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## **3.8 Marine Mammals**



**Gulf of Alaska Navy Training Activities**  
**Draft Supplemental Environmental Impact Statement/**  
**Overseas Environmental Impact Statement**

**TABLE OF CONTENTS**

<b>3.8</b>	<b>Marine Mammals.....</b>	<b>3.8-1</b>
3.8.1	Introduction .....	3.8-1
3.8.2	Affected Environment.....	3.8-1
3.8.2.1	General Background .....	3.8-4
3.8.2.2	North Pacific Right Whale ( <i>Eubalaena japonica</i> ).....	3.8-25
3.8.2.3	Humpback Whale ( <i>Megaptera novaeangliae</i> ) .....	3.8-26
3.8.2.4	Blue Whale ( <i>Balaenoptera musculus</i> ).....	3.8-30
3.8.2.5	Fin Whale ( <i>Balaenoptera physalus</i> ) .....	3.8-32
3.8.2.6	Sei Whale ( <i>Balaenoptera borealis</i> ) .....	3.8-32
3.8.2.7	Minke Whale ( <i>Balaenoptera acutorostrata</i> ) .....	3.8-33
3.8.2.8	Gray Whale ( <i>Eschrichtius robustus</i> ) .....	3.8-34
3.8.2.9	Sperm Whale ( <i>Physeter macrocephalus</i> ).....	3.8-35
3.8.2.10	Killer Whale ( <i>Orcinus orca</i> ) .....	3.8-36
3.8.2.11	Pacific White-Sided Dolphin ( <i>Lagenorhynchus obliquidens</i> ) .....	3.8-36
3.8.2.12	Harbor Porpoise ( <i>Phocoena phocoena</i> ).....	3.8-37
3.8.2.13	Dall's Porpoise ( <i>Phocoenoides dalli</i> ) .....	3.8-37
3.8.2.14	Cuvier's Beaked Whale ( <i>Ziphius cavirostris</i> ) .....	3.8-38
3.8.2.15	Baird's Beaked Whale ( <i>Berardius bairdii</i> ) .....	3.8-38
3.8.2.16	Stejneger's Beaked Whale ( <i>Mesoplodon stejnegeri</i> ).....	3.8-39
3.8.2.17	Steller sea lion ( <i>Eumetopias jubatus</i> ) .....	3.8-39
3.8.2.18	California Sea Lion ( <i>Zalophus californianus</i> ).....	3.8-40
3.8.2.19	Northern Fur Seal ( <i>Callorhinus ursinus</i> ).....	3.8-42
3.8.2.20	Northern Elephant Seal ( <i>Mirounga angustirostris</i> ) .....	3.8-43
3.8.2.21	Harbor Seal ( <i>Phoca vitulina</i> ) .....	3.8-44
3.8.2.22	Ribbon Seal ( <i>Histiophoca fasciata</i> ).....	3.8-45
3.8.2.23	Northern Sea Otter ( <i>Enhydra lutris neris</i> ).....	3.8-46
3.8.3	Environmental Consequences .....	3.8-47
3.8.3.1	Acoustic Stressors .....	3.8-50
3.8.3.2	Explosive Stressors.....	3.8-156
3.8.3.3	Secondary Stressors.....	3.8-192

3.8.4	Summary of Stressor Assessment (Combined Impacts of All Stressors) on Marine Mammals.....	3.8-193
3.8.4.1	Summary of Monitoring and Observations During Navy Activities Since 2016.....	3.8-195
3.8.5	Endangered Species Act Determinations.....	3.8-196
3.8.6	Marine Mammal Protection Act Determinations .....	3.8-197
3.8.6.1	Summary of Science in the Temporary Maritime Activities Area by the Navy Related to Potential Effects on Marine Mammals Since 2006.....	3.8-197

## List of Tables

Table 3.8-1:	Marine Mammals with Possible or Confirmed Presence Within the TMAA.....	3.8-4
Table 3.8-2:	Species Within Marine Mammal Hearing Groups Likely Found in the TMAA .....	3.8-12
Table 3.8-3:	Cutoff Distances for Moderate Source Level, Single Platform Training Events and for All Other Events with Multiple Platforms or Sonar with Source Levels at or Exceeding 215 dB re 1 $\mu$ Pa at 1 m .....	3.8-121
Table 3.8-4:	Range to Permanent Threshold Shift for Three Representative Sonar Systems .....	3.8-125
Table 3.8-5:	Ranges to Temporary Threshold Shift for Sonar Bin MF1 over a Representative Range of Environments Within the Study Area .....	3.8-126
Table 3.8-6:	Ranges to Temporary Threshold Shift for Sonar Bin MF4 over a Representative Range of Environments Within the Study Area .....	3.8-126
Table 3.8-7:	Ranges to Temporary Threshold Shift for Sonar Bin MF5 over a Representative Range of Environments Within the Study Area .....	3.8-127
Table 3.8-8:	Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF1 over a Representative Range of Environments Within the Study Area.....	3.8-128
Table 3.8-9:	Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF4 over a Representative Range of Environments Within the Study Area .....	3.8-129
Table 3.8-10:	Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF5 over a Representative Range of Environments Within the Study Area.....	3.8-130
Table 3.8-11:	Estimated Impacts on Individual North Pacific Right Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1.....	3.8-135
Table 3.8-12:	Estimated Impacts on Individual Humpback Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1....	3.8-136
Table 3.8-13:	Estimated Impacts on Individual Blue Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-137
Table 3.8-14:	Estimated Impacts on Individual Fin Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-138

Table 3.8-15: Estimated Impacts on Individual Sei Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-138
Table 3.8-16: Estimated Impacts on Individual Minke Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-139
Table 3.8-17: Estimated Impacts on Individual Sperm Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-143
Table 3.8-18: Estimated Impacts on Individual Killer Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-144
Table 3.8-19: Estimated Impacts on Individual Pacific White-Sided Dolphin Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-144
Table 3.8-20: Estimated Impacts on Individual Dall’s Porpoise Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 ....	3.8-146
Table 3.8-21: Estimated Impacts on Individual Baird’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 ....	3.8-147
Table 3.8-22: Estimated Impacts on Individual Cuvier’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 ....	3.8-147
Table 3.8-23: Estimated Impacts on Individual Stejneger’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 .....	3.8-148
Table 3.8-24: Estimated Impacts on Individual Northern Fur Seal Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 ....	3.8-151
Table 3.8-25: Estimated Impacts on Individual Northern Elephant Seal Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1 ....	3.8-151
Table 3.8-26: Criteria to Quantitatively Assess Non-Auditory Injury Due to Explosions in Water .....	3.8-164
Table 3.8-27: Navy Phase III Weighted Sound Exposure Thresholds for Underwater Explosive Sounds .....	3.8-167
Table 3.8-28: Ranges to Non-Auditory Injury (in meters) for All Marine Mammal Hearing Groups ..	3.8-169
Table 3.8-29: Ranges to Mortality (in meters) for All Marine Mammal Hearing Groups as a Function of Animal Mass .....	3.8-169
Table 3.8-30: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for High-Frequency Cetaceans .....	3.8-170
Table 3.8-31: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for High-Frequency Cetaceans .....	3.8-170
Table 3.8-32: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Low-Frequency Cetaceans .....	3.8-171
Table 3.8-33: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Low-Frequency Cetaceans .....	3.8-171
Table 3.8-34: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Mid-Frequency Cetaceans .....	3.8-172

Table 3.8-35: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Mid-Frequency Cetaceans .....	3.8-172
Table 3.8-36: SEL Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Otariids and Mustelids .....	3.8-173
Table 3.8-37: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Otariids and Mustelids.....	3.8-173
Table 3.8-38: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Phocids <sup>1</sup> .....	3.8-174
Table 3.8-39: Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Phocids <sup>1</sup> ....	3.8-174
Table 3.8-40: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Phocids (Elephant Seals) <sup>1</sup> .....	3.8-175
Table 3.8-41: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Phocids (Elephant Seals) <sup>1</sup> .....	3.8-175
Table 3.8-42: Estimated Impacts on Individual North Pacific Right Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-178
Table 3.8-43: Estimated Impacts on Individual Humpback Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1.....	3.8-180
Table 3.8-44: Estimated Impacts on Individual Blue Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-181
Table 3.8-45: Estimated Impacts on Individual Fin Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-182
Table 3.8-46: Estimated Impacts on Individual Sei Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-182
Table 3.8-47: Estimated Impacts on Individual Minke Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-183
Table 3.8-48: Estimated Impacts on Individual Dall’s Porpoise Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1 .....	3.8-187
Table 3.8-49: Estimated Impacts on Individual Cuvier’s Beaked Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1.....	3.8-188
Table 3.8-50: Estimated Impacts on Individual Northern Elephant Seal Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1.....	3.8-190

## List of Figures

Figure 3.8-1: Composite Audiograms for Hearing Groups Likely Found in the TMAA .....	3.8-13
Figure 3.8-2: Proposed Humpback Whale Critical Habitat Overlapping the TMAA .....	3.8-28
Figure 3.8-3: Two Hypothetical Threshold Shifts.....	3.8-56
Figure 3.8-4: Odontocete Critical Ratios .....	3.8-66
Figure 3.8-5: Critical Ratios for Different Noise Types.....	3.8-67
Figure 3.8-6: Navy Auditory Weighting Functions for All Species Groups.....	3.8-115

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Figure 3.8-7: TTS and PTS Exposure Functions for Sonar and Other Transducers.....	3.8-116
Figure 3.8-8: Behavioral Response Function for Odontocetes .....	3.8-118
Figure 3.8-9: Behavioral Response Function for Pinnipeds .....	3.8-119
Figure 3.8-10: Behavioral Response Function for Mysticetes .....	3.8-119
Figure 3.8-11: Behavioral Response Function for Beaked Whales .....	3.8-120
Figure 3.8-12: Relative Likelihood of a Response Being Significant Based on the Duration and Severity of Behavioral Reactions.....	3.8-122
Figure 3.8-13: Navy Phase III Weighting Functions for All Species Groups .....	3.8-165
Figure 3.8-14: Navy Phase III Behavioral, TTS, and PTS Exposure Functions for Explosives.....	3.8-166

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### **3.8 Marine Mammals**

#### **3.8.1 Introduction**

As presented in Chapter 1 (Purpose and Need), the United States (U.S.) Department of the Navy (Navy) analysis presented in this document supplements both the 2011 Gulf of Alaska (GOA) Final Environmental Impact Statement (EIS)/Overseas Environmental Impact Statement (OEIS) (U.S. Department of the Navy, 2011a) and the 2016 GOA Final Supplemental EIS (SEIS)/OEIS (U.S. Department of the Navy, 2016a). The Proposed Action would occur over a maximum time period of up to 21 consecutive days during the months of April–October. Though the types of activities and number of events in the Proposed Action are the same as in the previous documents (Alternative 1 in both the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS), there have been changes in the platforms and systems used as part of those activities. Consistent with the previous analysis for Alternative 1, the sinking exercise (SINKEX) activity will not be part of the Proposed Action for this SEIS/OEIS. As was also the case for the previous analysis, the National Marine Fisheries Service (NMFS) is a cooperating agency with the Navy for this supplemental analysis, specifically where it relates to marine mammals and other marine resources under that agency’s regulatory purview.

The purpose of the marine mammals section in this SEIS/OEIS is to provide any new or changed information since the 2016 GOA Final SEIS/OEIS that are relevant to an analysis of potential impacts on marine mammals associated with the continuation of Navy training activities in the Temporary Maritime Activities Area (TMAA) beyond May 2022 as a result of the Proposed Action. The TMAA is beyond 12 nautical miles (NM) from shore, outside of the U.S. Territorial Sea. The current regulations pursuant to the Marine Mammal Protection Act (MMPA) authorization from NMFS for Navy training in the TMAA (82 Federal Register [FR] 24679 issued on May 30, 2017) are effective from April 26, 2017, through April 26, 2022.

The marine mammal species order of presentation is the same as presented in the 2016 GOA Final SEIS/OEIS. Background information in the 2016 GOA Final SEIS/OEIS for the marine mammal species that occur in the TMAA will not be repeated in this section unless necessary for context in support of new information and emergent relevant best available science. In addition to the annually updated marine mammal stock abundance estimates from NMFS in the applicable Stock Assessment Reports (SARs), there have been changes to the status for some species and stocks, new Distinct Population Segments (DPSs) designated, and newly proposed critical habitat since the 2016 GOA Final SEIS/OEIS. These data points and any other similarly changed information are presented in the subsections that follow.

#### **3.8.2 Affected Environment**

There have been three rounds of analysis for Navy training and testing activities at-sea in various Navy range complexes in the Pacific by the Navy and NMFS (see for example 83 FR 66846, December 27, 2018). In addition, and since 2006, the Navy, non-Navy marine mammal scientists, and research groups and institutions have conducted scientific monitoring and research in and around ocean areas in the Atlantic and Pacific where the Navy has been and proposes to continue training and testing. Data collected from Navy monitoring, scientific research findings, and annual reports have been provided to NMFS, and this public<sup>1</sup> record is an important contribution to the analysis of potential impacts on

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<sup>1</sup> Navy monitoring reports are available at the Navy website ([www.navy.mil/speciesmonitoring.us/](http://www.navy.mil/speciesmonitoring.us/)) and also at the NMFS website ([www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities](http://www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities)).

marine mammals. For example, these data provide information relevant to species distribution, habitat use, and evaluation of potential responses to Navy activities. New findings regarding marine mammal habitat use have been made available since the 2016 GOA Final SEIS/OEIS, but this research involves locations outside of the TMAA (Abrahms et al., 2019b; Becker et al., 2016; Becker et al., 2018; Becker et al., 2017; Mannocci et al., 2017; National Oceanic and Atmospheric Administration, 2019b; Pirodda et al., 2018b; Rockwood et al., 2017; Santora et al., 2017).

Monitoring projects are performed using a variety of methods, including visual surveys from surface vessels and aircraft, as well as passive acoustics before, during, and after Navy activities have been conducted. The Navy also has continued to contribute to funding of basic research, including behavioral response studies specifically designed to determine the effects to marine mammals from the Navy's main mid-frequency surface ship anti-submarine warfare sonar and other acoustic sources and explosives.

Training events that have occurred in the TMAA are much less frequent and generally smaller in scope, in comparison to training occurring in Navy range complexes in the waters off Washington, Oregon, California, and Hawaii. As a result, the majority of the Navy's marine mammal monitoring and research efforts have been focused on those other locations where Navy activities are more numerous and generally more intense. For more than a decade, the Navy has been submitting exercise reports and monitoring reports to NMFS for the Navy's range complexes in the Pacific and the Atlantic. These publicly available exercise reports, monitoring reports, and the associated research findings have been integrated into adaptive management decisions regarding the focus for subsequent research and monitoring as determined in collaborations between Navy, NMFS, Marine Mammal Commission, and other marine resource subject matter experts using an adaptive management approach. For example, see the 2017 U.S. Navy Annual Marine Species Monitoring Report for the Pacific that was made available to the public in April 2018 (U.S. Department of the Navy, 2018a).

The reporting, monitoring, and research efforts in the Pacific are relevant to the TMAA, since in many cases the marine mammals occurring in the GOA are part of the same trans-boundary populations (for example, Hawaii DPS humpback whales, gray whales, elephant seals) occurring in those other range complexes, and the Navy activities that have been monitored are fundamentally the same. This research and monitoring has continued to broaden the sample of observations regarding the general health of marine mammal populations in locations where the Navy has been conducting training and testing activities for decades, which has been considered in the analysis of marine mammal impacts presented in this SEIS/OEIS. This public record of training and testing activities, monitoring, and research from across the Navy range complexes in the Pacific and Atlantic since 2006 now spans more than 14 years.

Based on the monitoring to date, it has been the Navy's assessment and that of NMFS that it is unlikely there would be impacts on populations of marine mammals (such as whales, dolphins, and pinnipeds) having any long-term consequences as a result of the proposed continuation of training in the TMAA. This assessment of likelihood is based on four indicators from areas in the Pacific where Navy training and testing has been ongoing for decades: (1) evidence suggesting or documenting increases in the numbers of marine mammals present, (2) examples of documented presence and site fidelity of species and long-term residence by individual animals of some species, (3) use of training and testing areas for breeding and nursing activities, and (4) 14 years of comprehensive monitoring data indicating a lack of any observable effects to marine mammal populations such as direct mortalities or strandings occurring as a result of Navy training. The evidence to date suggests the continuing viability of marine mammal populations where Navy trains and tests, and an absence of any direct evidence suggesting Navy training

and testing has had or may have any long-term consequences to marine mammal populations. Barring any evidence to the contrary, therefore, what limited evidence there is from the monitoring reports and additional other focused scientific investigations should be considered in the analysis of impacts on marine mammals. Examples include information suggesting that

- the Endangered Species Act (ESA)-listed blue whale population in the Pacific, which includes the TMAA as part of their habitat, may have recovered and been at a stable level based on recent surveys and scientific findings (Barlow, 2016; Campbell et al., 2015; Carretta et al., 2017b; Monnahan et al., 2015; Rockwood et al., 2017; Širović et al., 2015b);
- gray whales in the Eastern North Pacific have recovered and are no longer listed under the ESA (International Whaling Commission, 2014);
- fin whale densities in the California Current Ecosystem have reached “current ecosystem limits” (Moore & Barlow, 2011);
- Cuvier’s beaked whales have been documented showing long-term residency and a population with higher densities than expected based on other nearby regions around the Southern California (SOCAL) Range Complex where the Navy has been intensively training and testing for decades, (Falcone & Schorr, 2012; Falcone et al., 2009; Hildebrand & McDonald, 2009; Schorr et al., 2014; Schorr et al., 2018); and
- the sea otter population at San Nicolas Island has increased about 10.5 percent per year, which is higher than the trend for the remainder of the population along the California coast (Hatfield et al., 2018; Hatfield et al., 2019).

In general, the evidence from reporting, monitoring, and research over more than a decade indicates that while the Proposed Action may result in the incidental harassment of marine mammals and may include auditory injury to some individuals, these impacts are expected to be inconsequential at the level of the marine mammal populations. There is no direct evidence that routine Navy training spanning decades has negatively impacted marine mammal populations at any Navy Range Complex or in the TMAA. In fact, for some of the most intensively used Navy training and testing areas, the continued multi-year presence of long-term resident individual animals and small populations (Baird, 2018; Baird et al., 2015; Baird et al., 2017; Baird et al., 2018; Baird et al., 2016; Lammers et al., 2017; Schorr et al., 2014; Schorr et al., 2018; Tinker & Hatfield, 2016; U.S. Department of the Navy, 2017b), resident females documented with and without calves from year to year, and high abundances on the Navy ranges for some species in comparison to other off-range locations (Moore & Barlow, 2017; Schorr et al., 2018; U.S. Department of the Navy, 2017b) provide no indications of significant impact and do provide evidence of generally increasing and healthy marine mammal populations. This background information forms part of the basis for the analysis of environmental consequences resulting from the Proposed Action consisting of the continuation of Navy training in the TMAA. Since the 2016 GOA Final SEIS/OEIS, there has been an approximate four-year period of additional monitoring with reports presented to NMFS regarding observations in and around Navy training and testing activities in the Pacific (see for example, U.S. Department of the Navy (2018a)), which provide further information for the assessment of significance of likely impacts resulting from the Navy’s Proposed Action in the TMAA.

As part of the baseline for analysis in this SEIS/OEIS, it is important to recognize that Navy training events in around the TMAA have been occurring there since the mid-1990s, without any indications of significant impact on the environment in general or marine mammals in particular. The most recent Navy activities in the TMAA as covered under the 2011 GOA EIS/OEIS and the 2016 GOA Final SEIS/OEIS, and as

authorized by NMFS have been the exercises “Northern Edge 17,” which was conducted in May 2017, and “Northern Edge 19,” which was conducted in May 2019.

The approach to analysis of potential impacts on marine mammals due to the Proposed Action was also based on the review of scientific publications cited in this section, from recent Navy documents that analyzed potential impacts from the same or similar activities on marine mammals (U.S. Department of the Navy, 2018b, 2018c), and from conclusions in this regard provided by NMFS pursuant to the MMPA (82 FR 19530) and as analyzed pursuant to the ESA in the current NMFS Biological Opinion (National Marine Fisheries Service, 2017b).

### 3.8.2.1 General Background

The ESA provides for listing species, subspecies, or DPSs, all of which are referred to as “species” under the ESA. The Interagency Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the ESA provides that, “... any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature” (61 FR 4722, February 7, 1996; 81 FR 62660, September 8, 2016). In short, a DPS is a portion of a species' or subspecies' population that is both discrete from the remainder of the population and significant in relation to the entire species, with the DPS then defined geographically instead of biologically.

As consistent with 2016 GOA Final SEIS/OEIS, the Navy follows the naming conventions presented by NMFS in the applicable annual SARs for the Pacific and Alaska covering the marine mammals present in the TMAA (Carretta et al., 2020b; Muto et al., 2020). These species and stocks present in the TMAA are provided in Table 3.8-1 along with an abundance estimate and associated coefficient of variation value as provided by the SARs (Carretta et al., 2020b; Muto et al., 2020; U.S. Fish and Wildlife Service, 2017).

