit. National Marine Fisheries Service, Southwest Region, Santa Rosa, California.

NMFS. 2012b. Southern California steelhead recovery plan. National Marine Fisheries Service Southwest Regional Office, Long Beach, CA.

NMFS. 2013a. Biological opinion on the continued management of the drift gillnet fishery under the FMP for U.S. West Coast fisheries for highly migratory species. NOAA, National Marine Fisheries Service, Southwest Region, Protected Resources Division.

NMFS. 2013b. Hawksbill sea turtle (*Eremochelys imbricata*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS. 2013c. South-Central California steelhead recovery plan.

NMFS. 2014a. Final Recovery Plan for the Southern Oregon/Northern California Coast Evolutionarily Significant Unit of Coho Salmon (*Oncorhynchus kisutch*). National Marine Fisheries Service, Arcata, CA.

NMFS. 2014b. 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. ESA Section 7 Biological Opinion on U.S. Navy's Training Exercises and Testing Activities in the Hawaii-Southern California Training and Testing Study Area. Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service.

NMFS. 2015b. Marine Turtle Strandings in the Hawaiian Islands January – December 2014. Protected Species Division, Turtle Research Program, NOAA Pacific Islands Fisheries Science Center.

NMFS. 2015c. Sperm Whale (*Physeter macrocephalus*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources

Silver Spring, MD.

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 and Evaluation of Eulachon. West Coast Region, Portland, OR.

NMFS. 2016e. ESA Section 7(a)(2) Biological Opinion for the Continued Opertation of the West Coast-based Deep-set Longline Fishery managed under the FMP for U.S. West Coast HMS. NOAA, National Marine Fisheries Service, West Coast Region.

NMFS. 2016f. Final Coastal Multispecies Recovery Plan. National Marine Fisheries Service, West Coast Region, Santa Rosa, California.

NMFS. 2016g. Five-Year Review: Summary and Evaluation of Southern California Coast Steelhead Distinct Population Segment. National Marine Fisheries Service. West Coast Region. California Coastal Office, Long Beach, California.

NMFS. 2017a. 2016 West Coast Entanglement Summary: Overview of Entanglement Data. U.S. Department of Commerce, NOAA, National Marine Fisheries Service, West Coast Region.

NMFS. 2017b. 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, Silver Spring, Maryland.

NMFS. 2017c. Endangered Species Act Recovery Plan for the Southern Distinct Population Segment of Eulachon (*Thaleichthys pacificus*). Prepared by: National Marine Fisheries Service, West Coast Region.

NMFS. 2017d. Recovery Plan for the Southern Distinct Population Segment of Eulachon (*Thaleichthys pacificus*). P. R. D. NMFS West Coast Region, editor, Portland, Oregon.

NMFS. 2018a. 2018 Revision to: Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (Version 2.0).

NMFS. 2018b. Biological Opinion on U.S. Navy Hawaii-Southern California Training and Testing and the National Marine Fisheries Service's Promulgation of Regulations Pursuant to the Marine Mammal Protection Act for the Navy to "Take" Marine Mammals Incidental to Hawaii-Southern California Training and Testing. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

NMFS. 2018c. Designation of critical habitat for the endangered Main Hawaiian Islands Insular false killer whale Distinct Population Segment. Biological Report. 73 pp.

NMFS. 2019a. 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. 2019b. 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. 2019c. Endangered Species Act (ESA) Section 7 Biological Opinion (1) United States Navy's Surveillance Towed Array Sensor System Low Frequency Active Sonar Routine Training and Testing Activities in the Western and Central North Pacific and Eastern Indian Oceans from August 2019 and continuing into the reasonably foreseeable future (2) National Oceanic and Atmospheric Administration's National Marine Fisheries Service, Office of Protected Resources, Permits and Conservation Division's Promulgation of Regulations and Issuance of a Letter of Authorization for the United States Navy to "Take" Marine Mammals Incidental to Surveillance Towed Array Sensor System Low Frequency Active Sonar routine training and testing activities in the Western and Central North Pacific and Eastern Indian Oceans from August 2019 to August 2026. Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

NMFS. 2019d. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion: Continued Authorization of the Hawaii Pelagic Shallow-Set Longline Fishery National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division

NMFS. 2019e. Fin Whale (*Balaenoptera physalus*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office Protected Resources, Silver Spring, MD.

NMFS. 2019f. Giant Manta Ray Recovery Outline.

NMFS. 2020a. 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. 2020b. Loggerhead Turtle Research.

NMFS. 2020c. NOAA Announces Regions for First Two Aquaculture Opportunity Areas under Executive Order on Seafood.

NMFS. 2020d. Recovery Plan for the Blue Whale (*Balaenoptera musculus*): First Revision to the July 1998 Recovery Plan for the Blue Whale. Office of Protected Resources and West Coast Region, National Marine Fisheries Service.

NMFS. 2020e. Scalloped Hammerhead Shark (Sphyrna lewini) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service Office of Protected Resources, Silver Spring, MD.

NMFS. 2021a. 2015–2021 Guadalupe Fur Seal Unusual Mortality Event in California, Oregon and Washington. Marine Life in Distress.

NMFS. 2021b. 2020 West Coast Whale Entanglement Summary. U.S. Department of Commerce, National Oceanic & Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

NMFS. 2021c. Final Endangered Species Act Recovery Plan for the Main Hawaiian Islands Insular False Killer Whale (Pseudorca crassidens) Distinct Population Segment. NOAA Fisheries Pacific Islands Regional Office, Honolulu, HI.

NMFS. 2021d. Main Hawaiian Islands Insular False Killer Whale (*Pseudorca crassidens*) Distinct Population Segment: Final Endangered Species Act (ESA) Recovery Plan. NOAA Fisheries Pacific Islands Regional Office.

NMFS. 2021e. Recovery status review for the main Hawaiian Islands insular false killer whale (*Pseudorca crassidens*) distinct population segment. NOAA Fisheries, Pacific Islands Regional Office, Honolulu, HI 96818.

NMFS. 2021f. Revision of the Critical Habitat Designation for Southern Resident Killer Whales: Final Biological Report (to accompany the Final Rule). National Marine Fisheries Service West Coast Region.

NMFS. 2021g. Sei Whale (*Balaenoptera borealis*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2022a. 2022 5-Year Review: Summary & Evaluation of Eulachon, Southern DPS National Marine Fisheries Service West Coast Region

NMFS. 2022b. Endangered Species Act (ESA) Section 7 Biological Opinion on (1) U.S. Navy Point Mugu Sea Range (PMSR) Testing and Training Activities; 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 PMSR Activities from February 2022 through February 2029. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

NMFS. 2022c. Main Hawaiian Islands Insular False Killer Whale (Pseudorca crassidens) Distinct Population Segment 5-Year Review. NOAA Fisheries Pacific Islands Regional Office.

NMFS. 2022d. Recovery Outline for the Central America, Mexico, and Western North Pacific Distinct Population Segments of Humpback Whales. NOAA National Marine Fisheries Service.

NMFS. 2023a. 2015–2016 Large Whale Unusual Mortality Event in the Western Gulf of Alaska, United States and British Columbia (Closed). Marine Life in Distress. NOAA Fisheries.

NMFS. 2023b. 2023 5-Year Review: Summary & Evaluation of Southern California Steelhead. National Marine Fisheries Service West Coast Region.

NMFS. 2023c. Draft Biological Report for the Designation of Marine Critical Habitat for Six Distinct Population Segments of the Green Turtle, *Chelonia mydas*. National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

NMFS. 2023d. Draft Recovery Plan for the Oceanic Whitetip Shark (*Carcharhinus longimanus*). NMFS, Office of Protected Resources, Silver Spring, MD.

NMFS. 2023e. DRAFT U.S. Pacific Marine Mammals Stock Assessments: 2022. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. 2023f. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion Authorization of the Hawaii Deep-Set Longline Fishery. National Marine Fisheries Service, Pacific Islands Region.

NMFS. 2023g. Endangered Species Act Recovery Status Review for the Oceanic Whitetip Shark (*Carcharhinus longimanus*). NOAA, National Marine Fisheries Service.

NMFS. 2023h. Endangered Species Act Section 7(a)(2) Biological Opinion: Reinitiation of ESA Consultation on the Continued Operation of the Large Mesh Drift Gillnet Fishery under the U.S. West Coast Fishery Management Plan for Highly Migratory Species. NMFS West Coast Region (WCR) Sustainable Fisheries Division.

NMFS. 2023i. Gray Whale, Western North Pacific Distinct Population Segment (*Eschrichtius robustus*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Office of Protected Resources Silver Spring, MD.

NMFS. 2023j. Memorandum: Occurrence and Distribution of Eastern North Pacific and Western North Pacific Populations of Gray Whales in the North Pacific Ocean. Pages 13 *in* N. M. F. S. NOAA, Office of Protected Resources, editor. Memorandum from Kimberly Damon-Randall, Director, Office of Protected Resources, Silver Spring, MD.

NMFS. 2024a. 2023 Annual Report on the National Marine Fisheries Service Cetacean Research and Enhancement Permitting Program. Permits and Conservation Division, Office of Protected Resources, National Marine Fisheries Service.

NMFS. 2024b. Concurrence Letter and Conference for the Endangered Species Act Section 7 Informal Consultation for the Training and Testing of the Extra Large Unmanned Undersea Vehicles and Unmanned Surface Vehicles at Naval Base Ventura County, Port Hueneme, California NOAA, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2024c. Endangered Species Act (ESA) Recovery Plan Oceanic Whitetip Shark.