**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA**

<i>Common Name</i>	<i>Scientific Name<sup>1</sup></i>	<i>Stock<sup>2</sup></i>	<i>Stock Abundance<sup>3</sup> (CV)</i>	<i>Occurrence in TMAA<sup>4</sup></i>	<i>ESA/MMPA Status</i>
<b>Order Cetacea</b>					
<b><i>Suborder Mysticeti (baleen whales)</i></b>					
<b><i>Family Balaenidae (right whales)</i></b>					
North Pacific right whale	<i>Eubalaena japonica</i>	Eastern North Pacific	31 (0.226)	Rare	Endangered/ Depleted
<b><i>Family Balaenopteridae (rorquals)</i></b>					
Humpback whale	<i>Megaptera novaeangliae</i>	Central North Pacific	10,103 (0.300)	Seasonal; highest likelihood June to September	-
		California, Oregon, and Washington <sup>5</sup>	4,973 (0.05)	Seasonal; highest likelihood June to September	Threatened/ Endangered /Depleted
		Western North Pacific	1,107 (0.300)	Seasonal; highest likelihood June to September	Endangered/ Depleted

**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA  
(continued)**

Common Name	Scientific Name <sup>1</sup>	Stock <sup>2</sup>	Stock Abundance <sup>3</sup> (CV)	Occurrence in TMAA <sup>4</sup>	ESA/MMPA Status
Order Cetacea (continued)					
Suborder Mysticeti (baleen whales) (continued)					
Family Balaenopteridae (rorquals) (continued)					
Blue whale	<i>Balaenoptera musculus</i>	Eastern North Pacific	1,898 (0.08)	Seasonal; highest likelihood June to December	Endangered/ Depleted
		Central North Pacific	133 (1.09)	Seasonal; highest likelihood June to December	Endangered/ Depleted
Fin whale	<i>Balaenoptera physalus</i>	Northeast Pacific	Not available	Likely	Endangered/ Depleted
Sei whale	<i>Balaenoptera borealis</i>	Eastern North Pacific <sup>6</sup>	519 (0.4)	Rare	Endangered/ Depleted
Minke whale	<i>Balaenoptera acutorostrata</i>	Alaska	Not available	Likely	-
Family Eschrichtiidae (gray whale)					
Gray whale	<i>Eschrichtius robustus</i>	Eastern North Pacific	26,960 (0.05)	Likely: Highest numbers during seasonal migrations (June through August)	-
		Western North Pacific	290 (N/A)	Rare: Individuals migrate through GOA	Endangered/ Depleted
Suborder Odontoceti (toothed whales)					
Family Physeteridae (sperm whale)					
Sperm whale	<i>Physeter macrocephalus</i>	North Pacific	Not available	Likely; More likely in waters > 1,000 m depth, most often > 2,000 m	Endangered/ Depleted
Family Delphinidae (dolphins)					
Killer whale	<i>Orcinus orca</i>	Eastern North Pacific Alaska Resident <sup>7</sup>	2,347 (N/A)	Likely	-
		Eastern North Pacific Northern Resident <sup>7</sup>	302 (N/A)	Extralimital	-

**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA  
(continued)**

<i>Common Name</i>	<i>Scientific Name<sup>1</sup></i>	<i>Stock<sup>2</sup></i>	<i>Stock Abundance<sup>3</sup> (CV)</i>	<i>Occurrence in TMAA<sup>4</sup></i>	<i>ESA/MMPA Status</i>
<b>Order Cetacea (continued)</b>					
<b>Suborder Odontoceti (toothed whales) (continued)</b>					
<b>Family Delphinidae (dolphins) (continued)</b>					
Killer whale	<i>Orcinus orca</i>	West Coast Transient <sup>7</sup>	243 (N/A)	Infrequent: few sightings	-
		AT1 Transient <sup>7</sup>	7 (N/A)	Rare; more likely inside Prince William Sound and Kenai Fjords	-
		Eastern North Pacific GOA, Aleutian Island, and Bering Sea Transient <sup>7</sup>	587 (N/A)	Likely	-
Pacific white-sided dolphin	<i>Lagenorhynchus obliquidens</i>	North Pacific	26,880 (N/A)	Likely	-
<b>Family Phocoenidae (porpoises)</b>					
Harbor porpoise	<i>Phocoena phocoena</i>	GOA	31,046 (0.21)	Rare; more likely nearshore but some inshore to the slope	-
		Southeast Alaska	Not available	Rare; more likely nearshore but some inshore to the slope	-
Dall's porpoise	<i>Phocoenoides dalli</i>	Alaska	83,400 (0.097)	Likely	-
<b>Family Ziphiidae (beaked whales)</b>					
Cuvier's beaked whale	<i>Ziphius cavirostris</i>	Alaska	Not available	Likely	-
Baird's beaked whale	<i>Berardius bairdii</i>	Alaska	Not available	Likely	-
Stejneger's beaked whale	<i>Mesoplodon stejnegeri</i>	Alaska	Not available	Likely	-

**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA  
(continued)**

Common Name	Scientific Name <sup>1</sup>	Stock <sup>2</sup>	Stock Abundance <sup>3</sup> (CV)	Occurrence in TMAA <sup>4</sup>	ESA/MMPA Status
Order Carnivora					
Suborder Pinnipedia <sup>8</sup>					
Family Otariidae (fur seals and sea lions)					
Steller sea lion	Eumetopias jubatus	Eastern U.S.	41,201 (N/A)	Rare (Nearshore and over the shelf east of the TMAA)	-
		Western U.S.	54,624 (N/A)	Likely in the inshore portion of the TMAA	Endangered/ Depleted
California sea lion	Zalophus californianus	U.S.	257,606 (N/A)	Rare (April and May)	-
Northern fur seal	Callorhinus ursinus	Eastern Pacific	620,660 (0.2)	Likely	Depleted
		California	14,050 (N/A)	Rare	-
Family Phocidae (true seals)					
Northern elephant seal	Mirounga angustirostris	California Breeding	179,000 (N/A)	Seasonal (highest likelihood July-September)	-
Harbor seal	Phoca vitulina	N. Kodiak	8,677 (N/A)	Likely in the inshore portion of the TMAA	-
		S. Kodiak	26,448 (N/A)	Likely in the inshore portion of the TMAA	-
		Prince William Sound	44,756 (N/A)	Likely in the inshore portion of the TMAA	-
		Cook Inlet/Shelikof	28,411 (N/A)	Likely in the inshore portion of the TMAA	-
Ribbon seal	Histiophoca fasciata	Alaska	184,697 (N/A)	Rare	

**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA  
(continued)**

<i>Common Name</i>	<i>Scientific Name<sup>1</sup></i>	<i>Stock<sup>2</sup></i>	<i>Stock Abundance<sup>3</sup> (CV)</i>	<i>Occurrence in TMAA<sup>4</sup></i>	<i>ESA/MMPA Status</i>
<i>Family Mustelidae</i>					
Northern sea otter	<i>Enhydra lutris kenyoni</i>	Southeast Alaska	25,712 (N/A)	Extralimital	-
		Southcentral Alaska	18,297 (N/A)	Rare	-
		Southwest Alaska	54,771 (N/A)	Rare	Threatened

<sup>1</sup>Taxonomy follows the naming conventions of the Society for Marine Mammalogy Committee on Taxonomy (2017); (Committee on Taxonomy, 2018) and the NMFS Stock Assessment Reports (Carretta et al., 2020b; Muto et al., 2020; U.S. Fish and Wildlife Service, 2018).

<sup>2</sup>Stock names and designations for the U.S. Exclusive Economic Zones are from the Pacific Stock Assessment Report (Carretta et al., 2020b; U.S. Fish and Wildlife Service, 2018), Alaska Stock Assessment Report (Muto et al., 2020) and USFWS (U.S. Fish and Wildlife Service, 2018).

<sup>3</sup>The stated coefficient of variation (CV) from the NMFS Stock Assessment Reports is an indicator of uncertainty in the abundance estimate and describes the amount of variation with respect to the population mean. It is expressed as a fraction or sometimes a percentage and can range upward from zero, indicating no uncertainty, to high values. For example, a CV of 0.85 would indicate high uncertainty in the population estimate. When the CV exceeds 1.0, the estimate is very uncertain. The uncertainty associated with movements of animals into or out of an area (due to factors such as availability of prey or changing oceanographic conditions) is much larger than is indicated by the CVs that are given.

<sup>4</sup>EXTRALIMITAL: There may be sightings, acoustic detections, or stranding records, but the TMAA and GOA are outside the species range of normal occurrence. RARE: The distribution of the species is near enough to the TMAA that the species could occur there, or there are a few confirmed sightings. INFREQUENT: Confirmed, but irregular sightings or acoustic detections. LIKELY: Year-round sightings or acoustic detections of the species in the TMAA, although there may be variation in local abundance over the year. SEASONAL: Species absence and presence as documented by surveys or acoustic monitoring. Names for the four areas within the TMAA follow the survey strata terminology as presented in Rone et al. (2017).

<sup>5</sup>Humpback whales in the Central North Pacific stock and the California, Oregon, and Washington stock are from three Distinct Population Segments based on animals identified in breeding areas in Hawaii, Mexico, and Central America (Carretta et al., 2020b; Muto et al., 2020; National Marine Fisheries Service, 2016a, 2016d, 2016e; Titova et al., 2017; Wade et al., 2016). All three stocks and all three DPSs co-occur in the TMAA (National Marine Fisheries Service, 2016d, 2016i).

<sup>6</sup>This analysis assumes that these individuals are from the Eastern North Pacific stock; however, they are not discussed in the West Coast or the Alaska Stock Assessment Reports (Carretta et al., 2020b; Muto et al., 2020).

<sup>7</sup>The abundance is based on counts of individual animals identified from photo-identification catalogues. Surveys for abundance estimates for these stocks are conducted infrequently (Muto et al., 2020).

<sup>8</sup>There are no data regarding the CV for some of the pinniped species given that abundance is determined by different methods than those used for cetaceans.



**Table 3.8-1: Marine Mammals with Possible or Confirmed Presence Within the TMAA  
(continued)**

<i>Common Name</i>	<i>Scientific Name<sup>1</sup></i>	<i>Stock<sup>2</sup></i>	<i>Stock Abundance<sup>3</sup> (CV)</i>	<i>Occurrence in TMAA<sup>4</sup></i>	<i>ESA/MMPA Status</i>
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<sup>9</sup>The Pribilof Islands stock abundance estimate is simply the count of seals ashore during the survey and does not include a correction for seals in the water.

<sup>10</sup>There are no data regarding the CV for sea otter given that abundance is determined by different methods than those used for cetaceans.

Notes: The stocks and stock abundance number are as provided in Carretta et al. (2020b); Muto et al. (2020) with exceptions for blue whales and the California, Oregon, Washington stock of humpback whales which reflect more recent data Calambokidis and Barlow (2020) than what is presented in the 2019 SARs.

CV = coefficient of variation, ESA = Endangered Species Act, GOA = Gulf of Alaska, m = meter(s), MMPA = Marine Mammal Protection Act, N/A = not available, U.S. = United States.

The abundance provided is the number of animals in a stock that NMFS has estimated are present in the specific portion of U.S. waters covered by that SAR (National Marine Fisheries Service, 2016c). For example, 2018 abundance for the North Pacific stock of Pacific white-sided dolphins (26,880) is only the number of those animals present within 200 NM of the Alaska coast (the Economic Exclusion Zone [EEZ]), even though the total population that must be used by NMFS to determine what constitutes a negligible impact numbered an estimated 931,000 individuals when last counted (Muto et al., 2020). Most marine mammal species are transboundary animals, and given that most counts are based on surveying only within the EEZ, the stock abundance estimates are not always inclusive of the total population number for a stock or species. The coefficient of variation provided for each of the abundances is a statistical term that describes the variation possible in the estimate of the stock abundance. The minimum population estimate is either a direct count (e.g., pinnipeds on land) or the lower 20th percentile of a statistical abundance estimate for a stock.

#### **3.8.2.1.1 Species Unlikely to be Present in the Temporary Maritime Activities Area**

There has been no change in the species unlikely to be present in the TMAA since the 2016 GOA Final SEIS/OEIS. The species carried forward for analysis are those likely to be found in the TMAA based on the most recent data available and do not include species that may have once inhabited or transited the area but have not been sighted in recent years (e.g., species which were extirpated from factors such as nineteenth and twentieth century commercial exploitation). Several species that may be present in the northeast Pacific Ocean have an extremely low probability of presence in the TMAA. These species are considered extralimital, meaning there may be a small number of sighting or stranding records within the TMAA, but the area of concern is outside the species range of normal occurrence. These species include beluga whale (*Delphinapterus leucas*), false killer whale (*Pseudorca crassidens*), short-finned pilot whale (*Globicephala macrorhynchus*), northern right whale dolphin (*Lissodelphis borealis*), and Risso's dolphin (*Grampus griseus*), and have been excluded from subsequent analysis for the same reasons as described in the 2016 GOA Final SEIS/OEIS.

#### **3.8.2.1.2 Group Size**

Group size characteristics are again incorporated into acoustic effects modeling for the given marine mammal density, but this is now based on information that has been updated (U.S. Department of the Navy, 2020b) since the 2016 GOA Final SEIS/OEIS.

### 3.8.2.1.3 Diving Behavior

Diving behavior has again been incorporated into acoustic effects modeling for the various marine mammal species, but this is now based on information that has been updated (U.S. Department of the Navy, 2020b) since the 2016 GOA Final SEIS/OEIS.

### 3.8.2.1.4 Hearing and Vocalization

The typical terrestrial mammalian ear (which is ancestral to that of marine mammals) consists of an outer ear that collects and transfers sound to the tympanic membrane and then to the middle ear (Fay & Popper, 1994; Rosowski, 1994). The middle ear contains ossicles that amplify and transfer acoustic energy to the sensory cells (called hair cells) in the cochlea, which transforms acoustic energy into electrical neural impulses that are transferred by the auditory nerve to high levels in the brain (Møller, 2013). All marine mammals display some degree of modification to the terrestrial ear; however, there are differences in the hearing mechanisms of marine mammals with an amphibious ear versus those with a fully aquatic ear (Wartzok & Ketten, 1999). Marine mammals with an amphibious ear include the marine carnivores: pinnipeds, sea otters, and polar bears (Ghoul & Reichmuth, 2014b; Owen & Bowles, 2011; Reichmuth et al., 2013). Outer ear adaptations in this group include external pinnae (ears) that are reduced or absent, and in the pinnipeds, cavernous tissue, muscle, and cartilaginous valves seal off water from entering the auditory canal when submerged (Wartzok & Ketten, 1999). Marine mammals with the fully aquatic ear (cetaceans and sirenians) use bone and fat channels in the head to conduct sound to the ear; while the auditory canal still exists, it is narrow and sealed with wax and debris, and external pinnae are absent (Castellini et al., 2016; Ketten, 1998).

The most accurate means of determining the hearing capabilities of marine mammal species are direct measurements of auditory system sensitivity (Nachtigall et al., 2000; Supin et al., 2001). Studies using these methods produce audiograms—plots describing hearing threshold (the quietest sound a listener can hear) as a function of frequency. Marine mammal audiograms, like those of terrestrial mammals, typically have a “U-shape,” with a frequency region of best hearing sensitivity at the bottom of the “U” and a progressive decrease in sensitivity outside of the range of best hearing (Fay, 1988; Mooney et al., 2012; Nedwell et al., 2004; Reichmuth et al., 2013). The “gold standard” for producing audiograms is the use of behavioral (psychophysical) methods, where marine mammals are trained to respond to acoustic stimuli (Nachtigall et al., 2000). For species that are untrained for behavioral psychophysical procedures, those that are difficult to house under human care, or in stranding rehabilitation and temporary capture contexts, auditory evoked potential (AEP) methods are used to measure hearing sensitivity (e.g., Castellote et al., 2014; Finneran et al., 2009; Montie et al., 2011; Mulsow et al., 2011; Nachtigall et al., 2008; Nachtigall et al., 2007; Supin et al., 2001). For odontocetes, the procedure for determining audiograms through auditory evoked potential methods has recently been standardized (American National Standards Institute & Acoustical Society of America, 2018).

These AEP methods, which measure electrical potentials generated by the auditory system in response to sound and do not require the extensive training needed for psychophysical methods, can provide an efficient estimate of hearing sensitivity (Finneran & Houser, 2006; Schlundt et al., 2007; Yuen et al., 2005). The thresholds provided by AEP methods are, however, typically elevated above behaviorally measured thresholds, and AEP methods are not appropriate for estimating hearing sensitivity at frequencies much lower than the region of best hearing sensitivity (Finneran, 2015; Finneran et al., 2016). For marine mammal species for which access is limited and therefore psychophysical or AEP

testing is impractical (e.g., mysticete whales and rare species), some aspects of hearing can be estimated from anatomical structures, frequency content of vocalizations, and extrapolations from related species.

Direct measurements of hearing sensitivity exist for approximately 25 of the nearly 130 species of marine mammals. Table 3.8-2 summarizes hearing capabilities for marine mammal species in the study area. For this analysis, marine mammals are arranged into the following functional hearing groups based on their generalized hearing sensitivities: high-frequency cetaceans (HF group: porpoises, *Kogia* spp.), mid-frequency cetaceans (MF group: delphinids, beaked whales, sperm whales), low-frequency cetaceans (LF group: mysticetes), otariids and other non-phocid marine carnivores in water and air (OW and OA groups: sea lions, otters), and phocids in water and air (PW and PA groups: true seals). Note that the designations of high-, mid-, and low-frequency cetaceans are based on relative differences of sensitivity between groups, as opposed to conventions used to describe active sonar systems.

For Phase III analyses, a single representative composite audiogram (Figure 3.8-1) was created for each functional hearing group using audiograms from published literature. For discussion of all marine mammal functional hearing groups and their derivation see technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects (Phase III)* (U.S. Department of the Navy, 2017a). These auditory composite audiograms were recently published by Southall et al. (2019c). The mid-frequency cetacean composite audiogram is consistent with behavioral audiograms of killer whales (Branstetter et al., 2017a) and audiograms of healthy wild belugas obtained via auditory evoked potential methods (Mooney et al., 2018) that were published following development of the technical report. The high-frequency cetacean composite audiogram is consistent with behavioral audiograms of harbor porpoises (Kastelein et al., 2017b) published after the technical report.

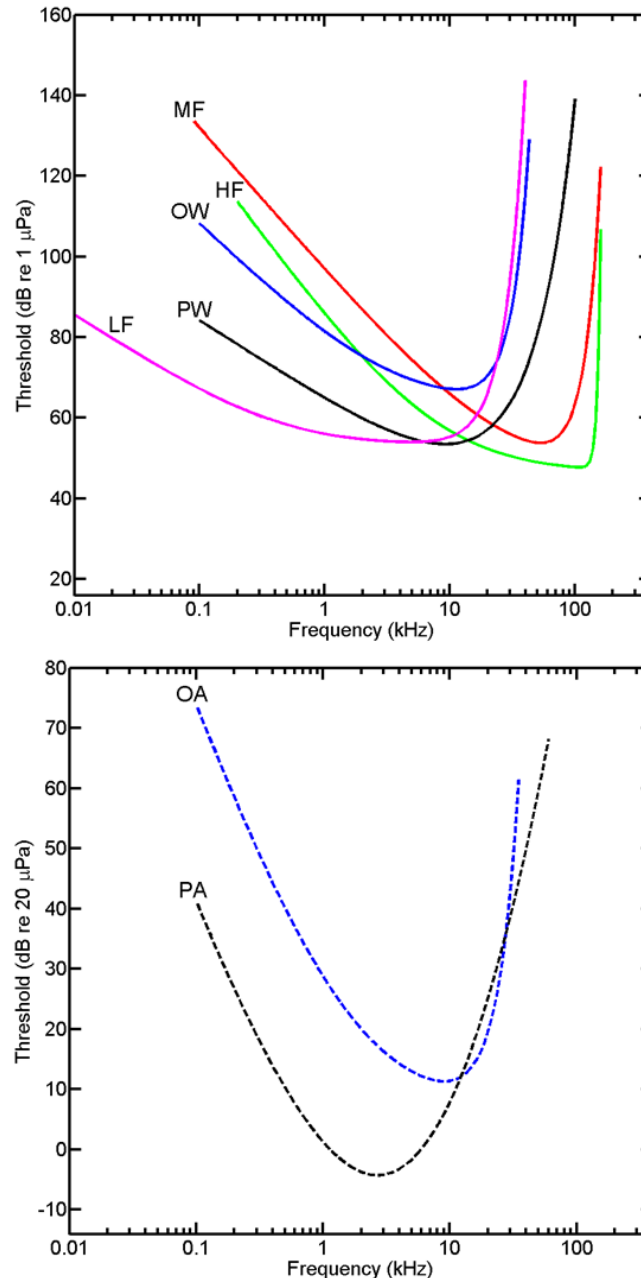
Few field studies aim to determine the hearing range of low-frequency cetaceans. However, Frankel and Stein (2020) exposed migrating gray whales to moored-source IMAPS sonar transmissions in the 21–25 kilohertz (kHz) frequency band (estimated RL = 148 decibels referenced to 1 micropascal squared [dB re 1  $\mu\text{Pa}^2$ ]), demonstrating that whales moved closer inshore when the vessel range was 1–2 kilometers (km) during sonar transmissions. The authors concluded that gray whales can hear up to 21 kHz. This evidence supports the mysticete hearing range extending up to 30 kHz, as reflected in the LF cetacean composite audiogram estimated by Southall et al. (2019c) and the Navy (U.S. Department of the Navy, 2017a).

Lastly, the otariid and phocid composite audiograms are consistent with recently published behavioral audiograms (Cunningham & Reichmuth, 2015; Kastelein et al., 2019b). This recent work shows that phocid detection thresholds are around 4 decibels (dB) lower for longer-duration sounds with harmonics than shorter-duration tonal sounds without harmonics (Kastelein et al., 2019b), and pinniped hearing sensitivity at frequencies and thresholds far above the range of best hearing may drop off at a slower rate than previously predicted (Cunningham & Reichmuth, 2015).

Research has shown that hearing in bottlenose dolphins is directional, i.e., the relative angle between the sound source location and the dolphin affects the hearing threshold (Accomando et al., 2020; Au & Moore, 1984). Hearing sensitivity becomes more directional as the sound frequency increases, with the greatest sensitivity to sounds presented in front and below the dolphin. Other odontocete species with less elongated skull anatomy than the bottlenose dolphin also exhibit direction-dependent hearing, but to a lesser degree (Kastelein et al., 2019b; Kastelein et al., 2005a; Popov & Supin, 2009). Byl et al. (2019) showed that harbor seals likely have well-developed directional hearing for biologically relevant sounds (Section 3.8.3.1.1.4, Masking).

**Table 3.8-2: Species Within Marine Mammal Hearing Groups Likely Found in the TMAA**

<i>Hearing Group</i>	<i>Species within the Study Area</i>
High-frequency cetaceans	Dall's porpoise
	Harbor porpoise
Mid-frequency cetaceans	Baird's beaked whale
	Cuvier's beaked whale
	Killer whale
	Pacific white-sided dolphin
	Sperm whale
	Stejneger's beaked whale
Low-frequency cetaceans	Blue whale
	Fin whale
	Gray whale
	Humpback whale
	Minke whale
	North Pacific right whale
	Sei whale
Otariids and other non-phocid marine carnivores	California sea lion
	Northern fur seal
	Northern sea otter
	Steller sea lion
Phocids	Harbor seal
	Northern elephant seal
	Ribbon Seal



Source: *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017).

Notes: For hearing in water (top) and in air (bottom, phocids and otariids only). LF = low-frequency, MF = mid-frequency, HF = high-frequency, OW = otariids and other non-phocid marine carnivores in water, PW = phocids in water, OA = otariids and other non-phocid marine carnivores in air, PA = phocids in air.

**Figure 3.8-1: Composite Audiograms for Hearing Groups Likely Found in the TMAA**

Similar to the diversity of hearing capabilities among species, the wide variety of acoustic signals used in marine mammal communication (including biosonar or echolocation) is reflective of the diverse ecological characteristics of cetacean, sirenian, and carnivore species (see Avens, 2003; Richardson et al., 1995b). This makes a succinct summary difficult (see Richardson et al., 1995b; Wartzok & Ketten,

1999 for thorough reviews); however, a division can be drawn between lower frequency communication signals that are used by marine mammals in general, and the specific, high-frequency biosonar signals that are used by odontocetes to sense their environment.

Non-biosonar communication signals span a wide frequency range, primarily having energy up into the tens of kilohertz range. Of particular note are the very low-frequency calls of mysticete whales that range from tens of hertz (Hz) to several kilohertz, and have source levels of 150–200 decibels referenced to 1 micropascal (dB re 1  $\mu$ Pa) (Cummings & Thompson, 1971; Edds-Walton, 1997; Širović et al., 2007; Stimpert et al., 2007; Wartzok & Ketten, 1999). These calls most likely serve social functions such as mate attraction, but may serve an orientation function as well (Green, 1994; Green et al., 1994; Richardson et al., 1995b). Humpback whales are a notable exception within the mysticetes, with some calls exceeding 10 kHz (Zoidis et al., 2008).

Odontocete cetaceans and marine carnivores use underwater communicative signals that, while not as low in frequency as those of many mysticetes, likely serve similar functions. These include tonal whistles in some odontocetes and the wide variety of barks, grunts, clicks, sweeps, and pulses of pinnipeds. Of additional note are the aerial vocalizations that are produced by pinnipeds, otters, and polar bears. Again, the acoustic characteristics of these signals are quite diverse among species, but can be generally classified as having dominant energy at frequencies below 20 kHz (Richardson et al., 1995b; Wartzok & Ketten, 1999).

Odontocete cetaceans generate short-duration (50–200 microseconds), specialized clicks used in biosonar with peak frequencies between 10 and 200 kHz to detect, localize, and characterize underwater objects such as prey (Au, 1993; Wartzok & Ketten, 1999). These clicks are often more intense than other communicative signals, with reported source levels as high as 229 dB re 1  $\mu$ Pa peak-to-peak (Au et al., 1974). The echolocation clicks of high-frequency cetaceans (e.g., porpoises) are narrower in bandwidth (i.e., the difference between the upper and lower frequencies in a sound) and higher in frequency than those of mid-frequency cetaceans (Madsen et al., 2005; Villadsgaard et al., 2007).

In general, frequency ranges of vocalization lie within the audible frequency range for an animal (i.e., animals vocalize within their audible frequency range); however, auditory frequency range and vocalization frequencies do not perfectly align. For example, odontocete echolocation clicks contain a broad range of frequencies, and not all of the frequency content is necessarily heard by the individual that emitted the click. The frequency range of vocalization in a species can therefore be used to infer some characteristics of their auditory system; however, caution must be taken when considering vocalization frequencies alone in predicting the hearing capabilities of species for which no data exist (i.e., mysticetes). It is important to note that aspects of vocalization and hearing sensitivity are subject to evolutionary pressures that are not solely related to detecting communication signals. For example, hearing plays an important role in detecting threats (e.g., Deecke et al., 2002), and high-frequency hearing is advantageous to animals with small heads in that it facilitates sound localization based on differences in sound levels at each ear (Heffner & Heffner, 1982). This may be partially responsible for the difference in best hearing thresholds and dominant vocalization frequencies in some species of marine mammals (e.g., Steller sea lions, Mulsow & Reichmuth, 2010).

#### **3.8.2.1.5 General Threats**

As noted in the 2016 GOA Final SEIS/OEIS and subsequently provided in various publications (Barcenas De La Cruz et al., 2017; Bradford & Lyman, 2015; Carretta et al., 2019a; Carretta et al., 2019b; Carretta et

al., 2020a; Delean et al., 2020; Esquible & Atkinson, 2019; Helker et al., 2019), investigations of stranded marine mammals are undertaken to monitor threats to marine mammals and out of concerns for animal welfare and ocean stewardship. For the marine mammal populations present in Alaska waters, data regarding human-caused mortality and injury to NMFS-managed stocks are available in a NMFS Technical Memorandum for marine mammal stocks in Alaska (Delean et al., 2020; Helker et al., 2019) and for stocks present on the U.S. West Coast (Carretta et al., 2019a; Carretta et al., 2020a). The known serious injury and mortalities resulting from non-Navy human activities that these reports summarize give important context in reviewing the analysis of potential impacts that may result from the continuation of Navy training in the TMAA.

Causes for strandings also include natural causes such as infectious disease, parasite infestation, climate change, harmful algal blooms and associated biotoxins, and tectonic events such as underwater earthquakes (for more information see NMFS Marine Mammal Stranding Response Fact Sheet; National Marine Fisheries Service (2016b) and National Marine Fisheries Service (2020a)). For a general discussion of strandings and their causes as well as strandings in association with U.S. Navy activity, see the technical report titled *Marine Mammal Strandings Associated with U.S. Navy Sonar* (U.S. Department of the Navy, 2017c).

### **Climate Change**

The global climate is warming and is having impacts on some populations of marine mammals (Garcia-Aguilar et al., 2018; Jefferson & Schulman-Janiger, 2018; National Marine Fisheries Service, 2020c; National Oceanic and Atmospheric Administration, 2015b, 2018b; Peterson et al., 2006; Salvadeo et al., 2010; Sanderson & Alexander, 2020; Shirasago-Germán et al., 2015; Silber et al., 2017; Simmonds & Elliott, 2009; Straley et al., 2017; Szpak et al., 2018; von Biela et al., 2019). Climate change can affect marine mammal species directly by causing shifts in distribution to match physiological tolerance under changing environmental conditions (Doney et al., 2012; National Marine Fisheries Service, 2018d; Peterson et al., 2006; Silber et al., 2017), which may or may not result in net habitat loss (some can experience habitat gains). Climate change can also affect marine mammals indirectly via impacts on prey, changing prey distributions and locations, and changes in water temperature (Cheung & Frolicher, 2020; Giorli & Au, 2017; Peterson et al., 2006; Straley et al., 2017; von Biela et al., 2019). In Prince William Sound between 2012 and 2016, researchers suggested the quality of sand lance (the prey of humpbacks whale and other species) may have been reduced by increased water temperatures in North Pacific in 2015–2016, which probably contributed to population declines and/or breeding failures observed among several predators in the Gulf of Alaska (von Biela et al., 2019); see also National Marine Fisheries Service (2018e); Savage (2017); Savage (2020). Also note that because many marine mammals to the TMAA through waters off California, it is relevant that Sanford et al. (2019) have noted that severe marine heatwaves occurring off California in 2014–2016 triggered marine mammal mortality events, harmful algal blooms, and declines in subtidal kelp beds.

Changes in prey can impact marine mammal foraging success, which in turn affects reproduction success and survival. Starting in January 2013, an elevated number of strandings of California sea lion pups were observed in Southern California counties, such as Santa Barbara County, Ventura County, Los Angeles County, and Orange County. This unusual number of strandings, continuing into 2016, were declared an Unusual Mortality Event (UME) by NMFS (National Oceanic and Atmospheric Administration, 2018a, 2018b). Although this UME was still considered as “ongoing” through 2017, the number of strandings recorded in 2017 were at or below average (National Oceanic and Atmospheric Administration, 2018a). This is the sixth UME involving California sea lions that has occurred in California since 1991. For this

2013–2015 event, NMFS biologists indicated that warmer ocean temperatures have shifted the location of prey species that are no longer adjacent to the rookeries, which thereby impacted the female sea lions' ability to find food and supply milk to their pups (National Oceanic and Atmospheric Administration, 2018a). As a result, this confluence of natural events causes the pups to be undernourished, and many are subsequently found stranded dead or emaciated due to starvation. In a similar occurrence for gray whales and since January 2019, an elevated number of gray whale strandings has occurred along the west coast of North America from Mexico through Alaska, resulting in NMFS declaring a UME for this species (National Marine Fisheries Service, 2019a). This is similar to a previous UME for gray whales that occurred in 1999–2000. Using photogrammetry to assess the condition of gray whales while foraging along the Oregon coast over the three-year period between 2016 and 2018, researchers determined that the body condition of whales correlated with environmental changes and hypothesized that low prey availability between 2016 and 2018 carried over to result in the UME starting in 2019 (Lemos et al., 2020).

Likely also due to changing prey distributions, data tagging efforts in July 2016 focusing on blue and fin whales had to be shifted north to central California waters when the majority of blue, fin, and humpback whales encountered in Southern California waters were found to be too thin or otherwise in poor body condition to allow for them to be tagged (Oregon State University, 2017). In central California waters, the researchers identified good numbers of blue, fin, and humpback whales in better condition and indicative of a good feeding area that was likely to be sustained that season (Oregon State University, 2017).

Harmful algal blooms may become more prevalent in warmer ocean temperatures with increased salinity levels such that blooms will begin earlier, last longer, and cover a larger geographical range (Edwards, 2013; Moore et al., 2008). Warming ocean waters have been linked to the spread of harmful algal blooms into the North Pacific where waters had previously been too cold for most of these algae to thrive. The spread of the algae and associated blooms has led to mortality in marine mammals in locations where algae-caused biotoxicity had not been previously known (Lefebvre et al., 2016).

Climate change may indirectly influence marine mammals through changes in human behavior, such as increased shipping and oil and gas extraction, which benefit from sea ice loss (Alter et al., 2010). Ultimately impacts from global climate change may result in an intensification of current and on-going threats to marine mammals (Edwards, 2013). In addition, the ability of marine mammals to alter behaviors may serve as a buffer against measurable climate change-induced impacts and could delay or mask any adverse effects until critical thresholds are reached (Baker et al., 2016).

Marine mammals are influenced by climate-related phenomena, including storms and other extreme weather patterns, such as the 2015–2016 El Niño in the ocean off the U.S. West Coast. Generally, not much is known about how large storms and other weather patterns affect marine mammals, other than that mass strandings (when two or more marine mammals become beached or stuck in shallow water) sometimes coincide with hurricanes, typhoons, and other tropical storms (Bradshaw et al., 2006; Marsh, 1989; Rosel & Watts, 2008) or other oceanographic conditions.

Concerns over climate change modifying the U.S. West Coast upwelling patterns, increasing levels of hypoxia, and ocean acidification have generated targeted research and monitoring efforts at selected "Sentinel Sites" (Lott et al., 2011). There remains scientific uncertainty about how or if such changes will affect marine mammals and their prey, but acidification of the ocean could potentially impact the mobility, growth, and reproduction of calcium carbonate-forming organisms such as crustaceans and



plankton, which are the direct prey of some marine mammals, as well as an important part of the overall food chain in the ocean; as well as slightly altering the propagation of sound underwater (Lynch et al., 2018; Meyers et al., 2019; Rossi et al., 2016).

Habitat deterioration and loss is a major factor for almost all coastal and inshore species of marine mammals, with effects ranging from depleting a habitat's prey base to the complete loss of habitat (Ayres et al., 2012; Kemp, 1996; Pine et al., 2016; Rolland et al., 2012; Smith et al., 2009; Veirs et al., 2015; Williams et al., 2014a). Many researchers predict that if oceanic temperatures continue to rise with an associated effect on marine habitat and prey availability, then either changes in foraging or life history strategies, including poleward shifts in many marine mammal species distributions, should be anticipated (Alter et al., 2010; Fleming et al., 2016; Ramp et al., 2015; Salvadeo et al., 2015; Silber et al., 2017; Sydeman & Allen, 1999). Poloczanska et al. (2016) analyzed climate change impact data that integrate multiple climate influenced changes in ocean conditions (e.g., temperature, acidification, dissolved oxygen, and rainfall) to assess anticipated changes to a number of key ocean fauna across representative areas. Poloczanska et al. (2016) predict a northward expansion in the distribution of zooplankton, fish, and squid, all of which are prey for many marine mammal species. Sanford et al. (2019) have noted that severe marine heatwaves in the northeast Pacific in 2014–2016 triggered marine mammal mortality events, harmful algal blooms, and declines in subtidal kelp beds.

### **Commercial Industries**

Human impacts on marine mammals have received much attention in recent decades and include: fisheries interactions, including bycatch (accidental or incidental catch), gear entanglement, and indirect effects from takes of prey species; noise pollution; marine debris (ingestion and entanglement); hunting (both commercial and native practices); vessel strikes; entrainment in power plant water intakes; increased ocean acidification; and general habitat deterioration or destruction.

### ***Fishery Bycatch***

Fishery bycatch is likely the most impactful threat to marine mammal individuals and populations and may account for the deaths of more marine mammals than any other cause (Geijer & Read, 2013; Hamer et al., 2010; Northridge, 2009; Read, 2008). In 1994, the MMPA was amended to formally address bycatch. The amendment requires the development of a take reduction plan when bycatch exceeds a level considered unsustainable and will lead to marine mammal population decline. In addition, NMFS develops and implements take reduction plans that help recover and prevent the depletion of strategic stocks of marine mammals that interact with certain fisheries (National Marine Fisheries Service, 2016c). For example, fishing-related injuries occurring in 2013–2017 involving the stocks of marine mammals present in the TMAA totaled 448 marine mammals in that five-year period (Delean et al., 2020; Helker et al., 2019).

### ***Hunting***

Commercial hunting, as in whaling and sealing operations, provided the original impetus for marine mammal management efforts and has driven much of the early research on cetaceans and pinnipeds (Twiss & Reeves, 1999). With the enactment of the MMPA and the 1946 International Convention for the Regulation of Whaling, commercial hunting-related mortality has decreased over the last 40 years. Unregulated harvests are still considered to be direct threats; however, since passage of the MMPA, there have been relatively few serious calls for culls of marine mammals in the United States compared to other countries, including Canada (Roman et al., 2013). Review of uncovered Union of Soviet Socialist Republics catch records in the North Pacific Ocean indicate extensive illegal whaling activity between

1948 and 1979, with a harvest totaling 195,783 whales. Of these, 169,638 were reported (over 26,000 takes unreported) by the Union of Soviet Socialist Republics to the International Whaling Commission (Ilyashenko et al., 2014; Ilyashenko & Chapham, 2014; Ilyashenko et al., 2013, 2015). On July 1, 2019, Japan resumed commercial whaling within its EEZ (BBC News, 2019; Nishimura, 2019; Victor, 2018). Japan had set an annual quota of 227 whales until the end of the 2019, which included 52 minke whales, 150 Bryde's whales, and 25 sei whales (Nishimura, 2019); the annual quota set for 2020 was 383 whales total (Hurst, 2020). Although the resumed commercial whaling will only take place within the Japanese Exclusive Economic Zone waters, it is possible that some of the whales found in those waters may be part of the same North Pacific populations that are also present seasonally in the TMAA.

For U.S. waters, there is a provision in the MMPA that allows for subsistence harvest of marine mammals, primarily by Alaska Natives. Subsistence hunting by Russia and Alaska Natives also occurs in the North Pacific, Chukchi Sea, and Bering Sea, involving marine mammal stocks that may be present in the TMAA. For whales, the quotas for "aboriginal subsistence whaling" are established by the International Whaling Commission (International Whaling Commission, 2020). For example, the International Whaling Commission quotas for 2019–2025 are for a total of 980 gray whales with not more than 140 landed in any one year by native people in Chukotka (Russia) and Washington State (International Whaling Commission, 2020). For example, in Russian waters in 2013, there were a total of 127 gray whales "struck" during subsistence whaling by the inhabitants of the Chukchi Peninsula between the Bering and Chukchi Sea (Ilyashenko & Zharikov, 2014). These gray whales harvested in Russian waters may be individuals from either the endangered Western North Pacific stock or the non-ESA-listed Eastern North Pacific stock that may migrate through the TMAA. In 2017 at the Kuskowim River in Alaska, a gray whale was killed and harvested in what NMFS described as being an "illegal hunt" (Carretta et al., 2019a). In 2018, a total of 106 gray whales were harvested for subsistence use (International Whaling Commission, 2019b). Subsistence hunting of certain pinnipeds are also permitted by NMFS such as in 84 FR 52372 (dated Wednesday, October 2, 2019) which authorized, "... Pribilovians who reside on St. Paul Island, Alaska, to kill for subsistence uses each year up to 2,000 male fur seals less than seven years old ...." Subsistence hunting in nearshore waters also occurs in communities on Kodiak Island and the Kenai Peninsula. For example, the most recent report<sup>2</sup> from the Kodiak Island communities indicated that in 2011 there were a reported 163 harbor seals and 20 Western DPS Steller sea lions taken in that year (Wolfe et al., 2012). This was the third-lowest recorded number of harbor seals taken since reporting began in 1992 (Wolfe et al., 2012). The U.S. Fish and Wildlife Service (USFWS) records show that in 2012 there were 1,281 sea otters reported taken in Alaska as part of that year's subsistence harvest (Lichtenstein, 2013).

### ***Vessel Strike***

Ship strikes are also a growing issue for most marine mammals, although mortality may be a more significant concern for species that occupy areas with high levels of vessel traffic, because the likelihood of encounter would be greater (Aleutian Islands Waterways Safety Committee, 2019; Currie et al., 2017a; Keen et al., 2019; Moore et al., 2018; Redfern et al., 2013; Redfern et al., 2019; Rockwood et al., 2017; Ryan, 2019; Van der Hoop et al., 2013; Van der Hoop et al., 2015; Wright et al., 2018). Most

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<sup>2</sup> The Alaska Department of Fish and Game no longer collects data related to the subsistence harvest assessment program, and the most recent report for the Kodiak Island communities in 2011 and for sea otters in the State of Alaska was 2012.

reported marine mammal vessel strikes involve commercial vessels transiting over or near the continental shelf hitting whales (Laist et al., 2001; National Marine Fisheries Service, 2008, 2019d; Nichol et al., 2017; Scordino et al., 2020; Silber et al., 2008), but strikes also occur in coastal areas frequented by smaller vessels and involve smaller marine mammals and other species (Schoeman et al., 2020).

Available data from NMFS indicate that in Alaska in the five-year period between 2013 and 2017, mortalities or serious injuries occurred to a minimum of 29 humpbacks as a result of vessel strike (Delean et al., 2020), and along the U.S. Pacific coast between 2013 and 2017, there were an additional 14 known strikes involving humpback whales (Carretta et al., 2019a); these animals struck off California may be part of the same populations inhabiting the TMAA.

Since 1995, the U.S. Navy and U.S. Coast Guard have reported all known or suspected vessel collisions with whales to NMFS. The assumed under-reporting of whale collisions by vessels other than U.S. Navy or U.S. Coast Guard makes any comparison of data involving vessel strikes between Navy vessels and other vessels heavily biased. This under-reporting of civilian vessel collisions with whales is recognized by NMFS (Bradford & Lyman, 2015). Within Alaska waters, there were 28 reported marine mammal vessel strikes between 2013 and 2017 (none of which were from U.S. Navy vessels) (Delean et al., 2020), and for the U.S. West Coast in the same period there were 65 reported vessel strikes to marine mammals (Carretta et al., 2020b), which is an approximate average consistent with previous reporting periods (Carretta et al., 2019a; Carretta et al., 2018b; Helker et al., 2019; Helker et al., 2017).

### **Noise**

In some locations, especially where urban or industrial activities or commercial shipping is intense, anthropogenic noise can be a potential habitat-level stressor (Castellote et al., 2019; Dunlop, 2016; Dyndo et al., 2015; Erbe et al., 2018; Erbe et al., 2014; Frisk, 2012; Gabriele et al., 2017; Gedamke et al., 2016; Haver et al., 2018; Hermannsen et al., 2014; Li et al., 2015; McKenna et al., 2012; Melcón et al., 2012; Merchant et al., 2014; Merchant et al., 2012; Mikkelsen et al., 2019; Miksis-Olds & Nichols, 2016; Nowacek et al., 2015; Pine et al., 2016; Rice et al., 2018a; Williams et al., 2014c). Noise is of particular concern to marine mammals because many species use sound as a primary sense for navigating, finding prey, avoiding predators, and communicating with other individuals. Noise associated with tourism (whale watch vessels and cruise ships) is also a concern in some areas of Alaska (Cates et al., 2020; Frankel & Gabriele, 2017; Schuler et al., 2019; Sprogis et al., 2020). Noise may cause marine mammals to leave a habitat, impair their ability to communicate, or cause physiological stress (Burnham & Duffus, 2019; Cholewiak et al., 2018; Courbis & Timmel, 2008; Erbe, 2002; Erbe et al., 2019; Erbe et al., 2016; Gabriele et al., 2018; Hildebrand, 2009; Holt et al., 2017; Putland et al., 2018; Rolland et al., 2012; Southall et al., 2018; Tyack et al., 2011; Tyne et al., 2017; Wieland et al., 2010; Williams et al., 2014b; Williams et al., 2019; Wisniewska et al., 2018). Noise can cause behavioral disturbances, mask other sounds including their own vocalizations, may result in injury, and in some cases may result in behaviors that ultimately lead to death (Erbe et al., 2019; Erbe et al., 2016; Erbe et al., 2014; National Research Council, 2003, 2005; Nowacek et al., 2007; Southall et al., 2009; Tsujii et al., 2018; Tyack, 2009; Würsig & Richardson, 2009). As noted in Section 3.0 (Introduction), anthropogenic noise in the TMAA is generated from a variety of sources, including commercial shipping, oil and gas exploration activities, commercial and recreational fishing (including fish finding sonar, fathometers, and acoustic deterrent and harassment devices), recreational boating, research (including sound from air guns, sonar, and telemetry).

Ships leaving ports in Japan and Korea travel in a direct line following the North Pacific Great Circle Route to ports in Canada and Washington via the Unimak Pass in the Aleutian Islands (Aleutian Islands

Waterways Safety Committee, 2019; Nuka Research and Planning Group LLC, 2015). For example, there were a total of 28,302 vessel transits in the Bering Sea in 2015 (Adams & Silber, 2017). In addition, vessels calling at ports in Alaska including Anchorage and Prince William Sound may travel directly through the TMAA. As a result, commercial vessel noise is the main source of underwater anthropogenic noise in the area (Klinck et al., 2016; Rice et al., 2018b; Wiggins et al., 2017; Wiggins & Hildebrand, 2018). Redfern et al. (2017a) found that commercial vessel noise in some locations may have degraded the habitat for right whales, blue whales, fin whales, and humpback whales due to the loss of communication space where important habitat for these species overlaps with commercial vessel traffic (Redfern et al., 2017a; Rolland et al., 2016). Commercial vessel traffic running adjacent to the coast in the TMAA may be adjacent to or run through portions of the designated critical habitat for North Pacific right whales and biologically important areas for fin, gray, Cook Inlet beluga whales, and humpback whales (Castellote et al., 2019; Ferguson et al., 2015; Wiggins et al., 2017).