NMFS. 2024d. Large whale stranding data for California from 2017 through May 2024. N. W. C. S. Team, editor. Marine Mammal Health and Stranding Response Program.

NMFS. 2024e. North Pacific Right Whale (*Eubalaena japonica*) Five-Year Review: Summary and Evaluation National Marine Fisheries Service, Alaska Regional Office, Protected Resources Division Anchorage, Alaska

NMFS. 2024f. Reinitiation of the Biological and Conference Opinion on (1) U.S. Navy Hawaii-Southern California Training and Testing, 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 Hawaii-Southern California Training and Testing Through December 2025. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

NMFS. 2024g. Update to: Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (Version 3.0): Underwater and In-Air Criteria for Onset of Auditory Injury and Temporary Threshold Shifts. U.S. Dept. of Commerce, NOAA.

NMFS. 2025a. 5-Year Review of the Oceanic Whitetip Shark (*Carcharhinus longimanus*). National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2025b. ESA Section 7(a)(2) Biological Opinion: Supplement to the Authorization of the U.S. West and Central Pacific Purse Seine Fishery; Effects on Oceanic Whitetip Sharks National Marine Fisheries Service. Pacific Islands Region, Protected Resources Division.

NMFS, and USFWS. 1991. Recovery Plan for U.S. Pacific Populations of Green Sea Turtle (*Chelonia mydas*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Department of the Interior, U.S. Fish and Wildlife Service, National Marine Fisheries Service, Silver Spring, MD.

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 U.S. Pacific Populations of the East Pacific Green Turtle (*Chelonia mydas*). National Oceanic and Atmospheric Administration,

National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1998c. Recovery plan for U.S. Pacific populations of the green turtle (*Chelonia mydas*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1998d. Recovery plan for U.S. Pacific populations of the hawksbill turtle (*Eretmochelys imbricata*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, MD.

NMFS, and USFWS. 1998e. Recovery plan for U.S. Pacific populations of the loggerhead turtle (Caretta caretta). National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1998f. Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle (*Lepidochelys olivacea*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2013. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2014. Olive ridley sea turtle (*Lepidochelys olivacea*) 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS, and USFWS. 2020a. 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.

NMFS, and USFWS. 2020b. Endangered Species Act status review of the leatherback turtle (Dermochelys coriacea) 2020.

NMFS, and USFWS. 2020c. Loggerhead Sea Turtle (Caretta caretta) North Pacific Ocean DPS 5-Year Review: Summary and Evaluation. . National Marine Fisheries Service, Office of Protected Resources and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

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. 2014 Report on the Occurrence and Health Effects of Anthropogenic Debris Ingested by Marine Organisms. NOAA Marine Debris Program. Silver Spring, MD. 19 pp.

NOAA. 2016. Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Silver Spring, Maryland.

NOAA. 2020. 2019 West Coast Whale Entanglement Summary. Pages 4 *in* N. F. W. C. Region, editor.

NOAA. 2021. Species in the Spotlight Priority Actions: 2021-2025 Pacific Leatherback Turtle *Dermochelys coriacea*. NOAA Fisheries.

NOAA. 2023. NOAA Fisheries News: Hawaiian Monk Seal Population Rounds Out a Decade of Growth (May 10, 2023). Web page: https://www.fisheries.noaa.gov/feature-story/hawaiian-monk-seal-population-rounds-out-decade-growth#:~:text=Thanks%20to%20these%20surveys%2C%20we,2022%2C%20the%20population%20crested%201%2C600. NOAA Fisheries, Pacific Islands Fisheries Science Center.

NOAA National Centers for Coastal Ocean Science. 2007. A Biogeographic Assessment off North/Central California: In Support of the National Marine Sanctuaries of Cordell Bank, Gulf of the Farallones and Monterey Bay. Phase II – Environmental Setting and Update to Marine Birds and Mammals.

NOAA Office of Response and Restoration. 2021. Environmental Response Management Application.

Norem, A. D. 2005. Injury assessment of sea turtles utilizing the neritic zone of the southeastern United States. University of Florida, Gainesville.

Noriega, R., J. M. Werry, W. Sumpton, D. Mayer, and S. Y. Lee. 2011. Trends in annual CPUE and evidence of sex and size segregation of Sphyrna lewini: Management implications in coastal waters of northeastern Australia. Fisheries Research 110(3):472-477.

Normandeau, Exponent, T. T., and A. Gill. 2011. Effects of EMFs from undersea powerbcables on elasmobranchs and other marine species. U.S. Department of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Pacific OCS 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. 2017. Updated abundance estimate for Guadalupe fur seals. Personal communication on August 18, 2017, between Tenaya Norris (The Marine Mammal Center) and Michael Zickel (Mantech International) via email.

Norris, T. 2019. Guadalupe Fur Seal Population Census and Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean. The Marine Mammal Center, Sausalito, California.

Northridge, S., A. Coram, A. Kingston, and R. Crawford. 2017. Disentangling the causes of protected-species bycatch in gillnet fisheries. Conservation Biology 31(3):686-695.

Noss, R. F., and coauthors. 2020. Improving species status assessments under the U.S. Endangered Species Act and implications for multispecies conservation challenges worldwide. Conservation Biology 2021(35):1715–1724.

Notarbartolo-di-Sciara, G., and E. V. Hillyer. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia:607-614.

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.

NRC. 1990. Sea turtle mortality associated with human activities. National Academy Press, National Research Council Committee on Sea Turtle Conservation, Washington, D.C.

NRC. 2003. Ocean Noise and Marine Mammals. National Academies Press.

NRC. 2005. Marine mammal populations and ocean noise. Determining when noise causes biologically significant effects. National Academy of Sciences, Washington, D. C.

NWFSC. 2015. 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'Hara, J., and J. R. Wilcox. 1990. Avoidance responses of loggerhead turtles, *Caretta caretta*, to low frequency sound. Copeia 2:564-567.

O'Malley, M. p., K. A. Townsend, P. Hilton, S. Heinrichs, and J. D. Stewart. 2017. Characterization of the trade in manta and devil ray gill plates in China and South-east Asia through trader surveys. Aquatic Conservation: Marine and Freshwater Ecosystems 27(2):394-413.

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.

O'Connor, S., R. Campbell, H. Cortez, and T. Knowles. 2009. Whale Watching Worldwide: tourism numbers, expenditures and expanding economic benefits, a special report from the International Fund for Animal Welfare. Yarmouth MA, USA, prepared by Economists at Large 228.

Occhipinti-Ambrogi, A. 2021. Biopollution by Invasive Marine Non-Indigenous Species: A Review of Potential Adverse Ecological Effects in a Changing Climate. International Journal of Environmental Research and Public Health 2021(18).

Ocean Alliance. 2010. The Voyage of the Odyssey: Executive Summary.

Ocean Conservancy. 2010. Trash travels: from our hands to the sea, around the globe, and through time. The Ocean conservancy.

Oedekoven, C., and L. Thomas. 2022. Effectiveness of Navy lookout teams in detecting cetaceans. Provided to HDR Inc.

Oestreich, W. K., and coauthors. 2022. Acoustic signature reveals blue whales tune life-history transitions to oceanographic conditions. Functional Ecology 36(4):882-895.

Oestreich, W. K., and coauthors. 2020. Animal-borne metrics enable acoustic detection of blue whale migration. Current Biology 30(23):4773-4779. e3.

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.

Ohsumi, S., and S. Wada. 1974. Status of whale stocks in the North Pacific, 1972. Report of the International Whaling Commission 24:114-126.

Oleson, E. M., and coauthors. 2010. Status review of Hawaiian insular false killer whales (Pseudorca crassidens) under the Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

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., 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, C., and coauthors. 2022. Exposure to whale watching vessels affects dive ascents and resting behavior in sperm whales. Frontiers in Marine Science 9:2189.

Oliver, E. C., and coauthors. 2018. Longer and more frequent marine heatwaves over the past century. Nature Communications 9(1):1-12.

Olson, J. K. 2013. The effect of human exposure on the anti–predatory response of harbor seals (*Phoca vitulina*).

Oros, J., O. M. Gonzalez-Diaz, and P. Monagas. 2009. High levels of polychlorinated biphenyls in tissues of Atlantic turtles stranded in the Canary Islands, Spain. Chemosphere 74(3):473-478.

Orós, J., A. Torrent, P. Calabuig, and S. Déniz. 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998–2001). Diseases of aquatic organisms 63(1):13-24.

Oswald, J. N., J. Barlow, and T. F. Norris. 2003. Acoustic identification of nine delphinid species in the eastern tropical Pacific Ocean. Marine Mammal Science 19(1):20-037.

Ou, M., and coauthors. 2015. Responses of pink salmon to CO 2-induced aquatic acidification. Nature Climate Change 5(10):950.

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.

Packard, A., H. E. Karlsen, and O. Sand. 1990. Low frequency hearing in cephalopods. Journal of Comparative Physiology A, 166, 501–505.

Pacoureau, N., and coauthors. 2021. Half a century of global decline in oceanic sharks and rays. Nature 589(7843):567-571.

Páez-Rosas, D., D. A. Pazmiño, and M. Riofrío-Lazo. 2020. Unprecedented records of Guadalupe and Juan Fernández fur seals in the Galapagos Archipelago. Aquatic Mammals 46(6):549-555.