In many areas of the world, oil and gas seismic exploration in the ocean is undertaken using a group of air guns towed behind large research vessels. The airguns convert high-pressure air into very strong shock wave impulses that are designed to return information off the various buried layers of sediment under the seafloor. Seismic exploration surveys last many days and cover vast overlapping swaths of the ocean area being explored. Most of the impulse energy (analogous to underwater explosions) produced by these airguns is heard as low-frequency sound, which can travel long distances and has the potential to impact marine mammals. The National Marine Fisheries Service routinely issues permits for the taking of marine mammals associated with these commercial activities (see for example, 84 FR 27246, Wednesday, June 12, 2019).

### **Marine Debris and Pollution**

Approximately 80 percent of marine debris in the ocean come from land-based sources (California Ocean Protection Council & National Oceanic and Atmospheric Administration Marine Debris Program, 2018; Thiel et al., 2018). In a seafloor survey off Southern California where the Navy has routinely trained and tested for decades, urban refuse (beverage cans, bottles, household items, and construction materials) constituted approximately 88 percent of the identified debris observed (Watters et al., 2010). Without improved waste management and infrastructure in underdeveloped coastal countries worldwide, the cumulative quantity of plastic waste available to enter the ocean from land is predicted to increase by an order of magnitude by 2025 (Jambeck et al., 2015). Marine debris is a global threat to marine mammals (National Oceanic and Atmospheric Administration Marine Debris Program, 2014a). For example, entanglement of California sea lions documented along the north coast of Washington from 2010 to 2018 were mostly from packing bands (Allyn & Scordino, 2020). A literature review by Baulch and Perry (2014), found that 56 percent of cetacean species are documented as having ingested marine debris. Interactions between marine mammals and marine debris, including derelict fishing gear and plastics, are significant sources of injury and mortality (Baulch & Perry, 2014). Comparing the Baulch and Perry review with that conducted by an earlier investigation (Laist, 1997), the percentage of marine mammal species with documented records of entanglement in or ingestion of marine debris has increased from 43 to 66 percent over the past 18 years (Bergmann et al., 2015). Ingestion of marine debris by marine mammals is a less well-documented cause of mortality than entanglement, but it is a growing concern (Bergmann et al., 2015; Jacobsen et al., 2010; Paul, 2019; Puig-Lozano et al., 2018). Baulch and Perry (2014) found that ingestion of debris has been documented in 48 cetacean species, with rates of ingestion as high as 31 percent in some populations. Attributing cause of death to marine debris ingestion is difficult (Laist, 1997), but ingestion of plastic bags and Styrofoam has been identified

as the cause of injury or death of minke whales (De Pierrepont et al., 2005) and deep-diving odontocetes, including beaked whales (Baulch & Perry, 2014; Paul, 2019; Puig-Lozano et al., 2018), pygmy sperm whales (Sadove & Morreale, 1989; Stamper et al., 2006; Tarpley & Marwitz, 1993), and sperm whales (Jacobsen et al., 2010; Sadove & Morreale, 1989). As noted elsewhere, without improved waste management and infrastructure in undeveloped coastal countries worldwide, the cumulative quantity of plastic waste available to enter the ocean from land is predicted to increase by an order of magnitude by 2025 (Jambeck et al., 2015).

Marine mammals migrating to Alaska also encounter threats outside the TMAA (Díaz-Torres et al., 2016; Lian et al., 2018; Thiel et al., 2018). In Alaska from 2011 through 2015, records of approximately 3,700 human-marine mammal interactions were reviewed by NMFS and determined to have resulted in 440 entanglement/entrapment-related marine mammal serious injury or mortality to various species (Helker et al., 2017). For example, between 2011 and 2015 the most common cause of serious injuries for the Eastern U.S. stock of Steller sea lions was entanglement in marine debris or fishery gear (totaling 146 sea lions) (Helker et al., 2017); for the period from 2012 to 2016 this total was 117 seriously injured Steller sea lions (Helker et al., 2019). Entanglement of California sea lions and Steller sea lions documented along the north coast of Washington from 2010 to 2018 were mostly from shipping packing bands, followed by salmon flashers during the local ocean salmon troll season (Allyn & Scordino, 2020). Likely reflecting fishery practices across the North Pacific, in the Northwest Hawaiian Islands, where there have been active efforts at marine debris removal since 1996, the National Oceanic and Atmospheric Administration marine debris team has removed 848 metric tons of derelict fishing nets and debris and estimates an additional 52 metric tons of derelict fishing gear collects on the shallow coral reefs and shores there every year (National Oceanic and Atmospheric Administration, 2018e).

On the U.S. West Coast for the marine mammal stocks that are present in the TMAA, marine debris resulted in mortalities to 129 marine mammals in the five-year period from 2013 to 2017 (the majority California sea lions), two gray whales, and one each of the following species: humpback whale, minke whale, and harbor porpoise (Barcenas De La Cruz et al., 2017; Carretta et al., 2019a). From 2013 through 2017, there were 10 blue whales, 54 humpback whales, and six sperm whales entanglements documented for those ESA-listed species (Carretta et al., 2019a). Marine debris documented off the Mexican Central Pacific coast (Díaz-Torres et al., 2016) and waters farther south (Thiel et al., 2018) also have the potential to impact marine mammals that migrate to Alaska, such as the ESA-listed humpback whale DPSs from Mexico and the stock of blue whales along the U.S. West Coast that move at least as far south as the Costa Rica Dome<sup>3</sup> located off the west coast of Central America.

An estimated 75 percent or more of marine debris consists of plastic (California Coastal Commission, 2018; Derraik, 2002; Hardesty & Wilcox, 2017). High concentrations of floating plastic have been reported in the central areas of the North Atlantic and Pacific Oceans (Cozar et al., 2014). Plastic pollution found in the oceans is primarily dominated by particles smaller than 1 centimeter, commonly referred to as microplastics (Hidalgo-Ruz et al., 2012). Other researchers have defined microplastics as particles with a diameter ranging from a few micrometers up to 5 millimeters and not readily visible to the naked eye (Andrady, 2015). Most microplastic fragments and fibers found throughout the oceans

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<sup>3</sup> The Costa Rica Dome is an area of deep ocean upwelling in the Eastern Tropical Pacific, centered approximately 500 km off the west coast of Costa Rica and Nicaragua. The size of the roughly elliptical area varies from approximately 300 to 1,000 km in an east-west direction and is an area of high productivity and known wintering location for blue whales.

result from the breakdown of larger items, such as clothing, packaging, and rope and have accumulated in the pelagic zone and sedimentary habitats (Thompson et al., 2004). Results from the investigation by Browne et al. (2011) have also suggested that microplastic fibers are discharged in sewage effluent resulting from the washing of synthetic fiber clothes. DeForges et al. (2014) sampled the Northeast Pacific Ocean in areas in and near the coastal waters of British Columbia, Canada, and found microplastics (those 62–5,000 micrometers in size) were abundant in all samples with elevated concentrations near urban centers; a finding that should be applicable to all urban centers such as those in the TMAA. Besseling et al. (2015) documented the first occurrence of microplastics in the intestines of a humpback whale; while the primary cause of the stranding was not determined, the researchers found multiple types of microplastics ranging in sizes from 1 millimeter to 17 centimeters. There is still a large knowledge gap about the negative effects of microplastics, but it remains a concern (Besseling et al., 2015). Specifically, the propensity of plastics to absorb and concentrate dissolved pollutant chemicals, such as persistent organic pollutants, is a concern because microfauna may be able to digest plastic nanoparticles, facilitating the delivery of dissolved pollutant chemicals across trophic levels and making them bioavailable to larger marine organisms, such as marine mammals (Andrady, 2015; Carlos de Sá et al., 2018; Gallo et al., 2018; Nelms et al., 2018).

#### ***Other Interactions (Including Derelict Fishing Gear)***

Fishery interactions other than bycatch include entanglement from abandoned or partial nets, fishing line, hooks, and the ropes and lines connected to fishing gear (Barcenas De La Cruz et al., 2017; California Coastal Commission, 2018; California Ocean Protection Council & National Oceanic and Atmospheric Administration Marine Debris Program, 2018; Carretta et al., 2019a; Carretta et al., 2019b; Carretta et al., 2020a; Currie et al., 2017b; Delean et al., 2020; Díaz-Torres et al., 2016; Esquible & Atkinson, 2019; Helker et al., 2019; Lowry et al., 2018; National Marine Fisheries Service, 2018c; National Oceanic and Atmospheric Administration, 2016a, 2018e; National Oceanic and Atmospheric Administration Marine Debris Program, 2014a; Polasek et al., 2017; Saez, 2018). The National Oceanic and Atmospheric Administration Marine Debris Program (2014b) reports that abandoned, lost, or otherwise discarded fishing gear constitutes the vast majority of mysticete and pinniped entanglements. For the five-year period between 2012 and 2016 there were 52 known cases of humpback whale entanglement in Alaska (Helker et al., 2019) and between 2013 and 2017 there were an additional 117 cases of reported interactions with fishing gear resulting in serious injuries or mortality off the U.S. West Coast (Carretta et al., 2019a; Carretta et al., 2019b). In the two-year period of 2018–2019, there were 51 confirmed entangled humpback whales along the U.S. West Coast (National Oceanic and Atmospheric Administration, 2020c). Humpback whales from Mexico and Central America have been identified feeding in Alaska (Bettridge et al., 2015; Calambokidis et al., 2008), so some proportion of the entanglements in Alaska could be to whales from the Mexico DPS and Central America DPS individuals. Humpback whales from Mexico and Central America have been identified feeding in Alaska (Bettridge et al., 2015; Calambokidis et al., 2008), so some proportion of the entanglements in Alaska could be to whales from the Mexico DPS and Central America DPS individuals. In total for Alaska between 2012 and 2016, there were 334 fishery-related serious injuries or mortalities (Helker et al., 2019), and for the U.S. West Coast between 2013 and 2017 there were 1,043 cases of fishery-related entanglements (Carretta et al., 2019a). In May 2017, a gray whale calf was discovered dead onshore near the mouth of the Columbia River after becoming entangled in crab pot fishing gear (Cascadia Research, 2017). Outside of U.S. waters, NMFS has identified incidental catches in coastal net fisheries off Japan, Korea, and northeastern Sakhalin Island as a significant threat to endangered Western North Pacific gray whales (Carretta et al., 2020b; Lowry et al., 2018); this species may be seasonally present in the TMAA. Species

of large whales found entangled in 2015 and 2016 included stocks that are present in the TMAA such as ESA-listed humpback, blue, and fin whales and also included gray whale and killer whales, with a total of 133 entanglements to those species in the two-year period (National Marine Fisheries Service, 2018c; National Oceanic and Atmospheric Administration, 2017). In the most recent five-year reporting period for Alaska and the U.S. West Coast, most humpback whale injuries and mortality were from entanglements in fishing gear totaling 169 known occurrences (Carretta et al., 2019a; Helker et al., 2019; National Oceanic and Atmospheric Administration, 2019a). For the identified sources of entanglement in these NMFS reports, none included Navy expended materials.

Along the U.S. West Coast, hook and line fishery and gunshot wounds are two of the primary causes of pinniped serious injuries or mortalities injuries found in strandings (Barcenas De La Cruz et al., 2017; Carretta et al., 2020a; Warlick et al., 2018). Between 2013 and 2017, there were 199 known cases of marine mammals being shot (Carretta et al., 2019a). In December 2018, due to the prevalence of known pinniped shootings, National Oceanic and Atmospheric Administration Fisheries was working on publishing guidelines for fishermen who take actions to deter pinnipeds and other marine mammals from their catch (Esquible & Atkinson, 2019; National Oceanic and Atmospheric Administration, 2018d, 2019c).

In waters off Alaska, Washington, and Southern California, Navy research involving the use of passive acoustic recording devices since 2009 have documented the routine use of non-military explosives at sea (Baumann-Pickering et al., 2013; Bland, 2017; Debich et al., 2014b; Kerosky et al., 2013; Rice et al., 2015; Rice et al., 2018b; Trickey et al., 2015; U.S. Department of the Navy, 2016b; Wiggins et al., 2019; Wiggins et al., 2017). Based on the spectral properties of the recorded sounds and their correspondence with known fishing seasons or activity, the source of these explosions has been linked to the use of explosive marine mammal deterrents, which as a group are commonly known as “seal bombs” (Baumann-Pickering et al., 2013; Wiggins et al., 2019). Seal bombs are intended to be used by commercial fishers to deter marine mammals, particularly pinnipeds, from preying upon their catch and to prevent marine mammals from interacting and potentially becoming entangled with fishing gear (Klint, 2016; National Marine Fisheries Service, 2015; U.S. Department of the Navy, 2016b).

Based on the number of explosions recorded over the past several years in the monitoring within the TMAA, the use of seal bombs is much more prevalent than might be expected by the general public (Rice et al., 2018b; Wiggins et al., 2017). Simonis et al. (2020b) have suggested that the noise from seal bomb use associated with purse seine fishing may be a potential threat to harbor porpoise in the region around the Monterey Bay National Marine Sanctuary. Seal bombs have resulted in at least three known marine mammal injuries in the past (Carretta et al., 2019a), but the seemingly routine use of seal bombs has likely had no significant effect on populations of marine mammals given that it is likely at least some individuals, if not larger groups of marine mammals, have been repeatedly exposed to this explosive stressor.

Since 2010, the Oregon Department of Fish & Wildlife and Washington Department of Fish & Wildlife have conducted a removal program for California sea lions that prey on ESA-listed Chinook salmon and steelhead stocks at Bonneville Dam (Schakner et al., 2016). This is the same population of California sea lions that seasonally inhabit the TMAA, Washington, Oregon, and California waters. Although non-lethal pyrotechnic and rubber buckshot are used as short-term deterrents, in 2016 (for example), these state Fish & Wildlife activities lethally removed (i.e., euthanized) 59 California sea lions (Madson et al., 2017). In December 2018, Congress signed into law the Endangered Salmon Predation Prevention Act, which allows NMFS to authorize the intentional lethal taking of California sea lions on the waters of the

Columbia River and its tributaries for the protection of endangered salmon. In the five-year period from 2013 to 2017, there were 124 pinniped “removals” for that purpose (Carretta et al., 2019a).

### ***Water Quality***

For a general discussion regarding potential impacts on the ocean’s water quality from Military Expended Material (MEM), see Section 3.2 (Expended Materials) of the 2016 GOA Final SEIS/OEIS. Chemical pollution and impacts on ocean water quality are of great concern, although their effects on marine mammals are just starting to be understood (Bachman et al., 2015; Bachman et al., 2014; Cossaboon et al., 2019; Desforges et al., 2016; Foltz et al., 2014; Godard-Coding et al., 2011; Hansen et al., 2015; Jepson & Law, 2016; Law, 2014; Lian et al., 2018; Peterson et al., 2015; Peterson et al., 2014; Ylitalo et al., 2009; Ylitalo et al., 2005). Oil and other chemical spills are a specific type of ocean contamination that can have damaging effects on some marine mammal species directly through exposure to oil or chemicals and indirectly due to pollutants’ impacts on prey and habitat quality (Engelhardt, 1983; Marine Mammal Commission, 2010; Matkin et al., 2008). For example, in the five-year period from 2013 to 2017 along the Pacific coast, there were 127 pinnipeds found stranded with a serious injury or mortality caused by oil or tar coating their body (Carretta et al., 2019a); some of the pinnipeds found seasonally in the TMAA spend part of the year in areas to the south along the Pacific Coast or in islands off that coast.

On a broader scale ocean contamination resulting from chemical pollutants inadvertently introduced into the environment by industrial, urban, and agricultural use is also a concern for marine mammal conservation and has been the subject of numerous studies (Cossaboon et al., 2019; Desforges et al., 2016; Fair et al., 2010; Krahn et al., 2007; Krahn et al., 2009; Moon et al., 2010; Ocean Alliance, 2010). For example, the chemical components of pesticides used on land flow as runoff into the marine environment and can accumulate in the bodies of marine mammals and be transferred to their young through mother’s milk (Fair et al., 2010). The presence of these chemicals in marine mammals has been assumed to put those animals at greater risk for adverse health effects and potential impact on their reproductive success given toxicology studies and results from laboratory animals (Fair et al., 2010; Godard-Coding et al., 2011; Krahn et al., 2007; Krahn et al., 2009; Peterson et al., 2015; Peterson et al., 2014). Desforges et al. (2016) have suggested that exposure to chemical pollutants may act in an additive or synergistic manner with other stressors, resulting in significant population-level consequences. Although the general trend has been a decrease in chemical pollutants in the environment following their regulation, chemical pollutants remain important given their potential to impact marine mammals (Bonito et al., 2016; Jepson & Law, 2016; Law, 2014).

### **Disease and Parasites**

Just as in humans, disease affects marine mammal health and especially older animals. (Pascual, 2015). Occasionally disease epidemics can also injure or kill a large percentage of a marine mammal population (Keck et al., 2010; Paniz-Mondolfi & Sander-Hoffmann, 2009; Simeone et al., 2015). Mass die-offs of some marine mammal species have been linked to toxic algal blooms, which occurs as larger organisms consume multiple prey containing those toxins, thereby accumulating fatal doses (McCabe et al., 2016; National Oceanic and Atmospheric Administration, 2016b). An example is domoic acid poisoning in California sea lions and northern fur seals from the diatom *Pseudo-nitzschia* spp. (Doucette et al., 2006; Fire et al., 2008; Lefebvre et al., 2016; Lefebvre et al., 2010; Torres de la Riva et al., 2009). A comprehensive study in Alaska that sampled over 900 marine mammals across 13 species, including several mysticetes, odontocetes, pinnipeds, and mustelids, found detectable concentrations of domoic



acid in all 13 species and saxitoxin, a toxin absorbed from ingesting dinoflagellates, in 10 of the 13 species (Lefebvre et al., 2016). Algal toxins may have contributed to the stranding and mortality of 34 whales found around the islands in the western GOA and the southern shoreline of the Alaska Peninsula and another 16 stranded whales in British Columbia starting in May 2015–2016 (National Oceanic and Atmospheric Administration, 2016b; Rosen, 2015; Savage et al., 2017; Summers, 2017).

Additionally, all marine mammals have parasites that, under normal circumstances, probably do little overall harm, but under certain conditions can cause serious health problems or even death (Barbieri et al., 2017; Bull et al., 2006; Fauquier et al., 2009; Hawaiian Monk Seal Research Program, 2015; Jepson et al., 2005; Rogers, 2016; Ten Doeschate et al., 2017). The most commonly reported parasitic infections are protozoans in sea otters (Burgess et al., 2018); other parasites known to cause disease in pinnipeds and sea otters include hookworms, lungworms, and thorny-headed worms (Simeone et al., 2015).

### **3.8.2.2 North Pacific Right Whale (*Eubalaena japonica*)**

#### **3.8.2.2.1 Status and Management**

There has been no change in the status or the management of North Pacific right whales since the 2016 GOA Final SEIS/OEIS (Clapham, 2016; Muto et al., 2020; National Marine Fisheries Service, 2013, 2017d; Wade et al., 2010). North Pacific right whales are listed as depleted under the MMPA and endangered under the ESA.

#### **3.8.2.2.2 Abundance**

The most recent estimated population for the North Pacific right whale as presented in the Alaska SAR is between 28 and 31 individuals (Muto et al., 2020). The current abundance in the SAR is an estimated 31 individuals (International Whaling Commission, 2019a). For purposes of the current analysis presented in this SEIS/OEIS, a new estimated North Pacific right whale density was derived in coordination with scientists from the NMFS Southwest Fisheries Science Center and the Alaska Fisheries Science Center. Based on the discussions with these subject matter experts, the Navy has assumed for purposes of acoustic effects modeling that five North Pacific right whales may be present within the TMAA during the 21-day period for the proposed Navy activities. This is a substantial increase in the assumed number of right whales present in comparison to the analysis done for the 2016 GOA Final SEIS/OEIS, but it will provide for a more conservative analysis erring on the side of overestimating potential effects to the species.

#### **3.8.2.2.3 Distribution**

The Navy has determined the North Pacific right whale's occurrence in the TMAA would be year round but rare with a potentially higher density between June and September. Since the 2016 GOA Final SEIS/OEIS there have been a few new sightings or acoustic detections of North Pacific right whales in the Arctic and locations farther south off the U.S. West Coast; off Hokkaido, Japan; and in the North Pacific Ocean to the southeast of Kamchatka Peninsula (Filatova et al., 2019; Hakamada & Matsuoka, 2016; Matsuoka et al., 2018a; Matsuoka et al., 2018b; Rice et al., 2018b; Širović et al., 2015a; U.S. Department of the Navy, 2017d; WorldNow, 2017; Wright et al., 2019; Wright et al., 2018). Right whales were acoustically detected in Barnabus Trough outside the TMAA in 2013, but were not visually observed during the Gulf of Alaska Line-Transect Survey for marine mammals within the TMAA; six of the possible detections shown in Figure 33 of Rone et al. (2014) occurred within the TMAA (Rone et al., 2014). Right whales were again acoustically detected in the same Barnabus Trough area in August of 2015 (Rone et al., 2015). A line transect survey was conducted in 2015 that had a primary focus and design to locate North Pacific right whales in the nearshore waters of the Gulf of Alaska, including the designated critical

habitat located off Kodiak Island, the Biologically Important Area for feeding, right whale habitat based on historical whale catch data, and the nearshore margins of the TMAA (Rone et al., 2017). This survey, which occurred from August 10 to September 8, 2015, had no right whales sighted (Rone et al., 2017). As noted in the 2016 GOA Final SEIS/OEIS, this species has generally been described as routinely observed or acoustically detected in the Bering Sea/Bristol Bay Alaska area (Matsuoka et al., 2018a; Muto et al., 2020). Acoustic monitoring occurring at five sites in the TMAA between July 2011 and September 2017 did not detect any North Pacific right whale calls (Rice et al., 2018b; Wiggins et al., 2017). For additional information about important North Pacific right whale feeding areas in the TMAA, see Section 5.4.1.1 (North Pacific Right Whales) of this SEIS/OEIS.