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.

Pankaew, K., and S. L. Milton. 2018. The effects of extended crawling on the physiology and swim performance of loggerhead and green sea turtle hatchlings. Journal of Experimental Biology 221(1):jeb165225.

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.

Parks, S. E. 2003. Response of North Atlantic right whales (Eubalaena glacialis) to playback of calls recorded from surface active groups in both the North and South Atlantic. Marine Mammal Science 19(3):563-580.

Parks, S. E. 2011. Assessment of acoustic adaptations for noise compensation in marine mammals. Office of Naval Research.

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. Journal of the Acoustical Society of America 122(6):3725-3731.

Parks, S. E., I. Urazghildiiev, and C. W. Clark. 2009. Variability in ambient noise levels and call parameters of North Atlantic right whales in three habitat areas. Journal of the Acoustical Society of America 125(2):1230-1239.

Parrish, F. A., M. P. Craig, T. J. Ragen, G. J. Marshall, and B. M. Buhleier. 2000. Identifying diurnal foraging habitat of endangered Hawaiian monk seals using a seal-mounted video camera. Marine Mammal Science 16(2):392-412.

Pashin, Y. V., and L. M. Bakhitova. 1979. Mutagenic and Carcinogenic Properties of Polycyclic Aromatic Hydrocarbons. Environmental Health Perspectives 30:185-189.

Pastene, L. A., M. Goto, M. Taguchi, and T. Kitakado. 2016. Genetic analyses based on mtDNA control region sequencing and microsatellite DNA confirmed the occurrence of a single stock of sei whales in oceanic regions of the North Pacific. International Whaling Commission.

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.

Patrício, A. R., and coauthors. 2017. Balanced primary sex ratios and resilience to climate change in a major sea turtle population. Marine Ecology Progress Series 577:189-203.

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.

Payne, P., J. Nicholas, L. O'Brien, and K. 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. Fisheries Bulletin 84:271-277.

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 selected prey. Fishery Bulletin 88:687-696.

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. 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. 2002. Marine nekton off Oregon and the 1997–98 El Nino. Progress in Oceanography 54(1-4):399-403.

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.

Peñaherrera-Palma, C., and coauthors. 2018. Evaluating abundance trends of iconic species using local ecological knowledge. Biological Conservation 225:197-207.

Peng, C., X. Zhao, and G. Liu. 2015. Noise in the sea and its impacts on marine organisms. International Journal of Environmental Research and Public Health 12(10):12304-12323.

Perazio, C. E., and E. Mercado III. 2018. Singing humpback whales *Megaptera novaeangliae* favor specific frequency bands. Proceedings of Meetings on Acoustics, Victoria, Canada.

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.

Petrochenko, S. P., A. S. Potapov, and V. V. Pryadko. 1991. Sounds, souce levels, and behavior of gray whales in the Chukotskoe Sea. Sov. Phys. Acoust. 37(6):622-624.

Petrosky, C., and H. Schaller. 2010. Influence of river conditions during seaward migration and ocean conditions on survival rates of Snake River Chinook salmon and steelhead. Ecology of Freshwater Fish 19(4):520-536.

Philippart, C. J. M., and coauthors. 2011. Impacts of climate change on European marine ecosystems: Observations, expectations and indicators☆. Journal of Experimental Marine Biology and Ecology.

Phu, J. L., and P. Palaniappan. 2019. Recaptured wild green turtles (Chelonia mydas) with newly documented boat strike injuries in Mabul Island, Sabah, Malaysia. Chelonian Conservation and Biology: Celebrating 25 Years as the World's Turtle and Tortoise Journal 18(2):265-272.

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.

PIFSC. 2017. 2016 Report on Hawaiian Monk Seal Vaccination Program. NOAA, NMFS, Pacifc Islands Fishery Science Center.

PIFSC. 2018. Health and Disease Research and Emergency Response Initiative. NOAA, NMFS, Pacific Island Fisheries Science Center.

PIFSC. 2025. Hawaiian sea turtle strandings from 2019-2024. NOAA, NMFS, Pacific Islands Fisheries Science Center, Marine Turtle Biology and Assessment Program.

Pike, D. A. 2014. Forecasting the viability of sea turtle eggs in a warming world. Global change biology 20(1):7–15.

Pike, D. A., E. A. Roznik, and I. Bell. 2015. Nest inundation from sea-level rise threatens sea turtle population viability. Royal Society Open Science 2:150127.

Pike, G. C., and I. B. Macaskie. 1969. Marine mammals of British Columbia. Bulletin of the Fisheries Research Board of Canada 171:1-54.

Pilcher, N. 2011. Community-based conservation of leatherback turtles along the Huon coast, Papua New Guinea, Honolulu, Hawaii.

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 52:273–288.

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.

Piniak, W. E., D. A. Mann, C. A. Harms, T. T. Jones, and S. A. Eckert. 2016. Hearing in the Juvenile Green Sea Turtle (*Chelonia mydas*): A Comparison of Underwater and Aerial Hearing Using Auditory Evoked Potentials. PLoS ONE 11(10):e0159711.

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., and coauthors. 2022. From individual responses to population effects: Integrating a decade of multidisciplinary research on blue whales and sonar. Animal Conservation.

Pirotta, E., and coauthors. 2021. Context-dependent variability in the predicted daily energetic costs of disturbance for blue whales. Conservation Physiology 9(1).

Pirotta, E., and coauthors. 2018. Understanding the population consequences of disturbance. Ecology and Evolution 8(19):9934-9946.

Pirotta, E., and coauthors. 2019. Anthropogenic disturbance in a changing environment: modelling lifetime reproductive success to predict the consequences of multiple stressors on a migratory population. Oikos 128(9):1340-1357.

PISC. 2016. Marine Debris: Removal and Assessment in the Northwestern Hawaiian Islands: 2015. NOAA Fisheries Pacific Islands Fisheries Science Center.

Plotkin, P. 2003. Adult migrations and habitat use. Pages 225-241 *in* L. Lutz, J. A. Musick, and J. Wyneken, editors. Biology of sea turtles, volume II. CRC Press, Boca Raton, Florida.

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.

Poloczanska, E. S., and coauthors. 2016. Responses of marine organisms to climate change across oceans. Frontiers in Marine Science 3:62.

Poloczanska, E. S., C. J. Limpus, and G. C. Hays. 2009. Vulnerability of marine turtles to climate change. Advances in Marine Biology 56:151–211.

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 UK1. Journal of Unmanned Vehicle Systems 3(3):102–113.

Popov, V. V., and coauthors. 2013. Hearing threshold shifts and recovery after noise exposure in beluga whales, *Delphinapterus leucas*. The Journal of Experimental Biology 216(9):1587–1596.

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. 2016. Effects of seismic air guns on pallid sturgeon and paddlefish. Pages 871-878 *in* The Effects of Noise on Aquatic Life II. Springer.

Popper, A. N., and coauthors. 2007. 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. Sound exposure guidelines for fishes and sea turtles: A technical report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI. Springer, New York, NY.

Popper, A. N., M. Salmon, and K. W. Horch. 2001. Acoustic detection and communication by decapod crustaceans. Journal of Comparative Physiology A 187(2):83-89.

Popper, A. N., and C. R. Schilt. 2009. Hearing and acoustic behavior: Basic and applied considerations. Pages 17-48 *in* J. F. Webb, R. R. Fay, and A. N. Popper, editors. Fish Bioacoustics.

Popper, M. B. H., T. Carlson, and A. N. 2013. Effects of exposure to pile-driving sounds on fish. Bioacoustics 17:305-307.

Possatto, F. E., M. Barletta, M. F. Costa, J. A. I. do Sul, and D. V. Dantas. 2011. Plastic debris ingestion by marine catfish: An unexpected fisheries impact. Marine Pollution Bulletin 62(5):1098–1102.

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.

Prentice, M. B., and coauthors. 2025. Vibrio pectenicida strain FHCF-3 is a causative agent of sea star wasting disease. Nature Ecology & Evolution:1-13.

Price, B., and C. Van Salisbury. 2007. Low-Profile Gillnet Testing in the Deep Water Region of Pamlico Sound, NC. North Carolina Department of Environment and Natural Resources, Division of Marine Fisheries, Morehead City, NC.

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. NOAA Technical Memorandum, NOS NCCOS 164.

Price, C. S., and coauthors. 2016. Protected Species & Longline Mussel Aquaculture Interactions. NOAA Technical Memorandum, 211.

Price, E. R., and coauthors. 2004. Size, growth, and reproductive output of adult female leatherback turtles *Dermochelys coriacea*. Endangered Species Research 5:1-8.

Prieto, R., D. Janiger, M. A. Silva, G. T. Waring, and J. M. Goncalves. 2012. The forgotten whale: a bibliometric analysis and literature review of the North Atlantic sei whale Balaenoptera borealis. Mammal Review 42(3):235.

Pughiuc, D. 2010. Invasive species: Ballast water battles. Seaways (March 2010):5–7.

Putman, N. F., J. Hawkins, and B. J. Gallaway. 2020. Managing fisheries in a world with more sea turtles. Proceedings of the Royal Society B 287(1930):20200220.

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.

Queiroz, N., and coauthors. 2019. Global spatial risk assessment of sharks under the footprint of fisheries. Nature 572(7770):461-466.

Queiroz, N., and coauthors. 2016. Ocean-wide tracking of pelagic sharks reveals extent of overlap with longline fishing hotspots. Proceedings of the National Academy of Sciences 113(6):1582-1587.

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. Journal of Marine Design and Operations B4:2–10.

Raaymakers, S., and R. R. Hilliard. 2002. Harmful aquatic organisms in ships' ballast water - Ballast water risk assessment. CIESM: The Mediterranean Science Commission, Villa Girasole, Monaco.

Rabalais, N. N., R. E. Turner, and D. Scavia. 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. Bioscience 52(2):129-142.

Ragen, T. J. 1999. Human activities affecting the population trends of the Hawaiian monk seal. American Fisheries Society Symposium 23:183 - 194.

Raimondi, P. 2012. Estimation of population size of the black abalone, Haliotis cracherodii, on San Clemente Island. D. o. E. a. E. B. University of Santa Cruz, editor, Santa Cruz, CA.

Ramanayaka, S., O. Hettithanthri, S. Sandanayake, and M. Vithanage. 2022. Ecological Effects of Chemical Contaminants Adsorbed to Microplastics. Pages 1019-1048 *in* Handbook of Microplastics in the Environment. Springer.

Rambahiniarison, J. M., and coauthors. 2018. Life history, growth, and reproductive biology of four mobulid species in the Bohol Sea, Philippines. Frontiers in Marine Science 5:269.

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.

Rassweiler, A., R. J. Schmitt, and S. J. Holbrook. 2010. Triggers and maintenance of multiple shifts in the state of a natural community. Oecologia 164:489-498.

Raven, J., and coauthors. 2005. Ocean acidification due to increasing atmospheric carbon dioxide. The Royal Society.

Read, A. J., P. Drinker, and S. Northridge. 2006. Bycatch of marine mammals in US and global fisheries. Conservation Biology 20(1):163-169.

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.

Redfern, J. V., E. A. Becker, and T. J. Moore. 2020. Effects of variability in ship traffic and whale distributions on the risk of ships striking whales. Frontiers in Marine Science:793.

Redfern, J. V., and coauthors. 2017. Assessing the risk of chronic shipping noise to baleen whales off Southern California, USA. Endangered Species Research 32:153-167.

Redfern, J. V., and coauthors. 2019. Evaluating stakeholder-derived strategies to reduce the risk of ships striking whales. Diversity and Distributions 25(10):1575-1585.

Reeves, R. R., G. K. Silber, and P. M. Payne. 1998. Draft Recovery Plan for the Fin Whale Balaenoptera physalus and Sei Whale Balaenoptera borealis. Silver Spring, MD.

Reeves, R. R., and H. Whitehead. 1997. Status of the sperm whale, *Physeter macrocephalus*, in Canada. The Canadian Field-Naturalist 111(2):15.

Reina, R. D., J. R. Spotila, F. V. Paladino, and A. E. Dunham. 2008. Changed reproductive schedule of eastern Pacific leatherback turtles Dermochelys coriacea following the 1997–98 El Niño to La Niña transition. Endangered Species Research.

Renan de Deus Santos, M., and coauthors. 2017. Stress Response of Juvenile Green Sea Turtles (Chelonia mydas) with Different Fibropapillomatosis Scores. Journal of Wildlife Diseases.

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.

Reusch, T. B., and S. L. Williams. 1998. Variable responses of native eelgrass Zostera marina to a non-indigenous bivalve Musculista senhousia. Oecologia 113(3):428-441.

Reusch, T. B., and S. L. Williams. 1999. Macrophyte canopy structure and the success of an invasive marine bivalve. Oikos:398-416.

Rey-Baquero, M. P., and coauthors. 2021. Understanding effects of whale-watching vessel noise on humpback whale song in the North Pacific coast of Colombia with propagation models of masking and acoustic data observations. Frontiers in Marine Science 8:623724.

Reyff, J. A. 2008. Underwater sound pressure levels associated with marine pile driving: assessment of impacts and evaluation of control measures. Journal of Transportation Research Board CD 11-S:481-490.

Reyff, J. A. 2012. Underwater sounds from unattenuated and attenuated marine pile driving. Pages 439-444 *in* A. N. Popper, and A. D. Hawkins, editors. The effects of noise on aquatic life. Springer Science + Business Media, LLC, New York, NY.

Rice, A. C., and coauthors. 2018. Passive Acoustic Monitoring for Marine Mammals in the SOCAL Range Complex April 2016–June 2017. Marine Physical Laboratory, Scripps Institution of Oceanography, La Jolla, California.

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, J., F. Carvalho, M. Fitchett, S. Harley, and A. Ishizaki. 2021. Future Stock Projections of Oceanic Whitetip Sharks in the Western and Central Pacific Ocean. Report to the WCPFC Scientific Committee. Seventeenth Regular Session.

Rice, J., and S. Harley. 2012. Stock assessment of oceanic whitetip sharks in the western and central Pacific Ocean. Western and Central Pacific Fisheries Commission Scientific Committee Eighth Regular Session.WCPFC-SC8-2012/SA-WP-06 Rev 1., 53. Pages 53 *in*.

Richardson, A. J., R. J. Matear, and A. Lenton. 2017. Potential impacts on zooplankton of seismic surveys. CSIRO, Australia.

Richardson, W. J., and coauthors. 1995a. Assessment of potential impact of small explosions in the Korea Strait on marine animals and fisheries. LGL Ltd. Environmental Research Associates, BBN Systems and Technologies.

Richardson, W. J., M. A. Fraker, B. Wursig, and R. S. Wells. 1985. 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. 1995b. Marine Mammals and Noise. Academic Press, San Diego, CA.

Richardson, W. J., and coauthors. 1995c. Acoustic effects of oil production activities on bowhead and white whales visible during spring migration near Pt. Barrow, Alaska - 1991 and 1994 phases: Sound propagation and whale responses to playbacks of icebreaker noise. U.S. Department of the Interior, Minerals Management Service.

Richerson, K., and coauthors. 2020. Observed and Estimated Total of Green Sturgeon in the 2002–2019 US West Coast Fisheries. National Marine Fisheries Service, Silver Spring, MD.

Richerson, K., and coauthors. 2022. Observed and Estimated Bycatch of Salmon in U.S. West Coast Fisheries, 2002–20. National Marine Fisheries Service, Silver Spring, MD.

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. Science for Conservation 219.

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.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. 1969a. Hearing in the giant sea turtle, *Chelonia mydas*. Proceedings of the National Academy of Science 64:884-890.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. 1969b. Hearing in the giant sea turtle, Chelonoa mydas. Proceedings of the National Academies of Science 64.

Rigby, C., and coauthors. 2019a. Oceanic whitetip shark (*Carcharhinus longimanus*): Supplementary Information for *Carcharhinus longimanus*. The IUCN Red List of Threatened Species: Carcharhinus longimanus–published in 2019:12.

Rigby, C. L., N. K. Dulvy, R. Barreto, J. Carlson, and e. al. 2019b. Sphyrna lewini. The IUCN Red List of Threatened Species 2019 e.T39385A2918526.

Rivers, J. A. 1997. Blue whale, *Balaenoptera musculus*, vocalizations from the waters off central California. Marine Mammal Science 13(2):186-195.

Roberts, L., S. Cheesman, M. Elliott, and T. Breithaupt. 2016. Sensitivity of Pagurus bernhardus (L.) to substrate-borne vibration and anthropogenic noise. Journal of Experimental Marine Biology and Ecology 474:185-194.

Robinson, R. A., and coauthors. 2008. Travelling through a warming world: climate change and migratory species. Endangered Species Research.

Robinson, R. A., and coauthors. 2005. Climate change and migratory species. Defra Research, British Trust for Ornithology, Norfolk, United Kingdom.

Rochman, C. M., E. Hoh, T. Kurobe, and S. J. Teh. 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Scientific Reports 3:3263.

Rockwood, R. C., J. Adams, G. Silber, and J. Jahncke. 2020. Estimating effectiveness of speed reduction measures for decreasing whale-strike mortality in a high-risk region. Endangered Species Research 43:145-166.

Rockwood, R. C., J. D. Adams, S. Hastings, J. Morten, and J. Jahncke. 2021. Modeling Whale Deaths From Vessel Strikes to Reduce the Risk of Fatality to Endangered Whales. Frontiers in Marine Science:919.

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).

Roden, S. E., and coauthors. 2023. Population structure of Pacific green turtles: a new perspective from microsatellite DNA variation. Frontiers in Marine Science 10:1116941.

Rogers-Bennett, L., P. L. Haaker, T. O. Huff, and P. K. Dayton. 2002. Estimating baseline abundances of abalone in California for restoration. Reports of California Cooperative Oceanic Fisheries Investigations 43:97-111.

Rohner, C., and coauthors. 2013. Trends in sightings and environmental influences on a coastal aggregation of manta rays and whale sharks. Marine Ecology Progress Series 482:153-168.

Rohrkasse-Charles, S., B. Würsig, and F. Ollervides. 2011. Social context of gray whale Eschrichtius robustus sound activity. Pages 255 *in* Nineteenth Biennial Conference on the Biology of Marine Mammals, Tampa, Florida.

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.

Romanenko, E. V., and V. Y. Kitain. 1992. The functioning of the echolocation system of Tursiops truncatus during noise masking. Pages 415-419 *in* Marine mammal sensory systems. Springer.