For additional information about North Pacific right whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.3 Humpback Whale (*Megaptera novaeangliae*)**

#### **3.8.2.3.1 Status and Management**

The status and management of humpback whales that are seasonally present in the TMAA has changed since the 2016 GOA Final SEIS/OEIS as a result of the 14 DPSs established under the ESA in September 2016 (81 FR 62259). Humpback whales in the TMAA are now managed as being from three stocks and three DPSs that are, "... both discrete from other conspecific populations and significant to the species of humpback whales to which they belong" (National Marine Fisheries Service, 2016a). The stock structure of humpback whales is defined by NMFS based on the stock's fidelity to feeding grounds (Gabriele et al., 2017), while the DPSs are based on humpback whales present at known breeding grounds (Bettridge et al., 2015; Carretta et al., 2020b; Darling et al., 2019b; Muto et al., 2020; National Marine Fisheries Service, 2016a). As noted in the 2018 Alaska SAR (Muto et al., 2020), NMFS is in the process of reviewing humpback whale stock structure in light of the 14 DPSs established under the ESA in September 2016 (81 FR 62259). Within the GOA, humpback whales of the Western North Pacific DPS and the Mexico DPS are listed as threatened under the ESA (National Marine Fisheries Service, 2016a). The Hawaii DPS humpback whales, which are the majority of the humpback whale in the TMAA, are no longer listed under ESA.

Humpback whales of the Western North Pacific Stock and DPS are humpback whales that mainly feed in Russian waters but that may also feed in the GOA (Muto et al., 2020; National Marine Fisheries Service, 2016d). This population winters in waters described as Okinawa/Osagawara/Philippines or Western North Pacific (Bettridge et al., 2015), which now also includes the Mariana Islands (Hill et al., 2017; Hill et al., 2016; National Marine Fisheries Service, 2016d; National Oceanic and Atmospheric Administration, 2015a, 2018c; Titova et al., 2017).

The Central North Pacific Stock and Hawaii DPS humpback whales are present in feeding areas off the coast of Alaska (including the nearshore waters of the TMAA), British Columbia, Washington, and Oregon in the summer and then migrate to winter in the Hawaiian Islands (Muto et al., 2020).

A portion of the California, Oregon, Washington stock consisting of the Mexico DPS individuals and the Central North Pacific Stock (the Hawaii DPS) are present in feeding areas off the coast of Alaska (including the nearshore waters of the TMAA), British Columbia, Washington, Oregon, and California in the summer and then return to waters off Mexico and Hawaii in the winter (Bettridge et al., 2015; Calambokidis et al., 2017a; Carretta et al., 2020b; Muto et al., 2020; National Marine Fisheries Service, 2016d, 2016e; Wade et al., 2016).

On October 9, 2019, NMFS proposed to designate critical habitat for the endangered Western North Pacific DPS, the threatened Mexico DPS, and the endangered Central America DPS of humpback whales along the coasts of California, Oregon, Washington, and Alaska (84 FR 54354; note that whales belonging to the Central America DPS should not be present in the GOA or the TMAA according to NMFS (Mate et al., 2018; National Marine Fisheries Service, 2016d, 2019b, 2019c). As shown on Figure 3.8-2 the northwestern portion of the TMAA over the continental shelf overlaps with approximately 29,222 square kilometers of the proposed areas in the NMFS designated Region/Units 5, 7, and 8.

Region/Unit 5 is “occupied critical habitat” for the Western North Pacific DPS and characterized as having a high conservation value (National Marine Fisheries Service, 2019b, 2019c). The National Marine Fisheries Service has determined that Region/Unit 7 (named the “Kenai Peninsula Area” by NMFS ) has low conservation value and concluded that exclusion of that and other similar areas from the designations will not result in extinction of any humpback whale DPSs (84 FR 54378), given the whales are not expected to rely on the area for feeding (National Marine Fisheries Service, 2019b, 2019c). Region/Unit 8 (the “Prince William Sound Area”) was also determined to have a low conservation value and “limited conservation benefit” for the Western North Pacific DPS, and the area was excluded because “... whales from the WNP DPS have not been directly observed ...” in Region/Unit 8 (National Marine Fisheries Service, 2019b, 2019c). Region/Unit 8 was determined to have a high conservation value as critical habitat for the threatened Mexico DPS humpback whales (84 FR 54378). Sighting data from three line transect surveys (in the summers of 2009, 2013, and 2015) that included Region/Unit 8 of the Critical Habitat had no sightings of humpback whales in any of the survey years in that portion of the Critical Habitat overlapping with the TMAA (see Rone et al. (2017)).

In the proposed rule to designate humpback whale Critical Habitat (84 FR 54354) (shown in Figure 3.8-2), NMFS has identified one essential feature of that habitat, but that essential feature is a composite of three factors defined as (1) sufficient quality, (2) abundance, and (3) accessibility of prey species within humpback whale feeding areas to support population growth of the ESA-listed humpback whale DPSs. Prey species identified by NMFS are primarily zooplankton/krill (euphausiids) and small pelagic schooling fishes of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth (84 FR 54354). In Alaska waters, humpback whales feed in association with high densities of zooplankton and fish near the Kodiak Archipelago (Witteveen et al., 2014; Witteveen & Wynne, 2017) and in associated with seasonal runs of herring in Prince William Sound (Moran et al., 2015).

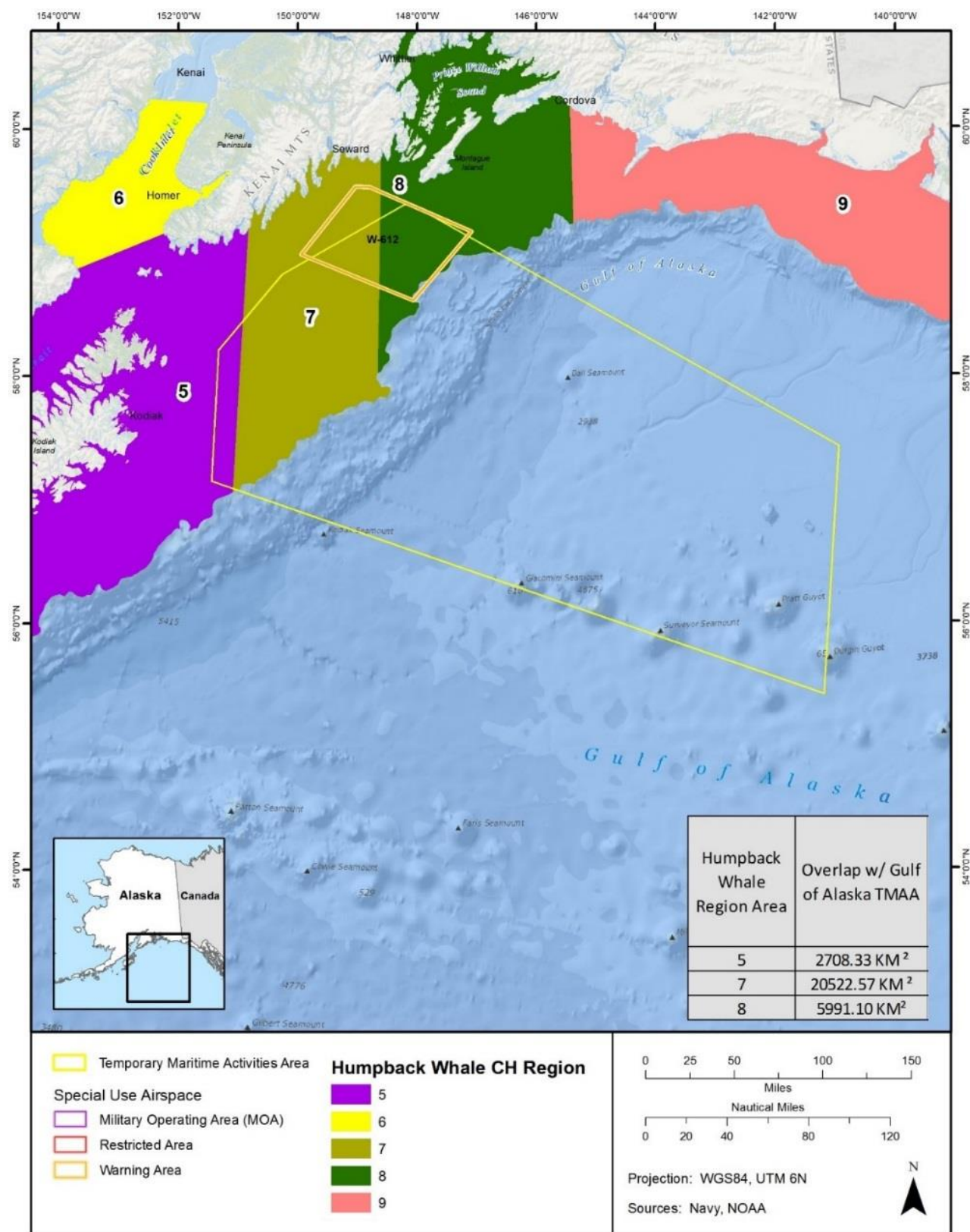


Figure 3.8-2: Proposed Humpback Whale Critical Habitat Overlapping the TMAA

### 3.8.2.3.2 Abundance

For the Western North Pacific stock and DPS, photographic identifications off Okinawa and Ogasawara were used to estimate that the abundance of humpback whales in the Western North Pacific population was approximately 1,000 individuals (Bettridge et al., 2015; Calambokidis, 2009; Muto et al., 2017). The inclusion of more recent data from photographic identifications off Okinawa have documented the presence of at least 1,402 unique individuals in the Western North Pacific DPS (Kobayashi et al., 2016). The 2018 Alaska SAR provides that it is reasonable to assume that the growth rate for this stock would be at least 7 percent annual rate of increase based on the other observations from the North Pacific (Muto et al., 2020); the most recent Alaska SAR provides that the rate of increase is unknown given the age of the data used in the previous assessment (Muto et al., 2020).

For the Central North Pacific stock and the Hawaii DPS portion of the humpback whale population in the TMAA, the Hawaii DPS was delisted under the ESA given that this population segment is believed to have fully recovered and now has an abundance greater than the pre-whaling estimate (Barlow et al., 2011; Bettridge et al., 2015; Muto et al., 2017; Muto et al., 2018b; Muto et al., 2020; National Marine Fisheries Service, 2016a; Wade et al., 2016).

For the California, Oregon, and Washington stock, data from the most recently published NMFS survey (in 2014) (Barlow, 2016) and other corresponding investigations (Calambokidis & Barlow, 2020; Calambokidis et al., 2017a; Henry et al., 2020; Smultea, 2014) appear consistent with the highest-yet abundance estimates of humpback whales along the U.S. West Coast (Carretta et al., 2020b). The new best overall estimate of abundance of humpback whales along the U.S. West Coast has been provided by photo identification data gathered between 2015 and 2018 along the U.S. West Coast (Calambokidis & Barlow, 2020). This estimate, which includes the Mexico DPS and the Central America DPS ( $n=4,973$ ;  $CV=0.05$ ), is significantly higher than the abundance ( $n=2,900$ ) presented in the 2019 Pacific SAR (Carretta et al., 2020b). This increase in the California, Oregon, Washington stock is estimated to have been between 7.5 and 8.2 percent, since the late 1980s, to have reached that new reported higher abundance (Calambokidis & Barlow, 2020).

The humpback whales in Glacier Bay and Icy Strait are potentially from all three stocks, and data collected from 1985 to 2014 found an increase in the number of individual whales counted averaging 5.1 percent per year with an accelerated rate of growth from 2002 to 2011 of 11.1 percent per year (Gabriele et al., 2017).

### 3.8.2.3.3 Distribution

There have been no changes to the general known distribution of humpback whales in the TMAA since the 2016 GOA Final SEIS/OEIS, however there has been new research relevant to the presence of humpbacks in the TMAA and GOA region. Consistent with the information presented in the 2016 GOA Final SEIS/OEIS, humpback whale typically are present in higher numbers during the summer in high-latitude nearshore feeding grounds (Barlow et al., 2011; Becker et al., 2016; Becker et al., 2017; Bettridge et al., 2015; Calambokidis et al., 2017a; Calambokidis et al., 2010; Keen et al., 2018; Pack et al., 2017; Wade et al., 2016). Migrations are variable and dynamic seasonally and timing of migration may change from year to year based on nutritional needs, oceanic conditions impacting the prey base, and competition for food between species of whales (Burrows et al., 2016; Gabriele et al., 2017; Moran et al., 2018). These factors can result in humpback whales lengthening their feeding time in northern latitudes, skipping the annual migration, and potentially increasing their predation on herring in the GOA (Straley et al., 2017). Passive acoustic monitoring (Debich et al., 2013; Debich et al., 2014a; Rice et

al., 2015; Rice et al., 2018b) have documented the presence of humpback whales year round in the TMAA, although they have been fewer in number based on three line transect surveys of the TMAA and surrounding waters (Rone et al., 2009; Rone et al., 2014; Rone et al., 2017) and the locations and destinations of satellite tagged humpback whales, as reported in Mate et al. (2018) and Palacios et al. (2020). This corresponds to the distribution of their prey, which is primarily concentrated on the shelf over shallow banks less than 100 meters (m) in depth (Burrows et al., 2016; Matta & Baker, 2020; McGowan et al., 2019; Moran et al., 2015; Straley et al., 2017).

Specific to the Western North Pacific humpback whale DPS designated since the 2016 GOA Final SEIS/OEIS, individuals in the Western North Pacific DPS mainly feed in Russian waters, but may also feed in the GOA, including the TMAA (Muto et al., 2020; National Marine Fisheries Service, 2016d). This population winters in waters described as Okinawa/Osagawara/Philippines or Western North Pacific (Bettridge et al., 2015), which now also includes the Mariana Islands (Hill et al., 2017; Hill et al., 2016; National Marine Fisheries Service, 2016d; National Oceanic and Atmospheric Administration, 2015a, 2018c; Titova et al., 2017). Completed analyses of genetic samples to date have found humpback whales in the Mariana Islands share four haplotypes common in humpback whales throughout the North Pacific and two haplotypes that are more common in Western North Pacific DPS whales, but which are also present in humpback whales throughout the North Pacific (Hill et al., 2018). This genetic data as well as early photo-identification data from Darling et al. (1996) and more recent data regarding the analysis of humpback vocalizations suggest mixing of the humpback whale populations throughout the Pacific (Darling et al., 2019a).

The Hawaii DPS humpback whales are present in feeding areas off the coast of Alaska (including the nearshore waters of the TMAA), British Columbia, Washington, and Oregon in the summer and then migrate to winter in the Hawaiian Islands (Muto et al., 2020; National Marine Fisheries Service, 2016d, 2016e).

The Mexico DPS individuals are also present in feeding areas off the coast of Alaska (including the nearshore waters of the TMAA), British Columbia, Washington, Oregon, and California in the summer and then return to waters off Mexico in the winter (Bettridge et al., 2015; Calambokidis et al., 2017a; Carretta et al., 2020b; Muto et al., 2020; National Marine Fisheries Service, 2016d, 2016e; Wade et al., 2016). For additional information about important humpback whale feeding areas in the TMAA, see Section 5.4.1.2 (Humpback Whales) of this SEIS/OEIS.

For additional information about humpback whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

#### **3.8.2.4 Blue Whale (*Balaenoptera musculus*)**

##### **3.8.2.4.1 Status and Management**

There has been no change in the status or the management of the two blue whale stocks as designated by NMFS for the TMAA since the 2016 GOA Final SEIS/OEIS. The blue whale is listed as endangered under the ESA and as depleted under the MMPA throughout its range, but there is no designated critical habitat for this species (Carretta et al., 2020b; Muto et al., 2020; National Marine Fisheries Service, 2018b). The National Marine Fisheries Service has determined that for blue whales with regards to critical habitat, more research is needed to rigorously and specifically define the environmental features that make an area biologically important to blue whales (National Marine Fisheries Service, 2018b). Blue whale subspecific taxonomy and population structure has not been fully resolved and is an area of active

research (International Whaling Commission, 2019b; National Marine Fisheries Service, 2018b). The number of blue whales in the population that inhabits the TMAA is complicated by there being uncertainty regarding the number of populations of blue whale in the Pacific, one to possibly three populations (Carretta et al., 2020b; International Whaling Commission, 2019b; Monnahan et al., 2015; National Marine Fisheries Service, 2018b). NMFS currently has designated two stock management units in the North Pacific, one for waters around Hawaii (the Central North Pacific stock) and one for the “U.S. West Coast” (the Eastern North Pacific stock), but with a description for the range for both stocks that includes Alaska waters (Carretta et al., 2020b; National Marine Fisheries Service, 2018b); blue whales in Alaska waters are not addressed in the Alaska SAR (Muto et al., 2020).

#### **3.8.2.4.2 Abundance**

Since the 2016 GOA Final SEIS/OEIS, multiple lines of evidence suggest that blue whales in the Pacific may have recovered and been at a stable level based on surveys and scientific findings (Barlow, 2016; Campbell et al., 2015; Carretta et al., 2020b; Carretta et al., 2015; International Whaling Commission, 2016, 2019b; Monnahan, 2013; Monnahan & Branch, 2015; Monnahan et al., 2015; Monnahan et al., 2014; National Marine Fisheries Service, 2018b; Rockwood et al., 2017; Širović et al., 2015b; Valdivia et al., 2019). The new best overall estimate of abundance of blue whales along the U.S. West Coast ( $n=1,898$ ;  $CV=0.08$ ) has been provided by photo identification data gathered between 2015 and 2018 (Calambokidis & Barlow, 2020). This estimate is higher than the abundance ( $n=1,496$ ) in the 2019 Pacific SAR (Carretta et al., 2020b) and suggests an increase in the abundance since the 1990s (Calambokidis & Barlow, 2020).

#### **3.8.2.4.3 Distribution**

There have been no changes to the known distribution of blue whales in the TMAA since the 2016 GOA Final SEIS/OEIS. There have not been a sufficient number of surveys in Alaska waters to support the type of habitat models that have been used to predict the species distribution elsewhere (Abrahms et al., 2019a; Becker et al., 2018; Becker et al., 2017; Carretta et al., 2020b; Forney et al., 2015; Redfern et al., 2017b). The Eastern North Pacific stock of blue whales range from the GOA to as far south as the waters off Costa Rica (Carretta et al., 2020b). Blue whales in the Central North Pacific Stock have been observed in the limited surveys of the U.S. EEZ around Hawaii (Carretta et al., 2020b; National Marine Fisheries Service, 2018b) and acoustically detected at Saipan and Tinian in the Mariana Islands (Oleson et al., 2015), but this reflects very limited survey coverage of the Central Pacific. There are no data suggesting or reason to believe that the two stocks do not overlap in their distribution when in Alaska waters.

Based on passive acoustic data, the Navy has determined the blue whale’s occurrence in the TMAA is year round with the highest numbers present in June to December (Debich et al., 2013; Debich et al., 2014a; Rice et al., 2015; Rice et al., 2018b). This is consistent with the conservative approach to the analysis provided in the 2016 GOA Final SEIS/OEIS and the analysis in this document, in which Navy assumed the species would be present during the Proposed Action.

For additional information about blue whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.5 Fin Whale (*Balaenoptera physalus*)**

#### **3.8.2.5.1 Status and Management**

There has been no change in the status or the management of fin whales since the 2016 GOA Final SEIS/OEIS. The fin whale is listed as depleted under the MMPA and endangered under the ESA throughout its range, but there is no designated critical habitat for this species in the Pacific (Carretta et al., 2020b; Muto et al., 2020; National Marine Fisheries Service, 2010).

#### **3.8.2.5.2 Abundance**

The National Marine Fisheries Service has determined there are no reliable estimates of current and historical abundances for the entire Northeast Pacific fin whale stock (Muto et al., 2020). In areas of the Pacific where research has occurred, various efforts and methodologies have indicated increases in the number of fin whales (Barlow, 2016; Širović et al., 2015b; Towers et al., 2018; Valdivia et al., 2019). These findings and the trend for an increase in population appear consistent with the highest-yet abundances of fin whales in the 2014 NMFS survey of the U.S. West Coast (Barlow, 2016).

#### **3.8.2.5.3 Distribution**

The Navy has determined the fin whale's occurrence in the TMAA would be likely year round with a potential for higher numbers between June and August. There have been no changes to the known distribution of fin whales in the TMAA since the 2016 GOA Final SEIS/OEIS, although there is evidence suggesting general connectivity among fin whales in the Pacific (Archer et al., 2019). Fin whales were found to feed in association with high density of zooplankton near the Kodiak Archipelago (Witteveen et al., 2014).

Passive acoustic monitoring between May and September in 2015 and 2017 detected only low numbers of fin whale vocalizations in the TMAA (Rice et al., 2018b; Wiggins & Hildebrand, 2018) or in the western GOA in the spring and fall (Archer et al., 2019). These acoustic data are not reflective of the survey data indicative of greater numbers (Rone et al., 2017), which was factored into the derivation of fin whale densities in the TMAA and consistent with the analysis provided in the 2016 GOA Final SEIS/OEIS and the analysis in this document.

For additional information about fin whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.6 Sei Whale (*Balaenoptera borealis*)**

#### **3.8.2.6.1 Status and Management**

There has been no change in the status or the management of sei whales since the 2016 GOA Final SEIS/OEIS. The Eastern North Pacific stock includes animals found within the U.S. West Coast EEZ and in adjacent high seas waters (Carretta et al., 2020b). The sei whale is listed as endangered under the ESA and as depleted under the MMPA throughout its range (National Marine Fisheries Service, 2011). Analysis of samples from sei whales in the Pacific by Huijser et al. (2018) did not identify significant levels of genetic structure or find support for the current stock management designations in the Pacific; there have been arguments made for a single stock of sei whales in the Pacific (International Whaling Commission, 2019b).



#### **3.8.2.6.2 Abundance**

Since the 2016 GOA Final SEIS/OEIS, there has been an estimate published that provides an abundance for sei whales in the North Pacific (Hakamada et al., 2017). Line transect surveys were conducted between 2010 and 2012 in the Pacific from 40° north latitude northward to the Aleutian Islands and eastward into the GOA provided the data used in that abundance estimate (n=29,632; Coefficient of Variation=0.242) (Hakamada et al., 2017). Based on that estimate, a revised density for sei whales in the TMAA has been incorporated into the new analysis presented in this document. This is consistent with survey results indicating that sei whales have increased in number off the U.S. West Coast (Barlow, 2016) and in the Pacific (Valdivia et al., 2019).

#### **3.8.2.6.3 Distribution**

The Navy has determined the sei whale's occurrence in the TMAA would be year round but rare. There have been no changes to the known distribution of sei whales in the TMAA since the 2016 GOA Final SEIS/OEIS. As was noted in the 2016 GOA Final SEIS/OEIS, whaling records documented high densities of sei whales in the northwestern and northeastern portions of the GOA (i.e., near Portlock Bank). The only confirmed sightings of sei whales in the GOA (and outside the TMAA) in modern times were in 2011 to the west of Kodiak Island (Davis et al., 2011), and two sightings in 2015; a sei whale within the aggregation of fin and humpback whales at Albatross Bank off Kodiak Island and a second observed approximately 300 km south of Kodiak Island (Rone et al., 2017). Although recent surveys (2009, 2013, 2015) have not produced confirmed sei whale sightings in the TMAA and passive acoustic monitoring at fixed sites has not detected their vocalizations (Rice et al., 2020), sei whale calls were acoustically detected in the TMAA during the 2013 survey (Rone et al., 2014). Based on the above considerations, sei whale occurrence in the TMAA during the summer time period is considered rare.

For additional information about sei whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

#### **3.8.2.7 Minke Whale (*Balaenoptera acutorostrata*)**

##### **3.8.2.7.1 Status and Management**

There has been no change in the status or the management of minke whales since the 2016 GOA Final SEIS/OEIS. The minke whale is not listed under the ESA. The stock structure for minke whales remains uncertain in the Pacific, and minke whales in the TMAA are considered the Alaska stock in the current SAR (Muto et al., 2020).

##### **3.8.2.7.2 Abundance**

There are no data on population trends for minke whales in the GOA, given that so few minke whales have been seen during surveys in the area (Muto et al., 2020; Rone et al., 2017).

##### **3.8.2.7.3 Distribution**

The Navy has determined the minke whale's occurrence in the TMAA would be likely year round. There have been no changes to the known distribution of minke whales in the TMAA since the 2016 GOA Final SEIS/OEIS.

For additional information about minke whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.8 Gray Whale (*Eschrichtius robustus*)**

#### **3.8.2.8.1 Status and Management**

There has been no change in the status or the management of gray whales since the 2016 GOA Final SEIS/OEIS. There are two North Pacific populations of gray whales present in the TMAA: the Western subpopulation and the Eastern subpopulation (Carretta et al., 2020b; Cooke, 2019a; Cooke, 2019b). The current stock structure for gray whales in the Pacific has been in the process of being re-examined for a number of years (see for example, Brüniche-Olsen et al. (2018)) and remains uncertain as of the most recent (2020) Pacific SAR (Carretta et al., 2020b); gray whales are not addressed in the Alaska SAR (Muto et al., 2020).

The Western North Pacific gray whale DPS is listed as endangered, and there has been no designated critical habitat for this species; the Eastern North Pacific DPS recovered from whaling exploitation, was delisted under the ESA in 1994, and is not considered depleted (Carretta et al., 2020b).

There are also a few hundred gray whales that feed along the Pacific coast as far north as Kodiak Island (Gosho et al., 2011) and as far south as Northern California throughout the summer and fall that are known as the Pacific Coast Feeding Group (Calambokidis et al., 2002; Calambokidis et al., 2017b; Carretta et al., 2017b; Mate et al., 2013; Weller et al., 2013). Photo-identification, telemetry, and genetic studies suggest that the Pacific Coast Feeding Group is demographically distinct from the Eastern North Pacific population (Calambokidis et al., 2017b; Calambokidis et al., 2010; Frasier et al., 2011; Lagerquist et al., 2018; Mate et al., 2010), but the Pacific Coast Feeding Group is not currently managed as a distinct stock in NMFS SARs (Carretta et al., 2020b).

#### **3.8.2.8.2 Abundance**

Recent analysis of the data available for 2005 through 2016 estimate the combined Sakhalin Island and Kamchatka populations that are part of the Western North Pacific stock are increasing in number (Bröker et al., 2020; Carretta et al., 2020b; Cooke, 2019a; Cooke, 2019b; Moore & Weller, 2018; Nakamura et al., 2017a; Nakamura et al., 2017b). Findings from Valdivia et al. (2019) indicate an average growth rate of 6.22 percent for the DPS overall. The combined Sakhalin Island and Kamchatka populations are estimated to be increasing from 2005 through 2016 at an average rate between 2 and 5 percent annually (Cooke, 2019a; Cooke, 2019b; Cooke et al., 2015). A recent increase in the occurrence of gray whales off Japan (Nakamura et al., 2017a), is also consistent with a positive population growth for Western North Pacific gray whales.

The eastern population has increased over several decades despite the 1999 and 2000 UMEs in which an unusually large number of gray whales stranded along the coast, from Mexico to Alaska (Gulland et al., 2005), when many scientists thought the population had reached “carrying capacity” (Carretta et al., 2018a; Carretta et al., 2017b; Durban et al., 2016). Starting in January of 2019, an elevated number of gray whale strandings occurred along the west coast of North America from Mexico through Alaska, which prompted NMFS to declare those strandings a UME (National Marine Fisheries Service, 2019a; National Oceanic and Atmospheric Administration, 2020b). As of February 2020, the strandings totaled 236 known individuals along their migratory corridor (National Oceanic and Atmospheric Administration, 2020b). Preliminary findings for several of the whales indicated signs of emaciation, although the findings were not consistent across the subset of the whales examined, and additional future research will be required to better identify factors resulting in the UME (National Oceanic and Atmospheric Administration, 2020b). Although the future trend for this population may be affected by the previously

mentioned 2019 UME, as of August 4, 2020, there have been 32 strandings in Alaska, and 63 total on the U.S. West Coast (National Oceanic and Atmospheric Administration, 2020a).

#### **3.8.2.8.3 Distribution**

The Navy has determined the gray whale's occurrence in the TMAA would be seasonal with their highest likelihood of occurring being between June and August. There have been no changes to the known distribution of gray whales in the TMAA since the 2016 GOA Final SEIS/OEIS. Consistent with results from their expected distribution, gray whale call detections are most common on the continental shelf (Rice et al., 2015; Rice et al., 2018b; Wiggins et al., 2017). For additional information about important gray whale migration areas in the TMAA, see Section 5.4.1.3 (Gray Whales) of this SEIS/OEIS. Both the Western subpopulation and the Eastern subpopulation are expected to migrate through the Gulf of Alaska, for example, as of 2013 there were 23 known cases of Western North Pacific DPS gray whales being identified along the coasts of Canada and the U.S., including 14 as far south as off Mexico (Mate et al., 2015; Moore & Weller, 2018; Weller et al., 2013; Weller et al., 2012).

For additional information about gray whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

#### **3.8.2.9 Sperm Whale (*Physeter macrocephalus*)**

##### **3.8.2.9.1 Status and Management**

There has been no change in the status or the management of sperm whales since the 2016 GOA Final SEIS/OEIS. The sperm whale has been listed as endangered since 1970 under the precursor to the ESA (National Marine Fisheries Service, 2009), and is depleted under the MMPA throughout its range, but there is no designated critical habitat for this species in the North Pacific.

##### **3.8.2.9.2 Abundance**

Sperm whale population abundance and trends based on line-transect surveys conducted off the U.S. West Coast from 1991 to 2014 include a high level of uncertainty but indicate that sperm whale abundance has appeared stable (Carretta et al., 2020b; Moore & Barlow, 2017; Moore & Barlow, 2014).

##### **3.8.2.9.3 Distribution**

The Navy has determined the sperm whale's occurrence in the TMAA would be likely year round in waters greater than 1,000 m and most often in waters greater than 2,000 m. A study found that although they are present year round in the GOA, they are potentially present in greater numbers between June and September due to higher numbers of acoustic detections (Diogou et al., 2019). There have been no changes to the known distribution of sperm whales in the TMAA since the 2016 GOA Final SEIS/OEIS. Sperm whale are somewhat migratory, and passive acoustic monitoring at five sites in the TMAA recorded sperm whale clicks throughout each summer between May and September in 2015 and 2017 at all sites, but were most common at the shelf break and offshore (Rice et al., 2018b).

For additional information about sperm whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.10 Killer Whale (*Orcinus orca*)**

#### **3.8.2.10.1 Status and Management**

There has been no change in the status or the management of killer whales since the 2016 GOA Final SEIS/OEIS. Killer whales likely present in the TMAA are not species listed under the ESA.

Four killer whale stocks are likely to be present in the TMAA. These stocks include (1) the Eastern North Pacific Alaska Resident stock; (2) the West Coast Transient stock; (3) the Eastern North Pacific GOA, Aleutian Islands, and Bering Sea Transient stock; and (4) the Eastern North Pacific Offshore stock (Carretta et al., 2020b; Muto et al., 2020). Preliminary genetic data for killer whales Alaska indicate that the current stock structure of killer whales in Alaska needs revision, but this revision is awaiting completion of a stock structure evaluation before any new stocks are identified (Muto et al., 2020).

#### **3.8.2.10.2 Abundance**

The Eastern North Pacific Alaska Resident stock of killer whales continues to increase in abundance by about 3 percent per year (GulfWatch Alaska, 2019; Matkin et al., 2018). No data are available on current population trends for the West Coast Transient stock or the Eastern North Pacific GOA, Aleutian Islands, and Bering Sea Transient stocks of killer whales (Carretta et al., 2020b; Muto et al., 2020). The National Marine Fisheries Service considers the population trajectory for Eastern North Pacific Offshore killer whales to be stable (Carretta et al., 2020b).

#### **3.8.2.10.3 Distribution**

The Navy has determined that the killer whale's occurrence in the TMAA would be likely year round. Based on data from Olsen et al. (2018), the Alaska Resident killer whales seasonally follow herring and salmon inshore during the summer runs of those species (Matkin et al., 2018). Transient killer whales have been sighted off of Alaska, British Columbia, and Washington State (Towers et al., 2012). As a clarification from the 2016 GOA Final SEIS/OEIS, all killer whale ecotypes may be present but the one offshore population and the two transient types are more likely in the majority of the TMAA given the Navy training area mostly consists of deep ocean far offshore.

For additional information about killer whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.11 Pacific White-Sided Dolphin (*Lagenorhynchus obliquidens*)**

#### **3.8.2.11.1 Status and Management**

There has been no change in the status or the management of Pacific white-sided dolphins since the 2016 GOA Final SEIS/OEIS. This species is not listed under the ESA. The National Marine Fisheries Service recognizes a single stock for the U.S. West Coast—the California, Oregon, and Washington stock (Carretta et al., 2020b).

#### **3.8.2.11.2 Abundance**

No data are available on current population trends for Pacific white-sided dolphins present in the TMAA (Muto et al., 2020). As a clarification from the 2016 GOA Final SEIS/OEIS and as noted in the 2018 Alaska SAR, the population of Pacific white-sided dolphins in the North Pacific Ocean was last estimated (in 1993) to number approximately 931,000 dolphins, but the subset number of those dolphins in North Pacific stock as managed by NMFS has been given as 26,880 dolphins (Muto et al., 2020).

### **3.8.2.11.3 Distribution**

The Navy has determined the Pacific white-sided dolphins' occurrence in the TMAA would be likely year round. There have been no changes to the known distribution of Pacific white-sided dolphins in the TMAA since the 2016 GOA Final SEIS/OEIS.

For additional information about Pacific white-sided dolphins occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.12 Harbor Porpoise (*Phocoena phocoena*)**

#### **3.8.2.12.1 Status and Management**

There has been no change in the status or the management of harbor porpoise since the 2016 GOA Final SEIS/OEIS. This species is not listed under the ESA. The stocks of harbor porpoise present in Alaska waters near the TMAA are not considered depleted under the MMPA.

#### **3.8.2.12.2 Abundance**

No data are available regarding population trends for the stock of harbor porpoises in the area given the last comprehensive survey of their habitat in and adjacent to the GOA occurred in 1998 (Muto et al., 2020).

#### **3.8.2.12.3 Distribution**

The Navy has determined the harbor porpoise's occurrence in the TMAA would be likely year round in the nearshore locations to the shelf break. There have been no changes to the known distribution of harbor porpoise in the TMAA since the 2016 GOA Final SEIS/OEIS.

For additional information about harbor porpoise occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.13 Dall's Porpoise (*Phocoenoides dalli*)**

#### **3.8.2.13.1 Status and Management**

There has been no change in the status or the management of Dall's porpoise since the 2016 GOA Final SEIS/OEIS. This species is not listed under the ESA (Muto et al., 2020).

#### **3.8.2.13.2 Abundance**

No data are available regarding population trends for the Alaska stock of Dall's porpoises, given the last comprehensive survey of their habitat in and adjacent to the GOA occurred in 1991 (Muto et al., 2020). Density estimates derived from line-transect survey data collected in and near the TMAA (Rone et al., 2017) were used in the analyses.

#### **3.8.2.13.3 Distribution**

The Navy has determined the Dall's porpoise's occurrence in the TMAA would be likely year round. There have been no changes to the known distribution of Dall's porpoise in the TMAA since the 2016 GOA Final SEIS/OEIS.

For additional information about Dall's porpoise occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.14 Cuvier's Beaked Whale (*Ziphius cavirostris*)**

#### **3.8.2.14.1 Status and Management**

There has been no change in the status or the management of the Alaska stock of Cuvier's beaked whales since the 2016 GOA Final SEIS/OEIS (Muto et al., 2020).

#### **3.8.2.14.2 Abundance**

No data are available regarding population trends for the stock of Cuvier's beaked whales in the TMAA (Muto et al., 2020).

#### **3.8.2.14.3 Distribution**

The Navy has determined the Cuvier's beaked whale's occurrence in the TMAA would be likely year round. Passive acoustic monitoring at five sites in the TMAA occurred between May and September in 2015 and April and September in 2017 (Rice et al., 2018b). That monitoring detected Cuvier's beaked whales most commonly in spring at the monitoring site located in the approximate middle of the TMAA (Site "AB"), while no detections occurred in the summer time period between July and September (Rice et al., 2018b). Cuvier's beaked whales were detected only once in 2015 at seamounts site (Site "QN") and were never detected at the monitoring site located at the shelf break (Site "CB") (Rice et al., 2018b). Although there are no data from the GOA, acoustic sampling of bathymetrically featureless areas off Southern California detected many beaked whales over an abyssal plain, which counters a common misperception that beaked whales are primarily found over slope waters, in deep basins, or over seamounts (Griffiths & Barlow, 2016); this is consistent with the 2015 and 2017 acoustic monitoring (Rice et al., 2018b). Research involving tagged Cuvier's beaked whales in Southern California has documented movements in excess of hundreds of kilometers. Schorr et al. (2014) reported that five out of eight tagged whales journeyed approximately 250 km from their tag deployment location, and one of these five made an extra-regional excursion over 450 km to the south to Mexico and back (Falcone & Schorr, 2011, 2012, 2013, 2014; Falcone et al., 2009).

For additional information about Cuvier's beaked whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.15 Baird's Beaked Whale (*Berardius bairdii*)**

#### **3.8.2.15.1 Status and Management**

There has been no change in the status or the management of Baird's beaked whale since the 2016 GOA Final SEIS/OEIS (Muto et al., 2020). The Alaska stock of Baird's beaked whales is not listed under the ESA (Muto et al., 2020).

#### **3.8.2.15.2 Abundance**

As was the case in for 2016 GOA Final SEIS/OEIS, there are no abundance or population trend data for the Alaska stock of Baird's beaked whale (Muto et al., 2020).

#### **3.8.2.15.3 Distribution**

The Navy has determined the Baird's beaked whale's occurrence in the TMAA would be likely year round. There have been no changes to the known distribution of Baird's beaked whales in the TMAA since the 2016 GOA Final SEIS/OEIS. Data from a satellite-tagged Baird's beaked whale off Southern California recently documented movement north along the shelf-edge for more than 400 NM over a six-and-a-half day period (Schorr et. al., Unpublished). If that one sample involving a 400 NM excursion is

reflective of more general behavior, Baird's beaked whales present in the TMAA may have much larger home ranges than the waters bounded by the TMAA.

For additional information about Baird's beaked whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.16 Stejneger's Beaked Whale (*Mesoplodon stejnegeri*)**

#### **3.8.2.16.1 Status and Management**

There has been no change in the status or the management of Stejneger's beaked whales since the 2016 GOA Final SEIS/OEIS. Stejneger's beaked whale is not listed under the ESA, and the Alaska stock is not a depleted stock (Muto et al., 2020).

#### **3.8.2.16.2 Abundance**

There have been no new data regarding the number of Stejneger's beaked whales present in the TMAA since the 2016 GOA Final SEIS/OEIS. As was the case in for 2016 GOA Final SEIS/OEIS, reliable estimates of abundance for this stock are currently unavailable (Muto et al., 2020).

#### **3.8.2.16.3 Distribution**

The Navy has determined the Stejneger's beaked whale's occurrence in the TMAA would be likely year round. There have been no changes to the known distribution of Stejneger's beaked whales in the TMAA since the 2016 GOA Final SEIS/OEIS. Stejneger's beaked whale vocalizations have been detected by passive acoustic monitoring sites at the shelfbreak and offshore in the TMAA (Rice et al., 2018b).

For additional information about Stejneger's beaked whale occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.17 Steller sea lion (*Eumetopias jubatus*)**

#### **3.8.2.17.1 Status and Management**

There has been no change in the status or the management of Steller sea lion stocks since the 2016 GOA Final SEIS/OEIS. The National Marine Fisheries Service has designated two Steller sea lion stocks in the North Pacific corresponding to two DPSs with the same names (Muto et al., 2020); both populations are potentially present within the TMAA. The Western U.S. stock (or DPS) consists of sea lions occurring west of 144°W longitude, and the Eastern U.S. stock (or DPS) is defined as the population occurring east of 144°W longitude (Muto et al., 2020). The Western U.S. stock is listed as depleted under the MMPA and endangered under the ESA. Critical habitat for the Western DPS was designated by NMFS in 1993 (58 FR 45269) and includes a 20 NM buffer around all major haulouts and rookeries, as well as associated terrestrial, air, and aquatic zones, and three large offshore foraging areas that are all in Alaska waters. As described in Section 5.4.1.4 (Steller Sea Lions) and Section 5.4 (Geographic Mitigation to be Implemented), the TMAA boundaries will continue to be located outside of the Steller sea lion critical habitat designated by NMFS in 1993 (58 FR 45269).

The Eastern U.S. stock (or DPS) of Steller sea lions is currently listed as depleted under the MMPA and in recognition of their recovery, Steller sea lions in the Eastern U.S. stock were removed from the List of Endangered and Threatened Wildlife in October 2013 (Muto et al., 2020; National Marine Fisheries Service, 2016f).

### 3.8.2.17.2 Abundance

Using data collected from 1978 through 2017, there are strong evidence for positive trends in pup and non-pup counts of western stock Steller sea lions in the GOA (Fritz et al., 2015; Muto et al., 2020; Sweeney et al., 2018). In the central and eastern GOA, pup counts declined sharply between 2015 and 2017, which may have been due to changes in availability of prey associated with warm ocean temperatures that occurred in the northern GOA from 2014 to 2016. No new data were collected for the GOA region in the 2018 survey, but the 2019 survey focused on the GOA and should contain more precise and accurate estimates of counts and trends for this species in the GOA (Muto et al., 2020).

### 3.8.2.17.3 Distribution

Steller sea lions from the Western DPS are likely to occur year round in the inshore portion of the TMAA. Unpublished data from the Alaska Department of Fish and Game show tagged female Steller sea lions repeatedly traveling from haulout sites to the shelf break (approximated as the 500 m isobath) to forage but not venturing off the shelf. Very little data exist on the offshore movements of male Steller sea lions.

Steller sea lions within the Western DPS are divided into three sub-groups: the Western Gulf of Alaska, Central Gulf of Alaska, and Eastern Gulf of Alaska (Sweeney et al., 2017). Of these three groups, only Steller sea lions from the Eastern Gulf of Alaska and Central Gulf of Alaska are expected to occur within the TMAA, based on proximity of haulout and breeding sites located along the coastline.

While the distribution of sea lions from the two DPSs overlap outside of the breeding season, only a few individuals from the Eastern DPS are expected to occur west of 144° W longitude for a portion of the non-breeding season (Fritz et al., 2016; Jemison et al., 2018). Steller sea lions from the Eastern DPS are expected to remain primarily over the continental shelf, consistent with tagging data, and are not expected to occur in the deeper waters far offshore in the portion of the TMAA east of 144° W longitude (Bishop et al., 2018; Jemison et al., 2018). Reports published since the 2016 GOA Final SEIS/OEIS have provided additional evidence of mixing of the stocks and suggest that it may be inappropriate to treat the eastern and central GOA as “closed” populations (Jemison et al., 2018). During the breeding season, sea lions, especially adult females, typically return to their natal rookery or a nearby breeding rookery to breed and pup (Hastings et al., 2017). The pooled-juvenile home range of Steller sea lions tagged between 2000 and 2014 in Prince William Sound extended from Kayak Island in the east to Kodiak Island in the west, and was generally coastal, with some evidence of excursions offshore onto the shelf, or to adjacent coastal and shelf regions, as well as movement between the two DPSs (Bishop et al., 2018; Jemison et al., 2018; Kuhn et al., 2017).

For additional information about Steller sea lion occurrence and distribution as well as important areas in the TMAA, see Section 5.4.1.4 (Steller Sea Lions) of this SEIS/OEIS and the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### 3.8.2.18 California Sea Lion (*Zalophus californianus*)

#### 3.8.2.18.1 Status and Management

There has been no change in the status or the management of California sea lion since the 2016 GOA Final SEIS/OEIS. The California sea lion is not listed under the ESA. The California sea lion is managed by NMFS as the U.S. stock in all areas where they occur along the U.S. West Coast and in Alaska (Carretta et al., 2020b).