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.

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.

Rosen, Y. 2015. More whales found dead in southern Alaska waters. Alaska Dispatch News.

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.

Rostad, A., S. Kaartvedt, T. A. Klevjer, and W. Melle. 2006. Fish are attracted to vessels. ICES Journal of Marine Science 63(8):1431–1437.

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.

Rubin, R., K. Kumli, and G. Chilcott. 2008. Dive characteristics and movement patterns of acoustic and satellite-tagged manta rays (Manta birostris) in the Revillagigedos Islands of Mexico. Joint Meeting of Ichthyologists and Herpetologists. Montreal, Canada.

Ruck, C. L. 2016. Global genetic connectivity and diversity in a shark of high conservation concern, the oceanic whitetip, *Carcharhinus longimanus*. Nova Southeastern University, Fort Lauderdale, Florida.

Ruck, C. L., M. S. Shivji, R. W. Jabado, and A. M. Bernard. 2024. Cross ocean-basin population genetic dynamics in a pelagic top predator of high conservation concern, the oceanic whitetip shark, Carcharhinus longimanus. Conservation Genetics 25(3):677-695.

Ruiz, G. M., P. Fofonoff, and A. H. Hines. 1999. Non-indigenous species as stressors in estuarine and marine communities: Assessing invasion impacts and interactions. Limnology and Oceanography 44(3):950–972.

Ruscher, B., and coauthors. 2025. Psychoacoustic data confirm reduced hearing sensitivity in Hawaiian monk seals relative to Phocinae seals. Endangered Species Research 56:19-26.

Ruscher, B., J. M. Sills, B. P. Richter, and C. Reichmuth. 2021. In-air hearing in Hawaiian monk seals: implications for understanding the auditory biology of Monachinae seals. Journal of Comparative Physiology A 207(4):561-573.

Russell, D. J., and G. H. Balazs. 2015. Increased use of non-native algae species in the diet of the green turtle (Chelonia mydas) in a primary pasture ecosystem in Hawaii. Aquatic Ecosystem Health & Management 18(3):342-346.

Ruud, J. T. 1956. The blue whale. (Balaenoptera musculus). Scientific American 195:46-50.

Ryan, J. P., and coauthors. 2022. Oceanic giants dance to atmospheric rhythms: Ephemeral wind-driven resource tracking by blue whales. Ecology Letters 25(11):2435-2447.

Saba, V. S. 2013. Oceanic habits and habitats. The biology of sea turtles 3:163.

Saez, L., D. Lawson, and M. DeAngelis. 2021. Large whale entanglements off the U.S. West Coast, from 1982-2017.

Saez, L., and coauthors. 2013. Understanding the co-occurrence of large whales and commercial fixed gear fisheries off the west coast of the United States. NOAA Technical Memorandum NMFS–SWR–044.

Sahoo, G., R. K. Sahoo, and P. Mohanty-Hejmadi. 1996. Distribution of heavy metals in the eggs and hatchlings of olive ridley sea turtles, *Lepidochelys olivacea*, from Gahirmatha, Orissa. Indian Journal of Marine Sciences 25(4):371-372.

Salas, A. K., A. M. Capuano, C. A. Harms, W. E. Piniak, and T. A. Mooney. 2023a. Temporary noise-induced underwater hearing loss in an aquatic turtle (Trachemys scripta elegans). The Journal of the Acoustical Society of America 154(2):1003-1017.

Salas, A. K., A. M. Capuano, C. A. Harms, W. E. D. Piniak, and T. A. Mooney. 2023b. Frequency-dependent temporary threshold shifts in the Eastern Painted Turtle (*Chrysemys picta picta*). In Progress.

Salimi, P. A., and coauthors. 2021. A review of the diversity and impact of invasive non-native species in tropical marine ecosystems. Marine Biodiversity Records 2021(14).

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.

Samson, J. E., T. A. Mooney, S. W. S. Gussekloo, and R. T. Hanlon. 2014. Graded behavioral responses and habituation to sound in the common cuttlefish <em>Sepia officinalis</em&gt. The Journal of Experimental Biology 217(24):4347.

Sanchez, I., C. Fernandez, and J. Arrontes. 2005. Long-Term Changes in the Structure of Intertidal Assemblages after Invasion by Sargassum Muticum (Phaeophyta)1. Journal of Phycology 41(5):942-949.

Sanford, E., J. L. Sones, M. García-Reyes, J. H. R. Goddard, and J. L. Largier. 2019. Widespread shifts in the coastal biota of northern California during the 2014–2016 marine heatwaves. Scientific Reports 9(1):1–14.

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.

Santos, B. S., and coauthors. 2024. Beyond boundaries: governance considerations for climate-driven habitat shifts of highly migratory marine species across jurisdictions. npj | ocean sustainability 2024(3).

Santos, R. G., R. Andrades, M. A. Boldrini, and A. S. Martins. 2015. Debris ingestion by juvenile marine turtles: an underestimated problem. Marine Pollution Bulletin 93(1):37-43.

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.

Sardain, A., E. Sardain, and B. Leung. 2019. Global forecasts of shipping traffic and biological invasions to 2050. Nature Sustainability 2(4):274-282.

Savage, K., and coauthors. 2017. Abstract: 2015 Gulf of Alaska Large Whale Unusual Mortality Event. Kodiak Area Marine Science Symposium, Kodiak, Alaska.

Savery, L. C., and coauthors. 2014. Global assessment of oceanic lead pollution using sperm whales (Physeter macrocephalus) as an indicator species. Marine Pollution Bulletin 79(1-2):236-244.

Savoca, M. S., and coauthors. 2022. Towards a North Pacific Ocean long-term monitoring program for plastic pollution: A review and recommendations for plastic ingestion bioindicators. Environmental Pollution:119861.

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. The 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.

Schultz, J. K. 2009. Conservation genetics of the Hawaiian monk seal. University of Hawaii, Manoa.

Schultz, J. K., J. D. Baker, R. J. Toonen, A. L. Harting, and B. W. Bowen. 2011. Range-wide genetic connectivity of the Hawaiian monk seal and implications for translocation. Conservation Biology 25(1):124-132.

Schulze-Haugen, M., and N. E. Kohler. 2003. Guide to Sharks, Tunas, & Billfishes of the U.S. Atlantic and Gulf of Mexico. RI Sea Grant and National Marine Fisheries Service.

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.

Secor, D. H., and E. J. Niklitschek. 2001. Hypoxia and sturgeons. University of Maryland Center for Environmental Science, Chesapeake Biological Laboratory, Technical Report Series No. TS-314-01-CBL, Solomons, Maryland.

Seki, T., T. Taniuchi, H. Nakano, and M. Shimizu. 1998. Age, Growth and Reproduction of the Oceanic Whitetip Shark from the Pacific Ocean. Fisheries science 64(1):14-20.

Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (Chelonia mydas) under the Engangered Species Act. C. Southwest Fisheries Science, editor. National Marine Fisheries Service, Southwest Fisheries Science Center.

Seminoff, J. A., and coauthors. 2014. Loggerhead sea turtle abundance at a foraging hotspot in the eastern Pacific Ocean: Implications for at-sea conservation. Endangered Species Research 24(3):207-220.

Senko, J., A. Mancini, J. A. Seminoff, and V. Koch. 2014. Bycatch and directed harvest drive high green turtle mortality at Baja California Sur, Mexico. Biological Conservation 169:24-30.

Sequeira, A. M., and coauthors. 2018. Convergence of marine megafauna movement patterns in coastal and open oceans. Proceedings of the National Academy of Sciences:201716137.

Shallenberger, E., M. M. Commission, U. States, and M. Corporation. 1981. The status of Hawaiian cetaceans; Final Report to the U.S. Marine Mammal Comission. U.S. Department of Commerce, National Technical Information Service MMC-77/23.

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.

Shattuck, E. F. 2011. Geographic origins of illegally harvested hawksbill sea turtle products. Michigan State University. Forensic Science.

Shaw, K. R., and coauthors. 2023. Green sea turtles (Chelonia mydas) accumulate heavy metals near a former skeet shooting range in Kailua, O'ahu, Hawai'i. Environmental Toxicology and Chemistry 42(5):1109-1123.

Shen, Y., N. E. Hussey, M. David, F. Wu, and Y. Li. 2024. Vertebral microchemistry as an indicator of habitat use of the oceanic whitetip shark Carcharhinus longimanus in the central and eastern Pacific Ocean. Journal of Fish Biology 104(6):1732-1742.

Shillinger, G. L., and coauthors. 2011. Vertical and horizontal preferences of post-nesting leatherback turtles in the South Pacific Ocean. Marine Ecology Progress Series 422:275-289.

Shivji, M., D. Parker, B. Hartwick, M. Smith, and N. Sloan. 1983. Feeding and distribution study of the sunflower sea star *Pycnopodia helianthoides* (Brandt, 1835).

Siegel, J. E., and L. G. Crozier. 2020. Impacts of Climate Change on Salmon of the Pacific Northwest: A review of the scientific literature published in 2019. U.S. National Marine Fisheries Service, Northwest Region.

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 4:413.

Silber, G. K., D. W. Weller, R. R. Reeves, J. D. Adams, and T. J. Moore. 2021. Co-occurrence of gray whales and vessel traffic in the North Pacific Ocean. Endangered Species Research 44:177-201.