**3.8.2.18.2 Abundance**

The current population estimate of California sea lions in the U.S. stock is 257,606 (Carretta et al., 2020b). The total population in U.S. waters cannot be directly counted because all age and sex classes are not on shore at the same time during field surveys (i.e., a segment of the population is always in the water). In lieu of counting all sea lions, pups are counted during the breeding season, because this is the only age class that is ashore in its entirety. The size of the U.S. stock is then estimated from the number of births and the proportion of pups observed at the surveyed rookeries (Carretta et al., 2020b; Laake et al., 2018). The abundance of California sea lions in the TMAA is not likely to have changed since the 2016 GOA Final SEIS/OEIS; however, warmer water temperatures and changes in the ocean environment may be factors that have favored California sea lions over Steller sea lions in the southern part of the Steller sea lion range in Alaska (Muto et al., 2020). California sea lions are often observed hauled out with Steller sea lions, including on Middleton Island. Counts in the hundreds of California sea lions have been reported at Dry Bay, Alaska, located north of Glacier Bay National Park on the eastern shore of the Gulf of Alaska (based on unpublished data collected by the Alaska Department of Fish and Game).

**3.8.2.18.3 Distribution**

The Navy has determined the California sea lion's occurrence in the TMAA would be seasonal with their highest likelihood of occurring being between April and May. There have been no changes to the known distribution of California sea lions since the 2016 GOA Final SEIS/OEIS. The California sea lion occurs in the eastern North Pacific from Puerto Vallarta, Mexico, through the Gulf of California and north along the west coast of North America to the GOA (Barlow et al., 2008; DeLong et al., 2017; Jefferson et al., 2008; Maniscalco et al., 2004). Typically, during the summer, California sea lions congregate near rookery islands and specific open-water areas. The primary rookeries off the coast of the United States are on San Nicolas, San Miguel, Santa Barbara, and San Clemente Islands (Carretta et al., 2000; Le Boeuf & Bonnell, 1980; Lowry et al., 1992; Lowry & Forney, 2005; Lowry et al., 2017).

This species is prone to invade human-modified coastal sites that provide good haulout substrate, such as marinas, buoys, bait barges, and rip-rap tidal control structures. During the nonbreeding season, they occur most often over the slope or offshore; during the breeding season, they occur most often over the continental shelf (Melin & DeLong, 2000; Melin et al., 2008). Lowry and Forney (2005) estimated that 47 percent of sea lions would potentially be at sea during the cold seasons. Dive durations range from 1.4 to 5.0 minutes, with longer dives during El Niño events; surface intervals range from 0.7 to 17.0 minutes, with sea lions diving about 32–47 percent of the time at sea (Feldkamp et al., 1989; Kuhn & Costa, 2014; Melin et al., 2008; Melin et al., 2012). Adult females alternate between nursing their pup on shore and foraging at sea, spending approximately 67–77 percent of time at sea (Kuhn & Costa, 2014; Melin & DeLong, 2000). Data from satellite tags and time-depth recorders on 15 California sea lions indicated different foraging strategies, with some individuals spending energy at almost twice the rate of other individuals (McHuron et al., 2017). California sea lions would begin migrating out of the Gulf of Alaska to summer breeding sites in California by May, and their occurrence in the Gulf of Alaska is only likely in May or conservatively June. The offshore waters of the Gulf of Alaska are not considered part of the typical habitat for California sea lions, and their occurrence in the TMAA should be considered rare.

For additional information about California sea lion occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.19 Northern Fur Seal (*Callorhinus ursinus*)**

#### **3.8.2.19.1 Status and Management**

There has been no change in the status or the management of northern fur seals since the 2016 GOA Final SEIS/OEIS. Two stocks of northern fur seals are recognized in U.S. waters: an Eastern Pacific stock that breeds in southern Bering Sea and a California stock that breeds in the Farallon Islands and on San Miguel Island (Carretta et al., 2020b; Muto et al., 2020). The Eastern Pacific stock occurs year round in the TMAA, and pups from the California stock may also occur in the Gulf of Alaska year round. Northern fur seals are considered depleted under the MMPA but are not listed under the ESA (Carretta et al., 2020b).

#### **3.8.2.19.2 Abundance**

The abundance of the northern fur seal in the TMAA has not changed substantially since the 2016 GOA Final SEIS/OEIS. The abundance of the Eastern Pacific stock is currently estimated to be 620,660 animals (Muto et al., 2017; Muto et al., 2018b; Muto et al., 2020), and the abundance of the California Stock is estimate to be 14,050 (Carretta et al., 2020b); however, only a small portion of the California (mainly pups) would be expected to occur in the Gulf of Alaska. Nevertheless, the vast majority of fur seals in the Gulf of Alaska would be from the Eastern Pacific stock.

#### **3.8.2.19.3 Distribution**

Northern fur seals' occurrence in the TMAA would be seasonal with their highest likelihood of occurring being between December and July; however, yearlings are potentially present year round, with adults either migrating to or at breeding sites in the Pribilof Islands and Aleutian Islands between June and October (males) or August and November (females). There have been no changes to the known distribution of northern fur seals since the 2016 GOA Final SEIS/OEIS. Northern fur seals range throughout the North Pacific along the west coast of North America, from California (32° N) to the Bering Sea, and west to the Sea of Okhotsk and Honshu Island, Japan (36° N) (Baird & Hanson, 1997; Carretta et al., 2010; Gelatt & Gentry, 2018; Gentry, 2009; Jefferson et al., 2008; Kuhn et al., 2020; Lee et al., 2018; Orr et al., 2018; Ream et al., 2005; Zeppelin et al., 2019). Olesiuk (2012) characterized northern fur seals as ubiquitous in the North Pacific between 60° N and 40° N latitude, with their distribution at sea driven by prey concentrations associated with oceanographic features such as the boundary of the sub-arctic-sub-tropical transition zone near 42° N latitude (Polovina et al., 2001).

In general, the northern fur seals present in the TMAA are those that annually migrate from breeding sites in the Pribilof Islands and Aleutian Islands into the North Pacific each fall, returning in the late spring to give birth (Gelatt & Gentry, 2018). There are no rookeries or breeding sites for the species in or around the TMAA. Migrating fur seals and those along the U.S. West Coast are typically found beyond the continental shelf break and over the slope (Adams et al., 2014; Gentry, 2009; Kenyon & Wilke, 1953; Sterling & Ream, 2004), although two fur seals were tracked over 2,000 km offshore into the central North Pacific Ocean (Ream et al., 2005). Their offshore distribution has been correlated with oceanographic features (e.g., eddies and fronts) where prey may be concentrated (Ream et al., 2005; Sterling et al., 2014). Northern fur seals are found throughout their Pacific offshore range throughout the year, although seasonal fluctuations in distribution occur. Females and pups spend time ashore in the Pribilof Islands and Aleutian Islands of Alaska, then move south to the waters offshore of Oregon and California, while adult males generally move only as far south as the GOA and therefore would be more likely to be present than females or pups in the TMAA (Muto et al., 2020).

Most northern fur seals migrate along continental margins from low-latitude winter foraging areas to northern breeding islands (Gentry, 2009; Lee et al., 2018; Ragen et al., 1995). They leave the breeding islands in November and concentrate around the continental margins of the North Pacific in January and February, where they have access to vast, predictable food supplies and where the Eastern Pacific and the California stocks overlap (Gentry, 2009; Lee et al., 2018; Loughlin et al., 1994; Newsome et al., 2007; Ream et al., 2005). Juveniles have been known to conduct trips between 8 and 29 days in duration, ranging from 171 to 680 km (Sterling & Ream, 2004). Adult female fur seals equipped with radio transmitters have been recorded conducting roundtrip foraging trips of up to 740 km (National Marine Fisheries Service, 2007b; Robson et al., 2004).

For additional information about northern fur seal occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.20 Northern Elephant Seal (*Mirounga angustirostris*)**

#### **3.8.2.20.1 Status and Management**

There has been no change in the status or the management of northern elephant seal since the 2016 GOA Final SEIS/OEIS. The northern elephant seal is not listed under the ESA. The northern elephant seal population has recovered dramatically after being reduced to perhaps no more than 10–100 animals surviving in Mexico in the 1890s (Carretta et al., 2010; Hoelzel, 1999; Stewart et al., 1994). Movement and some genetic interchange occur between rookeries, but most elephant seals return to the rookeries where they were born to breed and thus may have limited genetic differentiation (Carretta et al., 2020b). There are two distinct populations of northern elephant seals: one that breeds in Baja, Mexico; and a population that breeds in California (Garcia-Aguilar et al., 2018). The National Marine Fisheries Service considers northern elephant seals in the TMAA to be from the California Breeding Stock. Although elephant seals from Baja California, Mexico, may migrate north as far as the TMAA, females breeding in Mexico are known to forage approximately 8° farther south than females from the California Breeding stock (Aurioles et al., 2006; Carretta et al., 2020b).

#### **3.8.2.20.2 Abundance**

Lowry et al. (2014) reported that 40,684 pups were born on U.S. rookeries in 2010. Based on the pup count, the population estimate in the California Breeding stock is approximately 179,000 elephant seals (Carretta et al., 2020b).

#### **3.8.2.20.3 Distribution**

Northern elephant seals occurrence in the TMAA would be seasonal with their highest likelihood of occurrence between March and September. Little to no occurrence is expected in January, February, and November. There have been no changes to the known distribution of northern elephant seals since the 2016 GOA Final SEIS/OEIS. Northern elephant seals are found in both coastal and deep waters of the eastern and central North Pacific. Northern elephant seals spend nearly 90 percent of their time at sea underwater, making long migrations to offshore foraging areas and feeding intensively to build up the blubber stores required to support them during breeding and molting seasons when they are hauled out (Hindell & Perrin, 2009; Le Boeuf & Laws, 1994; Worthy et al., 1992). Breeding and pupping take place on offshore islands and mainland rookeries in California (Carretta et al., 2010; Le Boeuf & Laws, 1994; Lowry et al., 2014; Lowry et al., 2017). Small colonies of northern elephant seals breed and haul out on Santa Barbara Island and San Clemente Island, while large colonies are found on Año Nuevo, San Nicolas, Santa, Rosa, and San Miguel islands. Elephant seals use these islands as rookeries from late

December to February, and to molt from April to July. Northern elephant seals spend little time nearshore and migrate through offshore waters four times a year as they travel to and from breeding and molting sites on islands and mainland sites along the Mexico and California coasts.

With most of their prey found in open oceans, northern elephant seal juveniles and females are often found in deepwater, while males tend to forage on the seafloor closer to shore and travel as far north as seamounts in the GOA (Le Boeuf et al., 2000; Le Boeuf et al., 1996; Robinson et al., 2012; Simmons et al., 2010; Simmons et al., 2007; Stewart & DeLong, 1995). The foraging range and distribution of northern elephant seals extends thousands of kilometers offshore into the central North Pacific Transition Zone Chlorophyll Front (Robinson et al., 2012); however, their range is not continuous across the North Pacific (Robinson et al., 2012; Simmons et al., 2010; Stewart & Huber, 1993). Adult males and females segregate while foraging and migrating (Simmons et al., 2010; Stewart, 1997; Stewart & DeLong, 1995). Adult females mostly range west to about 173° W, between the latitudes of 40° N and 45° N, whereas adult males range farther north into the GOA and along the Aleutian Islands to between 47° N and 58° N (Le Boeuf et al., 2000; Robinson et al., 2012; Stewart & DeLong, 1995; Stewart et al., 1993). Robinson et al. (2012) tracked 297 adult female northern elephant seals during post-breeding and post-molting migrations from a central California and a Baja California, Mexico, rookery to foraging areas in the eastern North Pacific. The data showed that female elephant seal foraging areas strongly correlated with the location of the stable boundary separating the sub-arctic and sub-tropical gyres. The boundary fluctuates seasonally but remains between 40° and 50° N latitude and is typically at or slightly north of 45° N latitude, well south of the Gulf of Alaska.

Adults mainly stay offshore during migration, while juveniles are often seen along the coasts of Oregon, Washington, and British Columbia (Le Boeuf et al., 1996; Stewart & Huber, 1993). The most far-ranging individual appeared on Nijima Island off the Pacific coast of Japan in 1989 (Kiyota et al., 1992). This demonstrates the great distances that these animals are capable of covering.

For additional information about northern elephant seal occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.21 Harbor Seal (*Phoca vitulina*)**

#### **3.8.2.21.1 Status and Management**

There has been no change in the status or the management of harbor seals since the 2016 GOA Final SEIS/OEIS. The harbor seal is not listed under the ESA. The Society of Marine Mammalogy's Committee on Taxonomy (2016) has determined that all harbor seals in the North Pacific should be recognized as a single subspecies (*Phoca vitulina richardii*) until the subspecies limits of various populations are better known. There are 17 stocks of harbor seal along the U.S. West Coast, including in Alaska, four of which have the greatest likelihood of occurring in the TMAA: the North Kodiak, South Kodiak, Prince William Sound, and Cook Inlet/Shelikof Strait stocks (Carretta et al., 2020b; Muto et al., 2020). While it is possible that harbor seals may travel farther offshore into the deeper waters of the central TMAA, the vast majority of harbor seals would remain closer to shore, over the continental shelf out to the shelf break, which is estimated as the 500 m isobath.

#### **3.8.2.21.2 Abundance**

The current statewide abundance estimate for Alaska harbor seals is 243,938 (Muto et al., 2020). Abundance estimates for the four stocks considered in this SEIS/OEIS are shown in Table 3.8-1. The eight-year population trend estimates for the Prince William Sound and Cook Inlet/Shelikof Strait stocks

have been decreasing while the North Kodiak and South Kodiak stocks have been increasing (Muto et al., 2020).

#### **3.8.2.21.3 Distribution**

Harbor seal occurrence in the TMAA would be rare year round, except for the nearshore portions of the TMAA that overlap with the continental shelf. Since the 2016 GOA Final SEIS/OEIS, abundance estimates have been updated and harbor seal distribution is expected to occur out to the shelf break. The harbor seal is one of the most widely distributed seals, found in nearly all temperate coastal waters of the northern hemisphere (Jefferson et al., 2008). Harbor seals are generally not present in the deep waters of the open ocean, which includes the majority of the TMAA. Harbor seals prefer coastal habitat, rarely found more than 20 km from shore, and spend much of their time hauled-out along rocky shorelines, and frequently occupying bays, estuaries, and inlets (Baird, 2001; Harvey & Goley, 2011; Huber et al., 2001; Jefferson et al., 2014). Individual seals have been observed several kilometers upstream in coastal rivers (Baird, 2001). Harbor seals are not considered migratory (Burns, 2009; Harvey & Goley, 2011; Jefferson et al., 2008), and data from 180 radio-tagged harbor seals in California indicated most remained within 10 km of the location where they were captured and tagged (Harvey & Goley, 2011). Ideal harbor seal habitat includes suitable haulout sites, shelter from high surf during the breeding periods, and sufficient food near haulout sites to sustain the population throughout the year (Bjorge, 2002). Haulout sites vary but include intertidal and subtidal rock outcrops, sandbars, sandy beaches, estuaries, and even peat banks in salt marshes (Burns, 2009; Gilbert & Guldager, 1998; Prescott, 1982; Schneider & Payne, 1983; Wilson, 1978).

For additional information about harbor seal occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

#### **3.8.2.22 Ribbon Seal (*Histriophoca fasciata*)**

##### **3.8.2.22.1 Status and Management**

There has been no change in the status or the management of ribbon seals since the 2016 GOA Final SEIS/OEIS. The Alaska stock of ribbon seals is not considered a strategic stock (Muto et al., 2020). Ribbon seals are not listed under the ESA.

##### **3.8.2.22.2 Abundance**

A reliable population estimate for the entire stock is not available; however, based on limited survey data, the abundance estimate of 184,697 is a reasonable estimate for the entire U.S. population, because relatively few ribbon seals are expected north of the Bering Strait (Muto et al., 2020).

##### **3.8.2.22.3 Distribution**

Ribbon seal occurrence in the TMAA would be rare year round; however, the highest likelihood of occurrence would be July to September. There is no known range for ribbon seals in Alaska (Muto et al., 2018a); however, ribbon seals inhabit the North Pacific and adjacent parts of the Arctic Ocean. In Alaska waters, ribbon seals occur in the western Beaufort sea, Chukchi sea, Bering Sea, and the North Pacific (Muto et al., 2018a). They are rarely found on shorefast ice or land and are more frequently seen on sea ice and are abundant in the northern part of the ice front in the central and western parts of the Bering Sea. When the ice recedes, they are known to move farther north in the Bering Sea, hauling out on receding ice edges and remnant ice from May through mid-July (Muto et al., 2018a). In 2009, a tagged ribbon seal traveled from the northern Bering Sea into the Gulf of Alaska, indicating that their summer

distribution includes the Gulf of Alaska; however, the number of ribbon seals that could occur in the Gulf of Alaska and the TMAA is unknown.

For additional information about ribbon seal occurrence and distribution in the TMAA, refer to the *U.S. Navy Marine Species Density Database Phase III Technical Report for the Gulf of Alaska Temporary Maritime Activities Area* (U.S. Department of the Navy, 2020b).

### **3.8.2.23 Northern Sea Otter (*Enhydra lutris neris*)**

#### **3.8.2.23.1 Status and Management**

There has been no change in the status or the management of sea otters since the 2016 GOA Final SEIS/OEIS. Unlike all other marine mammals in the TMAA, the Northern sea otter is a species under the federal jurisdiction of USFWS within the Department of the Interior. The range and habitat of sea otters in Alaska as provided in the literature (limited to waters less than 100 m in depth) is well inland of the TMAA boundaries (Coletti et al., 2011; Garlich-Miller et al., 2018; Muto et al., 2020; Tinker et al., 2019; U.S. Fish and Wildlife Service, 2009, 2013; Wolt et al., 2012). The Southwest Alaska Northern sea otter stock is listed as threatened under the ESA (70 FR 46366–46386) and, by definition, is considered a depleted stock under the MMPA (Carretta et al., 2017b). The recovery plan for the Southwest Alaska DPS sea otters includes five management units: (1) Western Aleutian Islands; (2) Eastern Aleutian Islands; (3) South Alaska Peninsula; (4) Bristol Bay; and (5) Kodiak, Kamishak, Alaska Peninsula (Lance et al., 2015; U.S. Fish and Wildlife Service, 2013). Critical habitat has been designated for the Southwest Alaska population of northern sea otters, and it encompasses approximately 15,000 square kilometers of nearshore habitat, none of which is within or near the TMAA. The Southcentral Alaska stock and the Southeast Alaska are also found along the Gulf of Alaska coast, but those populations are not ESA-listed. All sea otter stocks in Alaska are protected under the MMPA, although that same law also allows for sea otters to be hunted and harvested by Alaska Natives for subsistence use. For example, USFWS records for 2013 (not counting fall) indicated Alaska Natives had reported a harvest of 1,380 sea otters for that year (Lichtenstein, 2013).

#### **3.8.2.23.2 Abundance**

The abundance estimates for sea otter stocks in Alaska, as presented in the 2019 Stock Assessment Report, are based on disparate surveys covering parts of each stock's range in separate years; the reported Southeast Alaska stock surveys occurred between 2000 and 2008, the Southcentral Alaska stock surveys occurred between 2000-2010, and the Southwest Alaska stock between 2000 and 2014 (Lance et al., 2015; Muto et al., 2020). The threatened Southwest Alaska stock are stable and may be increasing in number (Muto et al., 2020). The Southcentral Alaska stock and Southeast Alaska stock also appear to be increasing in overall abundance (Muto et al., 2020).

#### **3.8.2.23.3 Distribution**

There have been no changes to the known distribution of sea otters since the 2016 GOA Final SEIS/OEIS. The Southeast Alaska stock extends from Dixon Entrance to Cape Yakataga; the Southcentral Alaska stock extends from Cape Yakataga to Cook Inlet including Prince William Sound, the Kenai Peninsula coast, and Kachemak Bay; and the Southwest Alaska stock includes Kodiak Island, Barren Island, the Alaska Peninsula and Bristol Bay coasts, the Pribilof Islands, and the Aleutian Islands.

Sea otters forage in shallow water, nearshore coastal habitats and are most commonly found in less than 40 m (131 feet [ft.]) of water or within 400 m of the shore (Bodkin, 2015; Bodkin et al., 2004; Coletti et al., 2011; Coletti et al., 2016; Fisheries and Oceans Canada, 2015; Garlich-Miller et al., 2018;

Schneider, 1977; Tinker et al., 2019). In general, sea otters are constrained by their ability to forage at depth deeper than 100 m (Bodkin, 2015; Bodkin et al., 2004; Coletti et al., 2011; Thometz et al., 2014; Tinker et al., 2019). Bodkin (2015) notes that sea otters can be found many kilometers from shore in locations where there are shoals far from land, however, there are no known offshore populations near the TMAA. It is possible that vagrant individuals from the Southcentral Alaska stock or the Southeast Alaska stock of sea otters could potentially occur in the nearshore margins of the TMAA. U.S. Fish and Wildlife Service (2011a) previously determined that the incidence of sea otters occurring offshore was rare and therefore discountable.

### 3.8.3 Environmental Consequences

The Proposed Action, presented as Alternative 1 in this supplemental analysis, consists of activities that have been occurring in the TMAA, that have all been analyzed in the past with regard to potential impacts on marine mammals. These prior analyses include the 2011 GOA EIS/OEIS (U.S. Department of the Navy, 2011d), 2011 Record of Decision (U.S. Department of the Navy, 2011d), the 2016 GOA Final SEIS/OEIS (U.S. Department of the Navy, 2016a), the 2017 Record of Decision (U.S. Department of the Navy, 2017e), regulations pursuant to the MMPA (see 82 FR 19530 dated Thursday April 27, 2017), and Navy activities analyzed pursuant to the ESA in the current NMFS Biological Opinion (National Marine Fisheries Service, 2017b). The National Marine Fisheries Service concluded in its Record of Decision and Final Rule (82 FR 19530) that the Navy's training activities would have a negligible impact on the marine mammal species and stocks present in the TMAA. In its Final Biological Opinion under the ESA, NMFS concluded that the Navy's training activities were not likely to jeopardize the continued existence of any ESA-listed marine mammal species and would not adversely modify any critical habitat. Additionally, the USFWS concurred in 2011 that the Navy's training activities were not likely to adversely affect the threatened Southwest Alaska stock of Northern sea otters under the ESA (U.S. Fish and Wildlife Service, 2011b).

As presented in Section 1.3 (Proposed Action), Alternative 1 (the Proposed Action) for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS (U.S. Department of the Navy, 2016a), the 2017 Record of Decision (U.S. Department of the Navy, 2017e), or the activities analyzed previously by NMFS (82 FR 19530; National Marine Fisheries Service (2017b)). This SEIS/OEIS analyzes the impacts on marine mammals under two alternatives, the No Action Alternative and Alternative 1.