Sills, J. M., and coauthors. 2021. Underwater hearing and communication in the endangered Hawaiian monk seal Neomonachus schauinslandi. Endangered Species Research 44:61-78.

Silve, L. D., P. Wensveen, F. Visser, and C. Cure. 2016. Naval sonar disrupts foraging in humpback whales. Marine Ecology Progress Series 562:211-220.

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.

Simeone, C. A., F. M. Gulland, T. Norris, and T. K. Rowles. 2015. A systematic review of changes in marine mammal health in North America, 1972-2012: the need for a novel integrated approach. PLoS ONE 10(11):e0142105.

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.

Simonis, A. E., and coauthors. 2020. Seal bomb noise as a potential threat to Monterey Bay harbor porpoise. Frontiers in Marine Science 7:142.

Širović, A., and coauthors. 2020. Chapter 20: Trends in inputs of anthropogenic noise into the marine environment. United Nations, New York, NY.

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.

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. .

Slabbekoorn, H., and coauthors. 2019. Population-level consequences of seismic surveys on fishes: An interdisciplinary challenge. Fish and Fisheries 20(4):653-685.

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, 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., and coauthors. 2018. A decade and a half of Pseudo-nitzschia spp. and domoic acid along the coast of southern California. Harmful Algae 79:87-104.

Smith, J. E., C. L. Hunter, and C. M. Smith. 2002. Distribution and reproductive characteristics of nonindigenous and invasive marine algae in the Hawaiian Islands. Pacific Science 56(3):299-315.

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, 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.

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.

Smultea, M. A., F. C. Robertson, and D. Fertl. 2022. Blue Whale (Balaenoptera musculus) Mother–Calf Pair Behavioral Response to Vessel in the Southern California Bight. Aquatic Mammals 48(6):690-692.

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. 2000. Field measurements of sonic boom penetration into the ocean. Journal of the Acoustical Society of America 107(6):3073-3083.

Solé, M., and coauthors. 2023. Marine invertebrates and noise. Frontiers in Marine Science 10:1129057.

Sole, M., and coauthors. 2016. Evidence of Cnidarians sensitivity to sound after exposure to low frequency noise underwater sources. Sci Rep 6:37979.

Sørensen, P. M., and coauthors. 2023. Anthropogenic noise impairs cooperation in bottlenose dolphins. Current Biology 33:1-6.

Sosa-Cornejo, I., and coauthors. 2021. Nesting trends of olive ridley sea turtles Lepidochelys olivacea (Testudinata: Cheloniidae) on two beaches in Northwestern Mexico after 30 and 40 years of conservation. Revista de Biología Tropical 69(3):1124-1137.

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. 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. L., and coauthors. 2023. Behavioural responses of fin whales to military mid-frequency active sonar. Royal Society Open Science 10(12):231775.

Southall, B. L., and coauthors. 2007. 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(5):jeb190637.

Southall, B. L., and coauthors. 2019b. Marine mammal noise exposure criteria: Updated scientific recommendations for residual hearing effects. Aquatic Mammals 45(2):125-232.

Southall, B. L., and coauthors. 2021. Marine mammal noise exposure criteria: assessing the severity of marine mammal behavioral responses to human noise. Aquatic Mammals 47(5):421-464.

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. .

Spargo, B. J. 1999. Environmental effects of RF chaff. Undersecretary of Defense for Environmental Security, Naval Research Laboratory, Washington, D.C.

Sparrow, V. W. 2002. Review and status of sonic boom penetration into the ocean. Journal of the Acoustical Society of America 111(1):537-543.

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.

Spotila, J. R. 2004. Sea turtles: A complete guide to their biology, behavior, and conservation. John Hopkins University Press, Baltimore. 227p.

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.

Stabeno, P. J., and coauthors. 2004. Meteorology and oceanography of the northern Gulf of Alaska. Continental Shelf Research 24-Jan(8-Jul):859-897.

Stachura, M. M., N. J. Mantua, and M. D. Scheuerell. 2013. Oceanographic influences on patterns in North Pacific salmon abundance. Canadian Journal of Fisheries and Aquatic Sciences 71(2):226-235.

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., 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.

Stanistreet, J. E., and coauthors. 2022. Changes in the acoustic activity of beaked whales and sperm whales recorded during a naval training exercise off eastern Canada. Scientific Reports 12(1):1973.

Starcovic, S., and J. Mintz. 2021. Characterization of Vessel Traffic within Navy Training and Testing Areas. Dept. of The Navy, Commander, Pacific Fleet, Environmental Readiness Division (COMPACFLT N465).

Stevens, J., and J. Lyle. 1989. Biology of three hammerhead sharks (*Eusphyra blochii*, *Sphyrna mokarran* and *S. lewini*) from northern Australia. Marine and Freshwater Research 40(2):129-146.

Stewart, J. D., and coauthors. 2016. Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. Biological Conservation 200:178-183.

Stewart, J. D., M. Nuttall, E. L. Hickerson, and M. A. Johnston. 2018. Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. Marine Biology 165(7):111.

Stierhoff, K. L., S. A. Mau, and D. W. Murfin. 2015. White abalone populations at Tanner Bank and Cortes Bank estimated from an optical survey. Unpublished paper.

Stierhoff, K. L., M. Neuman, and J. L. Butler. 2012. On the road to extinction? Population declines of the endangered white abalone, Haliotis sorenseni. Biological Conservation 152:46-52.

Stierhoff, K. L., M. Neumann, S. A. Mau, and D. W. Murfin. 2014. White abalone at San Clemente Island: population estimates and management recommendations. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

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.

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, and G. O. Marcotrigiano. 2007. Polychlorinated biphenyls and other chlorinated organic contaminants in the tissues of Mediterranean loggerhead turtle *Caretta caretta*. Science of The Total Environment 373(2-3):456-463.

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.

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.

Stringell, T. B., and coauthors. 2013. Marine turtle harvest in a mixed small-scale fishery: evidence for revised management measures. Ocean & coastal management 82:34-42.

Summers, D. J. 2016. Algal toxins found in Alaska marine mammals for first time. Alaska Journal of Commerce.

Sutherland, B. J., and coauthors. 2021. Population structure of eulachon (Thaleichthys pacificus) from Northern California to Alaska using single nucleotide polymorphisms from direct amplicon sequencing. Canadian Journal of Fisheries and Aquatic Sciences 78(1):78-89.

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.

SWFSC. 2023. Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest.

Swope, B., and J. McDonald. 2013. Copper-Based Torpedo Guidance Wire: Applications and Environmental Considerations. San Diego, CA.

Sydeman, W. J., and coauthors. 2014. Multivariate ocean-climate indicators (MOCI) for the central California Current: Environmental change, 1990–2010. Progress in Oceanography 120:352-369.

Szesciorka, A. R., and coauthors. 2019. A case study of a near vessel strike of a blue whale: perceptual cues and fine-scale aspects of behavioral avoidance. Frontiers in Marine Science 6:761.

Szesciorka, A. R., and coauthors. 2020. Timing is everything: Drivers of interannual variability in blue whale migration. Scientific Reports 10(1):1-9.

Szuwalski, C. S., K. Aydin, E. J. Fedewa, B. Garber-Yonts, and M. A. Litzow. 2023. The collapse of eastern Bering Sea snow crab. Science 382(6668):306–310.

Tackaberry, J., E. Dobson, K. Flynn, T. Cheeseman, and J. Calambokidis. 2022. Low resighting rate of entangled humpback whales within the California, Oregon, and Washington region based on photo-identification and long-term life history data. Frontiers in Marine Science 8:2092.

Taguchi, M., and coauthors. 2021. New Insights into the Genetic Structure of Sei Whales (*Balaenoptera Borealis*) at the Inter-Oceanic Scale. Cetacean Population Studies 3:152-163.

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.

Tambourgi, M., and coauthors. 2013. Reproductive aspects of the oceanic whitetip shark, *Carcharhinus longimanus* (Elasmobranchii: Carcharhinidae), in the equatorial and southwestern Atlantic Ocean. . Brazilian Journal of Oceanography 61:161-168.

Tamura, T., H. Yoshida, G. Yasunaga, M. Goto, and L. A. Pastene. 2019. Final conclusions of the JARPNII research based on refined analyses and additonal samples. International Whaling Commission.

Tapilatu, R. F., and coauthors. 2013. Long-term decline of the western Pacific leatherback, Dermochelys coriacea: a globally important sea turtle population. Ecosphere 4(2):1-15.

Tapilatu, R. F., and M. Tiwari. 2007. Leatherback turtle, Dermochelys coriacea, hatching success at Jamursba-Medi and Wermon beaches in Papua, Indonesia. Chelonian Conservation and Biology 6(1):154-158.

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.

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.

Thode, A., and coauthors. 2016. Using line acceleration to measure false killer whale (Pseudorca crassidens) click and whistle source levels during pelagic longline depredation. The Journal of the Acoustical Society of America 140(5):3941-3951.

Thomas, J., M. Stoermer, C. Bower, L. Anderson, and A. Garver. 1988. Detection abilities and signal characteristics of echolocating false killer whale (*Pseudorca crassidens*). Pages 321-334 *in* P. N. a. P. Moore, editor. Animal Sonar: Processes and Performance, Plenum, New York.

Thomas, J. A., and C. W. Turl. 1990. Echolocation characteristics and range detection threshold of a false killer whale (Pseudorca crassidens). Pages 321-334 *in* Sensory abilities of cetaceans: laboratory and field evidence. Springer.

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. 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. 1992. 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, T. J., H. E. Winn, and P. J. Perkins. 1979. Mysticete sounds. Pages 403-431 *in* H. E. Winn, and B. L. Olla, editors. Behavior of Marine Animals: Current Perspectives in Research Vol. 3: Cetaceans. Plenum Press, New York, NY.

Tierra Data, I. T. 2008. Naval Auxiliary Landing Field, San Clemente Island Black Abalone (Haliotis cracherodii) Survey. Prepared for NAVFAC SW Naval Station, San Diego, CA.

Tierra Data, I. T. 2013. Integrated Natural Resources Management Plan: Naval Auxiliary Landing Field, San Clemente Island, California. Prepared for: Naval Base Coronado, Natural Resources Office, San Diego, CA.

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).

Todd, V. L., and coauthors. 2014. A review of impacts of marine dredging activities on marine mammals. ICES Journal of Marine Science 72(2):328-340.

Toha, A. H. A., and coauthors. 2025. Population structure and genetic diversity of leatherback turtles (Dermochelys coriacea) in Bird's Head Seascape, Papua-Indonesia. Frontiers in Marine Science 12:1569466.

Tolotti, M. T., P. Bach, F. Hazin, P. Travassos, and L. Dagorn. 2015. Vulnerability of the Oceanic Whitetip Shark to Pelagic Longline Fisheries. PLoS ONE PLoS ONE 10(10).

Tremblay-Boyer, L., F. Carvalho, P. Neubauer, and G. Pilling. 2019. Stock assessment for oceanic whitetip shark in the Western and Central Pacific Ocean. Report to the WCPFC Scientific Committee. Fifteenth Regular Session.

Tremblay, C. J., S. M. Van Parijs, and D. Cholewiak. 2019. 50 to 30-Hz triplet and singlet down sweep vocalizations produced by sei whales (*Balaenoptera borealis*) in the western North Atlantic Ocean. Journal of the Acoustic Society of America 145(6):3351-3358.

Tricas, T., and J. New. 1997. Sensitivity and response dynamics of elasmobranch electrosensory primary afferent neurons to near threshold fields. Journal of Comparative Physiology A 182:89-101.

Tutschulte, T. C. 1976. The comparative ecology of three sympatric abalones. Doctoral dissertation. University of California, San Diego.

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. Reynolds III, and S. A. Rommel, editors. Biology of Marine Mammals. Smithsonian Institution Press, Washington.

Tyack, P. L., and C. W. Clark. 2000. Communication and acoustic behavior of dolphins and whales. Pages 156-224 *in* W. W. L. A. A. N. P. R. R. Fay, editor. Hearing by Whales and Dolphins. Springer-Verlag, New York.

Tyack, P. L., and coauthors. 2011b. Beaked whales respond to simulated and actual Navy sonar. PLoS ONE 6(3):e17009.

U.S. Department of the Air Force. 2016. United States Air Force F-35A Operational Beddown–Pacific Final Environmental Impact Statement.

U.S. Department of the Army. 1999. Finding of No Significant Impact (FONSI) for the Life Cycle Environmental Assessment (LCEA) for the HELLFIRE Modular Missile System.

U.S. Department of the Navy. 2001. *Airborne Mine Neutralization System (AMNS) Inert Target Tests: Environmental Assessment and Overseas Environmental Assessment*. Coastal Systems Station, Panama City, FL.

U.S. Department of the Navy. 2012. Biological Assessment for the Expeditionary Electronic Attack Squadron Realignment and Transition at Naval Air Station Whidbey Island, Oak Harbor, Washington. Pages 1–56 *in*. U.S. Department of the Navy, Washington, DC.

U.S. Department of the Navy. 2013. 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). N. M. F. S. Office of Protected Resources, National Oceanographic and Atmospheric Administration, editor, Point Mugu, CA.

U.S. Department of the Navy. 2017. Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III). Space and Naval Warfare System Command, Pacific.

U.S. Naval Research Advisory Committee. 2009. Report on Jet Engine Noise Reduction. U.S. Department of Defense, Patuxent River, MD.

U.S. Navy. 2022. Point Mugu Sea Range Final Environmental Impact Statement/Overseas Environmental Impact Statement January 2022 (Volumes 1 & 2). U.S. Department of the Navy, Pt. Mugu, California.

Upton, H. F. 2020. US offshore aquaculture regulation and development. Congressional Research Service.

Urbán R.J., and coauthors. 2019. New information on the gray whale migratory movements between the western and eastern North Pacific. Paper SC/68A/CMP/11 Rev1 presented to the International Whaling Commission Scientific Committee.

Urick, R. J. 1983. Principles of Underwater Sound, 3rd edition. Peninsula Publishing, Los Altos, California.

USAF, U. S. A. F. 2000. Supersonic Aircraft Noise at and Beneath the Ocean Surface: Estimation of Risk for Effects on Marine Mammals. .

USCOP. 2004. An ocean blueprint for the 21st century. Final report. U.S. Commission on Ocean Policy, Washington, D. C.

USFWS. 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.

USFWS. 2012. Hawksbill Sea Turtle Fact Sheet.

USFWS, N. 2020. Loggerhead Sea Turtle (*Caretta caretta*) North Pacific Ocean DPS 5-Year Review: Summary and Evaluation 2020. National Marine Fisheries Service Office of Protected Resources Silver Spring, Maryland and U.S. Fish and Wildlife Service Southeast Region North Florida Ecological Services Office Jacksonville, Florida.

USMC. 2017. Integrated Natural Resources Management Plan (INRMP) Update (2017 – 2021), MCBH. MCBH, editor. Environmental Compliance and Protection Department Marine Corps Base Hawaii, Kaneohe, Oahu HI

Valverde, R. A., and coauthors. 2012. Olive ridley mass nesting ecology and egg harvest at Ostional Beach, Costa Rica. Chelonian Conservation and Biology 11(1):1-11.

Van Bressem, M.-F., K. Van Waerebeek, and J. A. Raga. 1999. A review of virus infections of cetaceans and the potential impact of morbilliviruses, poxviruses and papillomaviruses on host population dynamics. Diseases of aquatic organisms 38(1):53-65.

Van de Merwe, J. P. V., and coauthors. 2009. Chemical contamination of green turtle (*Chelonia mydas*) eggs in peninsular Malaysia: Implications for conservation and public health. Environmental Health Perspectives 117(9):1397-1401.

Van Dolah, F. M. 2005. Effects of harmful algal blooms. Marine mammal research: Conservation beyond crisis:85-99.

van Weelden, C., J. R. Towers, and T. Bosker. 2021. Impacts of climate change on cetacean distribution, habitat and migration. Climate Change Ecology 1:100009.

VanBlaricom, G., and coauthors. 2009. Status Review Report for Black Abalone (*Haliotis cracherodii* Leach, 1814). NMFS Southwest Region, Long Beach, California.

VanBlaricom, G. R., M. J. Neuman, and D. A. Witting. 2015. Indications of recovery trends in three species of abalone in California, with apparent linkages to ocean-scale temperature anamolies. Pages 686-687 *in* Journal of Shellfish Research. NATL SHELLFISHERIES ASSOC C/O DR. SANDRA E. SHUMWAY, UNIV CONNECTICUT, 1080 SHENNECOSSETT RD, GROTON, CT 06340 USA.

Vandeperre, F., and coauthors. 2019. Relative abundance of oceanic juvenile loggerhead sea turtles in relation to nest production at source rookeries: implications for recruitment dynamics. Scientific Reports 9(1):13019.

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.

Venables, S. 2013. Short term behavioural responses of manta rays, Manta alfredi, to tourism interactions in Coral Bay, Western Australia. Murdoch University.

Vereide, E. H., and coauthors. 2023. Effects of airgun discharges used in seismic surveys on development and mortality in nauplii of the copepod Acartia tonsa. Environmental Pollution 327:121469.

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, 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).

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. M., and coauthors. 2021. Modeling potential masking of echolocating sperm whales exposed to continuous 1–2 kHz naval sonar. The Journal of the Acoustical Society of America 149:2908–2925.

Wada, S., and K.-I. Numachi. 1991. Allozyme analyses of genetic differentiation among the populations and species of the Balaenoptora. International Whaling Commission.

Wade, A. A., and coauthors. 2013. Steelhead vulnerability to climate change in the Pacific Northwest. Journal of Applied Ecology 50(5):1093-1104.

Wade, P. R., and coauthors. 2011. The world's smallest whale population? Biology Letters 7(1):83-85.

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.

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. 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.

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. 2010. Global patterns of marine turtle bycatch. Conservation letters 3(3):131-142.

Wallentinus, I., and C. D. Nyberg. 2007. Introduced marine organisms as habitat modifiers. Marine Pollution Bulletin 55(7-9):323-332.

Walsh, W., and J. Brodziak. 2022. CPUE Standardization for Oceanic Whitetip Shark, Carcharhinus longimanus, in the Hawaiian Longline Fishery during 1994-2019. Pacific Islands Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration Honolulu, Hawaii 96818.

Wang, G., and coauthors. 2017. The peak structure and future changes of the relationships between extreme precipitation and temperature. Nature Climate Change 7(4):268-274.

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.

Warlick, A. J., and coauthors. 2022. Cetacean strandings in the US Pacific northwest 2000–2019 reveal potential linkages to oceanographic variability. Frontiers in Marine Science 9:162.

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.

Watkins, W. A. 1977. Acoustic behavior of sperm whales. Oceanus 20:50-58.

Watkins, W. A. 1981a. Activities and underwater sounds of fin whales. Scientific Reports of the Whales Research Institute 33:83-117.

Watkins, W. A. 1981b. Activities and underwater sounds of fin whales (*Balaenoptera physalus*). Scientific Reports of the Whales Research Institute Tokyo 33:83-117 & 118.

Watkins, W. A. 1981c. Radio tagging of finback whales - Iceland, June-July 1980. Woods Hole Oceanagraphic Institution.

Watkins, W. A. 1985. Changes observed in the reaction of whales to human activities. National Marine Fisheries Service.

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., K. E. Moore, D. Wartzok, and J. H. Johnson. 1981. Radio tracking of finback (Balaenoptera physalus), and humpback (Megaptera novaeangliae) whales in Prince William Sound, Alaska, USA. Deep Sea Research Part A. Oceanographic Research Papers 28(6):577-588.

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.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. 2005. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Canadian Journal of Fisheries and Aquatic Sciences 62(5):965-981.

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, and ODFW. 2001. Washington and Oregon eulachon management plan. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

Weilgart, L. 2013. A review of the impacts of seismic airgun surveys on marine life. Pages 1-10 *in* CBD Expert Workshop on Underwater Noise and its Impacts on Marine and Coastal Biodiversity. London, United Kingdom.

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 20 Hz pulses measured in the Northeast Pacific Ocean. Journal of the Acoustical Society of America 133(2):741–749.

Welch, H., and coauthors. 2019. Environmental indicators to reduce loggerhead turtle bycatch offshore of Southern California. Ecological Indicators 98:657-664.

Weller, D. W., A. L. Bradford, A. R. Lang, R. L. Brownell Jr., and A. M. Burdin. 2009. Birth-Intervals and Sex Composition of Western Gray Whales Summer.

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. 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.

Wensveen, P. J., and coauthors. 2025. Distance matters to sperm whales: Behavioural disturbance in response to both sonar received level and source distance. Marine Pollution Bulletin 214:117742.

Western Australian Department of Industry and Resources. 2002. Petroleum Information Series - Guidelines Sheet 1. Guidelines on minimising acoustic disturbance to marine fauna. Available online at: http://www.doir.wa.gov.au/documents/mineralsandpetroleum/envsheetmar02.pdf Accessed on: 10/19/2006.

WGWAP. 2019. Open statement of concern from the Western Gray Whale Advisory Panel on the decline of the amphipod prey base in the nearshore feeding area of gray whales near Piltun Lagoon, Sakhalin Island, Russia, Far East.

White, E. R., M. C. Myers, J. M. Flemming, and J. K. Baum. 2015. Shifting elasmobranch community assemblage at Cocos Island—an isolated marine protected area. Conservation Biology 29(4):1186-1197.

Whitehead, H. 1997. Sea surface temperature and the abundance of sperm whale calves off the Galapagos Islands: Implications for the effects of global warming. Report of the International Whaling Commission 47:941-944.-Sc/48/O30).

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. 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., A. Coakes, N. Jaquet, and S. Lusseau. 2008. Movements of sperm whales in the tropical Pacific. Marine Ecology Progress Series 361:291-300.

Whitehead, H., and M. Shin. 2022. Current global population size, post-whaling trend and historical trajectory of sperm whales. Scientific Reports 12(1):19468.

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.

Wigand, C., J. C. Stevenson, and J. C. Cornwell. 1997. Effects of different submersed macrophytes on sediment biogeochemistry. Aquatic Botany 56:233-244.

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., 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. 2018. Summary of Five Years of Ambient and Anthropogenic Sound in the SOCAL Range Complex 2012 - 2017 (Marine Physical Laboratory Technical Memorandum). Scripps Institution of Oceanography, 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.

Wiles, G. 2017. Periodic status review for the blue, fin, sei, North Pacific right, and sperm whales in Washington. Washington Department of Fish and Wildlife, Olympia, Washington 46.

Wilkinson, C., and D. Souter. 2008. Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre, Townsville.

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., E. Ashe, and P. D. O'hara. 2011. 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. 2002. Behavioural responses of male killer whales to a leapfrogging vessel. Journal of Cetacean Research and Management 4(3):305-310.

Williams, R., C. Erbe, E. Ashe, A. Beerman, and J. Smith. 2014. Severity of killer whale behavioral responses to ship noise: A dose-response study. Marine Pollution Bulletin 79(1-2):254-260.

Williams, S. E., L. P. Shoo, J. L. Isaac, A. A. Hoffmann, and G. Langham. 2008. Towards an integrated framework for assessing the vulnerability of species to climate change. PLoS Biol 6(12):e325.

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.

Wiri, S., and coauthors. 2023. Dynamic monitoring of service members to quantify blast exposure levels during combat training using BlackBox Biometrics Blast Gauges: explosive breaching, shoulder-fired weapons, artillery, mortars, and 0.50 caliber guns. Frontiers in neurology 14:1175671.

Wisdom, S., A. Bowles, and J. Sumich. 1999. Development of sound production in gray whales, Eschrichtius robustus. Pages 203-204 *in* Thirteenth Biennial Conference on the Biology of Marine Mammals, Wailea, Maui, Hawaii.

Wisdom, S., A. E. Bowles, and K. E. Anderson. 2001. Development of behavior and sound repertoire of a rehabilitating gray whale calf. (Eschrichtius robustus). Aquatic Mammals 27(3):239-255.

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.

Wood, C., and coauthors. 2021. Sea turtles across the North Pacific are exposed to perfluoroalkyl substances. Environmental Pollution 279:116875.

Work, P. A., A. L. Sapp, D. W. Scott, and M. G. Dodd. 2010. Influence of small vessel operation and propulsion system on loggerhead sea turtle injuries. Journal of Experimental Marine Biology and Ecology 393(1-2):168-175.

Work, T. M., and G. H. Balazs. 2013. Tumors in sea turtles: The insidious menace of fibropapillomatosis. The Wildlife Professional:44–47.

Worm, B., and coauthors. 2024. Global shark fishing mortality still rising despite widespread regulatory change. Science 383(6679):225-230.

Wuebbles, D. J., and coauthors. 2017. Executive Summary. Pages 12-34 *in* D. J. Wubbles, and coeditors, editors. Climate Science Special Report: Fourth National Climate Assessment, volume 1. U.S. Global Change Research Program, Washington D.C.

Wursig, B., S. K. Lynn, T. A. Jeffereson, and K. D. Mullin. 1998. Behaviour of cetaceans in the northen 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.

Yagla, J., and R. Stiegler. 2003. Gun blast noise transmission across the air-sea interface. euronoise, Naples, FL.

Yan, M., and coauthors. 2018. The risk of polychlorinated biphenyls facilitating tumors in Hawaiian green sea turtles (Chelonia mydas). International Journal of Environmental Research and Public Health 15(6):1243.

Yang, W.-C., and coauthors. 2021. Anthropogenic sound exposure-induced stress in captive dolphins and implications for cetacean health. Frontiers in Marine Science 8.

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., D. R. Richmond, E. R. Fletcher, and R. K. Jones. 1973. Safe distances from underwater explosions for mammals and birds. Defense Nuclear Agency.

Ylitalo, G. M., and coauthors. 2008. Organochlorine contaminants in endangered Hawaiian monk seals from four subpopulations in the Northwestern Hawaiian Islands. Marine Pollution Bulletin 56(2):231-244.

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.

Young, C. N., and coauthors. 2017. Status review report: oceanic whitetip shark (*Carcharhinius longimanus*). Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Silver Spring, Maryland.

Young, C. N., and J. K. Carlson. 2020. The biology and conservation status of the oceanic whitetip shark (Carcharhinus longimanus) and future directions for recovery. Reviews in Fish Biology and Fisheries 30(2):293-312.

Young, N. C., and coauthors. 2024. Alaska marine mammal stock assessments, 2023. Pages 327 *in* NMFS, editor NOAA Technical Memorandum NMFS-AFSC-493. U.S. Department of Commerce.

Yuen, M. M. L., P. E. Nachtigall, M. Breese, and S. A. Vlachos. 2007. The perception of complex tones by a false killer whale (Pseudorca crassidens). Journal of the Acoustical Society of America 121(3):1768-1774.

Zedonis, P. A. 1992. The biology of steelhead (*Onchorynchus mykiss*) in the Mattole River estuary/lagoon, California. Humboldt State University, Arcata, CA.

Zhang, X. 2022. Predicting global seasonal distributions and population exchange routes of a Critically Endangered shark. Biological Conservation 275:109771.

Zickel MJ, and coauthors. 2021. Synopsis of Marine Environmental Conditions off Southern California from May - July, 2009 - 2021 and Potential Correlation with Occurrence and Distribution of Large Whales. Prepared for: U.S. Navy, U.S. Pacific Fleet, Pearl Harbor, HI, Prepared by: ManTech SRS, Solana Beach, CA.

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.

Zimmerman, B. 1983. Hawaii- Long log. Hawaii Fishing News 8(3):25.

ZoBell, V. M., and coauthors. 2025. Climatic and economic fluctuations revealed by decadal ocean soundscapes. J. Acoust. Soc. Am. 157:4233-4251.

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.