This section presents changes since the 2016 GOA Final SEIS/OEIS and evaluates how and to what degree the activities described in Proposed Action could impact marine mammals in the TMAA (see Section 3.0.3, Resources and Issues Considered for Re-Evaluation in this Document, to set framework for why the Alternative 1 required reanalysis). The stressors analyzed for impacts on marine mammals in the TMAA in the 2011 GOA Final EIS/OEIS (see Section 3.8.7, Environmental Consequences, in the 2011 GOA Final EIS/OEIS) included the following:

- Vessel movements
- Aircraft overflights
- Non-explosive practice ordnance
- High explosive ordnance (at-sea explosions)
- Active sonar
- Expended materials ((ordnance-related materials, targets, flares, chaff, sonobuoys, and marine dye markers)

The stressors analyzed for impacts on marine mammals in the TMAA in the 2016 GOA Final SEIS/OEIS (see Section 3.8.3, Environmental Consequences, in the 2016 GOA Final SEIS/OEIS) included the following:

- sonar and other active acoustic sources
- explosives

The Navy has reduced the number and types of explosives used in the TMAA because unlike the analyses in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS, future proposed training in the TMAA does not include a SINKEX event and its associated munitions. As a result of there being no SINKEX in the current Proposed Action, there are no explosives detonated underwater. The Proposed Action retains activities involving the use of explosives detonating at or near the surface,<sup>4</sup> which have been conservatively modeled as occurring underwater for purposes of estimating potential effects to marine mammals (see U.S. Department of the Navy (2018d)).

The assessment of which stressors are likely to have potential impact on marine mammals presented in the following sections in this SEIS/OEIS have been based on five main categories of information: (1) multiple previous analyses undertaken and conclusions reached by the Navy since 2001 for the same type of training and testing activities as are presented in the Proposed Action, (2) the best available science (see “References” at the end of this section), (3) analysis of strike stressor probabilities for in-water devices and MEM used in the TMAA, (4) regulations and authorizations pursuant to the MMPA reached by NMFS for all other Navy areas analyzed in the Pacific and Atlantic, and (5) Biological Opinions from NMFS and findings from USFWS analyzing the effects of the Navy’s activities on ESA-listed marine mammals for all other Navy areas analyzed in the Pacific and Atlantic. Based on that assessment, each of the potential stressors was evaluated to determine if that stressor should be carried forward for additional analysis of possible impacts on marine mammals resulting from Navy’s training activities in the TMAA.

Since 1995, the U.S. Navy has reported all known or suspected vessel collisions with whales to NMFS, and there have been no known collisions between Navy vessels and whales in the TMAA associated with any of the activities from the Proposed Action. The Navy has several standard operating procedures and mitigation measures for vessel safety that benefit marine mammals through a reduction in the potential for vessel strike, as discussed in Section 2.3.2 (Standard Operating Procedures) and Chapter 5 (Mitigation). Based on the absence of any Navy vessel strikes associated with the Proposed Action in the TMAA and the general reduction in strike incidents Navy-wide since introduction of the Marine Species Awareness Training in 2006, the Navy does not anticipate the occurrence of future vessel strikes to marine mammals within the TMAA during the Proposed Action. For these reasons, the Navy is not requesting authorization of a take by vessel strike during the Proposed Action in the TMAA.

Most in-water devices, such as unmanned underwater vehicles and towed devices, will move slowly through the water and are highly unlikely to strike marine mammals because the mammal could easily avoid the device. In-water devices towed by manned platforms are unlikely to strike a marine mammal

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<sup>4</sup> Throughout this document and in the context of the detonation of explosives, the words “...near the surface...” refer to a detonation occurring in air within 10 m of the ocean surface. These detonations are modeled as if the detonation occurs underwater with all peak pressure and acoustic energy contained with the water and not released at the surface. Unlike the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS, there are no training events involving underwater explosions in the current Proposed Action.



because of the observers on the towing platform and other standard safety measures employed when towing in-water devices. In-water devices that could pose a higher risk to marine mammals are those operated at high speeds and unmanned, but there have been no previous occurrences of a strike by a high speed unmanned in-water device over thousands of deployments across the Navy. As part of military expended materials, small-caliber munitions are inert, are meant to be aimed at targets, and are not long-range weapons. As a result, marine mammals are extremely unlikely to be disturbed or struck by expended small-caliber munitions. There have been no known instances of a seafloor device (such as an anchor) striking a marine mammal as it was being deployed or recovered.

In short, there have been no known instances of physical disturbance or strike to any marine mammals as a result of proposed activities prior to or since the 2016 GOA Final SEIS/OEIS. The Navy will continue to implement mitigation measures for applicable vessel movements, towed in-water devices, and military expended materials during non-explosive activities. The mitigation measures will further avoid or reduce the already low potential for impacts on marine mammals during activities involving physical disturbance or strike stressors. Therefore, the Navy did not carry physical disturbance or strike forward for further stressor re-analysis. The Navy determined (U.S. Department of the Navy, 2016a, 2017e) and NMFS agreed (82 FR 19530; 82 FR 24679; National Marine Fisheries Service (2017b)) that for Navy activities in the TMAA, only acoustics and explosives could potentially result in the incidental taking of marine mammals. An explanation of why the other stressors (such as non-explosive ordnance use [ingestion, and strikes], electronic combat [electromagnetic energy stressors], and discharges of expended materials [physical disturbance, strikes, entanglement, ingestion, sediments and water quality]) listed above are unlikely to result in the incidental taking of marine mammals is provided in the 2016 GOA Final SEIS/OEIS (U.S. Department of the Navy, 2016a) and the NMFS final rule for authorizing those activities under the MMPA (82 FR 19530, Thursday, April 27, 2017). There has been no emergent science since those prior determinations that would change or otherwise call into question those findings, as has been recently reaffirmed by NMFS for other Navy actions (see National Marine Fisheries Service (2020b); 85 FR 46302, Friday, July 31, 2020; and 85 FR 72312, Thursday, 12 November 2020). For these reasons, the stressors analyzed for impacts on marine mammals in the TMAA in this SEIS/OEIS include the following:

- Acoustic (sonar and other transducers, vessel noise, aircraft noise, weapon noise).
- Explosives (at or near the surface).

The majority of the changes in the results of the impact analyses presented in this SEIS/OEIS pursuant to requirements of the MMPA and ESA arise from changes in the model input; specifically, more accurate marine mammal density data, revised acoustic impact criteria, and revised computer modeling of predicted effects on marine mammals. These improvements are described in Section 3.0.1.2 (Navy's Quantitative Analysis to Determine Impacts to Sea Turtles and Marine Mammals). Assessment of likely long-term consequences to populations of marine mammals are provided by empirical data gathered from areas where the Navy routinely trains and tests. Substantial Navy-funded marine mammal survey data, monitoring data, and scientific research have been completed since 2006. These empirical data are beginning to provide insight on the qualitative analysis of the actual (as opposed to model-predicted numerical) impact on marine mammals resulting from Navy training and testing activities based on observations of marine mammals generally in and around Navy Range Complexes.

The following subsections of this SEIS/OEIS (Section 3.8.3.1, Acoustic Stressors; and Section 3.8.3.2, Explosive Stressors) present the potential environmental consequences based on modeling and the scientific observations and investigations made over 12 years of monitoring of Navy training and testing

activities in the Pacific and elsewhere that are representative of the type of activities proposed in this SEIS/OEIS.

### **3.8.3.1 Acoustic Stressors**

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 (National Research Council, 2003, 2005), 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). Many other factors besides just 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.

The ways in which an acoustic exposure could result in immediate effects or long-term consequences for an animal are explained in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). The following Background section discusses what is currently known about acoustic effects to marine mammals. These effects could hypothetically extend from physical injury or trauma to a behavioral or stress response that may or may not be detectable. Injury (physical trauma) can occur to organs or tissues of an animal (Section 3.8.3.1.1.1, Injury). Hearing Loss (Section 3.8.3.1.1.2, Hearing Loss) is a noise-induced decrease in hearing sensitivity, which can be either temporary or permanent. Physiological stress (Section 3.8.3.1.1.3, Physiological Stress) is an adaptive process that helps an animal cope with changing conditions; however, too much stress can result in physiological effects. Masking (Section 3.8.3.1.1.4, Masking) can occur when the perception of a biologically important sound (i.e., signal) is interfered with by a second sound (i.e., noise). Behavioral response (Section 3.8.3.1.1.5, Behavioral Reactions) ranges from brief distractions to avoidance of a sound source to prolonged flight. Extreme behavioral or physiological responses can lead to stranding (Section 3.8.3.1.1.6, Stranding). Long-term consequences (Section 3.8.3.1.1.7, Long-Term Consequences) are those impacts, or accumulation of impacts, that can result in decreases in individual fitness or population changes. To avoid or reduce potential impacts to the maximum extent practicable, the Navy will implement marine mammal mitigation measures during applicable training activities that generate acoustic stressors (see Chapter 5, Mitigation).

The Navy will rely on the previous 2016 GOA Final SEIS/OEIS for the analysis of vessel noise, aircraft noise, and weapon noise, and new applicable and emergent science in regard to these sub-stressors is presented in the sections which follow. Due to new acoustic impact criteria, marine mammal densities, and revisions to the Navy Acoustic Effects Model, the analysis provided in Section 3.8.3.1.2 (Impacts from Sonar and Other Transducers) of this SEIS/OEIS supplants the 2016 GOA Final SEIS/OEIS for marine mammals and changes estimated impacts for some species since the 2016 GOA Final SEIS/OEIS.

#### **3.8.3.1.1 Background**

##### **3.8.3.1.1.1 Injury**

Injury (i.e., physical trauma) refers to the effects on the tissues or organs of an animal due to pressure waves. Injury due to non-explosive acoustic stressors such as sonar is discussed below. Moderate- to low-level sound sources, including vessel and aircraft noise, would not cause injury. Section 3.0.4.3

(Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on injury (i.e., physical trauma) and the framework used to analyze this potential impact.

Several mechanisms of acoustically induced tissue damage (non-auditory) have been proposed and are discussed below.

#### **Injury due to Sonar-Induced Acoustic Resonance**

An object exposed to its resonant frequency will tend to amplify its vibration at that frequency, a phenomenon called acoustic resonance. Acoustic resonance has been proposed as a mechanism by which a sonar or sources with similar operating characteristics could damage tissues of marine mammals. In 2002, NMFS convened a panel of government and private scientists to investigate the potential for acoustic resonance to occur in marine mammals (National Oceanic and Atmospheric Administration, 2002). They modeled and evaluated the likelihood that Navy mid-frequency sonar caused resonance effects in beaked whales that eventually led to their stranding. The conclusions of the group were that resonance in air-filled structures was not likely to have caused the Bahamas stranding in 2000. The frequency at which resonance was predicted to occur in the animals' lungs was 50 Hz, well below the frequencies used by the mid-frequency sonar systems associated with the Bahamas event. Furthermore, air cavity vibrations, even at resonant frequencies, were not considered to be of sufficient amplitude to cause tissue damage, even under the unrealistic scenario in which air volumes would be undamped (unrestrained) by surrounding tissues and the amplitude of the resonant response would be greatest. These same conclusions would apply to other training activities involving acoustic sources. Therefore, the Navy concludes that acoustic resonance would not occur under real training conditions. The potential impact of acoustic resonance is not considered further in this analysis.

#### **Nitrogen Decompression**

Marine mammals mitigate nitrogen gas accumulation in their blood and other tissues, which is caused by gas exchange from the lungs under conditions of increased hydrostatic pressure during diving, through anatomical, behavioral, and physiological adaptations (Hooker et al., 2012).

Although not an injury caused by the interaction of sound with tissues, variations in marine mammal diving behavior or avoidance responses in response to sound exposure have been hypothesized to result in the off-gassing of nitrogen super-saturated tissues, possibly to the point of deleterious vascular and tissue bubble formation (Hooker et al., 2012; Jepson et al., 2003; Saunders et al., 2008) with resulting symptoms similar to decompression sickness (also known as "the bends").

Whether marine mammals can produce deleterious gas emboli has been under debate in the scientific community (Hooker et al., 2012; Saunders et al., 2008), although various lines of evidence have been presented in support of the phenomenon. Some of these postulations are described below.

1. Analyses of bycaught animals demonstrated that nitrogen bubble formation occurs in drowned animals when they are brought to the surface (Bernaldo de Quiros et al., 2013b; Moore et al., 2009). Since gas exchange with the lungs no longer occurs once drowned, tissues become supersaturated with nitrogen due to the reduction in hydrostatic pressure near the surface. This demonstrates that the phenomenon of bubble formation is at least physically possible.
2. The presence of osteonecrosis (bone death due to reduced blood flow) in deep-diving sperm whales has been offered as evidence of impacts due to chronic nitrogen supersaturation and a lifetime of decompression insults (Moore & Early, 2004).

3. Dennison et al. (2012) investigated dolphins stranded in 2009–2010. Using ultrasound, they identified gas bubbles in kidneys of 21 of the 22 live-stranded dolphins and in the liver of two of the 22. The authors postulated that stranded animals were unable to recompress by diving, and thus retained bubbles that would have otherwise re-absorbed in animals that continued to dive. However, the researchers concluded that the minor bubble formation observed could be tolerated since the majority of stranded dolphins released did not re-strand.
4. A fat embolic syndrome (out-of-place fat particles, typically in the bloodstream) was identified by Fernández et al. (2005) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream.
5. Findings of gas and fat emboli in a few stranded Risso's dolphin, and in which sonar exposure was ruled out as a cause of stranding, suggested that other factors, in this case struggling with a prey item, might cause significant variations in dive behavior such that emboli formation could occur (Fernandez et al., 2017).

Only one study has attempted to find vascular bubbles in a freely diving marine mammal (Houser et al., 2009). In that study, no vascular bubbles were imaged by ultrasound in a bottlenose dolphin that repeatedly dove to a 100 m depth and maintained a dive profile meant to maximize nitrogen gas uptake. Thus, although lines of evidence suggest that marine mammals manage excessive nitrogen gas loads, the majority of the evidence for the formation of bubble and fat emboli come from stranded animals in which physiological compromise due to the stranding event is a potential confounding factor.

Researchers have examined how dive behavior affects tissue supersaturation conditions that could put an animal at risk of gas bubble embolism. An early hypothesis was that if exposure to a startling sound elicits a rapid ascent to the surface, tissue gas saturation sufficient for the evolution of nitrogen bubbles might result (Fernandez et al., 2005; Jepson et al., 2003). However, modeling suggested that even unrealistically rapid rates of ascent from normal dive behaviors are unlikely to result in supersaturation to the extent that bubble formation would be expected in beaked whales (Zimmer & Tyack, 2007). Instead, emboli observed in animals exposed to mid-frequency active sonar (Fernandez et al., 2005; Jepson et al., 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth of lung collapse (Aguilar de Soto et al., 2006; Hooker et al., 2012; Tyack et al., 2006; Zimmer & Tyack, 2007). Longer times spent diving at mid-depths above lung collapse would allow gas exchange from the lungs to continue under high hydrostatic pressure conditions, increasing potential for supersaturation; below the depth of lung collapse, gas exchange from the lungs to the blood would likely not occur (Costidis & Rommel, 2016; Fahlman et al., 2014b). To estimate risk of decompression sickness, Kvadsheim et al. (2012) modeled gas exchange in the tissues of sperm, pilot, killer, and beaked whales based on actual dive behavior during exposure to sonar in the wild. Results predicted that venous supersaturation would be within the normal range for these species, which would presumably have naturally higher levels of nitrogen gas loading. Nevertheless, deep-diving whales, such as beaked whales, have also been predicted to have higher nitrogen gas loads in body tissues for certain modeled changes in dive behavior, which might make them more susceptible to decompression sickness (Fahlman et al., 2014b; Fernandez et al., 2005; Hooker et al., 2012; Jepson et al., 2003). Bernaldo de Quirós et al. (2019) summarized discussions from a 2017 workshop on potential sonar impacts on beaked whales, suggesting that the effect of mid-frequency active sonar on beaked whales varies among individuals or populations and that predisposing conditions such as previous exposure to sonar and

individual health risk factors may contribute to individual outcomes (such as decompression sickness) as well.

Modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of long-halftime tissues (i.e., tissues that take longer to give off nitrogen, e.g., fat and bone lipid) to the point that they are supersaturated when the animals are at the surface (Fahlman et al., 2014b; Hooker et al., 2009; Saunders et al., 2008). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al., 2006; Hooker et al., 2009), and because of the time it takes for tissue offloading, it is feasible that long-halftime tissues are not a concern for decompression insults under normal ventilation or dive (recompression) conditions. However, for beaked whale strandings associated with sonar use, one proposed hypothesis is that observed bubble formation may be caused by compromised blood flow due to stranding-related cardiovascular collapse. This would reduce the ability to remove nitrogen from tissues following rapid sonar-induced stranding and could preclude typical management of nitrogen in supersaturated, long-halftime tissues (Houser et al., 2009).

Predictive modeling conducted to date has been performed with many unknowns about the respiratory physiology of deep-diving breath-hold animals. For example, Denk et al. (2020) found intra-species differences in the compliance of tracheobronchial structures of post-mortem cetaceans and pinnipeds under diving hydrostatic pressures, which would affect depth of alveolar collapse. Although, as hypothesized by Garcia Parraga et al. (2018), mechanisms may exist that allow marine mammals to create a pulmonary shunt without the need for hydrostatic pressure-induced lung collapse, i.e., by varying perfusion to the lung independent of lung collapse and degree of ventilation. If such a mechanism exists, then assumptions in prior gas models require reconsideration, the degree of nitrogen gas accumulation associated with dive profiles needs to be re-evaluated, and behavioral responses potentially leading to a destabilization of the relationship between pulmonary ventilation and perfusion should be considered. Costidis and Rommel (2016) suggested that gas exchange may continue to occur across the tissues of air-filled sinuses in deep-diving odontocetes below the depth of lung collapse, if hydrostatic pressures are high enough to drive gas exchange across into non-capillary veins.

If feasible, kinetic gas models would need to consider an additional gas exchange route that might be functional at great depths within the odontocetes. Other adaptations potentially mitigating and defending against deleterious nitrogen gas emboli have been proposed (Blix et al., 2013). Researchers have also considered the accumulation of carbon dioxide produced during periods of high activity by an animal, theorizing that accumulating carbon dioxide, which cannot be removed by gas exchange below the depth of lung collapse, might also facilitate the formation of bubbles in nitrogen-saturated tissues (Bernaldo de Quiros et al., 2012; Fahlman et al., 2014b). In all of these cases, the hypotheses have received little in the way of experimentation to evaluate whether or not they are supported, thus leaving many unknowns as to the predictive accuracy of modeling efforts.

The appearance of extensive bubble and fat emboli in beaked whales was unique to a small number of strandings associated with certain high-intensity sonar events; the phenomenon has not been observed to the same degree in other stranded marine mammals, including other beaked whale strandings not associated with sonar use. It is uncertain as to whether there is some more easily-triggered mechanism for this phenomenon specific to beaked whales or whether the phenomenon occurs only following rapidly occurring stranding events (i.e., when whales are not capable of sufficiently decompressing). Nevertheless, based on the rarity of observations of bubble pathology, the potential for nitrogen

decompression sickness, or “the bends,” as a result of exposure to Navy sound sources is considered discountable.

#### **Acoustically Induced Bubble Formation due to Sonars**

A suggested cause of injury to marine mammals is rectified diffusion (Crum & Mao, 1996), the process of increasing the size of a microscopic gas bubble by exposing it to a sound field. The process is dependent upon a number of factors, including the sound pressure level (SPL) and duration. Under this hypothesis, microscopic bubbles assumed to exist in the tissues of marine mammals may experience one of three things: (1) bubbles grow to the extent they become emboli or cause localized tissue trauma, (2) bubbles develop to the extent that a complement immune response is triggered or the nervous tissue is subjected to enough localized pressure that pain or dysfunction occurs (a stress response without injury), or (3) the bubbles are cleared by the lung without negative consequence to the animal.

Rectified diffusion is facilitated if the environment in which the ensonified bubbles exist is supersaturated with gas. As discussed above, repetitive diving by marine mammals can cause the blood and some tissues to become supersaturated (Ridgway & Howard, 1979). The dive patterns of some marine mammals (e.g., beaked whales) are predicted to induce greater supersaturation (Houser et al., 2001). If rectified diffusion were possible in marine mammals exposed to high-level sound, conditions of tissue supersaturation could theoretically speed the rate and increase the size of bubble growth. Subsequent effects due to tissue trauma and emboli would presumably mirror those observed in humans suffering from decompression sickness.

It is unlikely that the short duration of sonar pulses would be long enough to drive bubble growth to any substantial size, if such a phenomenon occurs. However, an alternative but related hypothesis has also been suggested: stable microbubbles could be destabilized by high-level sound exposures such that bubble growth then occurs through static diffusion of gas out of supersaturated tissues. In such a scenario, the marine mammal would need to be in a gas-supersaturated state for a long enough time for bubbles to become a problematic size. The phenomena of bubble growth due to a destabilizing exposure was shown by Crum et al. (2005) by exposing highly supersaturated ex vivo bovine tissues to a 37 kHz source at 214 dB re 1  $\mu$ Pa. Although bubble growth occurred under the extreme conditions created for the study, these conditions would not exist in the wild because the levels of tissue supersaturation in the study (as high as 400–700 percent) are substantially higher than model predictions for marine mammals (Fahlman et al., 2009; Fahlman et al., 2014b; Houser et al., 2001; Saunders et al., 2008), and such high exposure levels would only occur in very close proximity to the most powerful sonars. For these reasons, it is improbable that this mechanism is responsible for stranding events or traumas associated with beaked whale strandings.

There has been considerable disagreement among scientists as to the likelihood of this phenomenon (Evans & Miller, 2003; Piantadosi & Thalmann, 2004). Although it has been argued that traumas from beaked whale strandings are consistent with gas emboli and bubble-induced tissue separations (Fernandez et al., 2005; Jepson et al., 2003), nitrogen bubble formation as the cause of the traumas has not been verified. The presence of bubbles postmortem, particularly after decompression, is not necessarily indicative of bubble pathology (Bernaldo de Quiros et al., 2012; Bernaldo de Quiros et al., 2013a; Bernaldo de Quiros et al., 2013b; Dennison et al., 2012; Moore et al., 2009), and other mechanisms by which bubble emboli might occur once animals are rapidly stranded (e.g., cardiovascular collapse preventing tissue off-gassing) have not been ruled out (Houser et al., 2009).

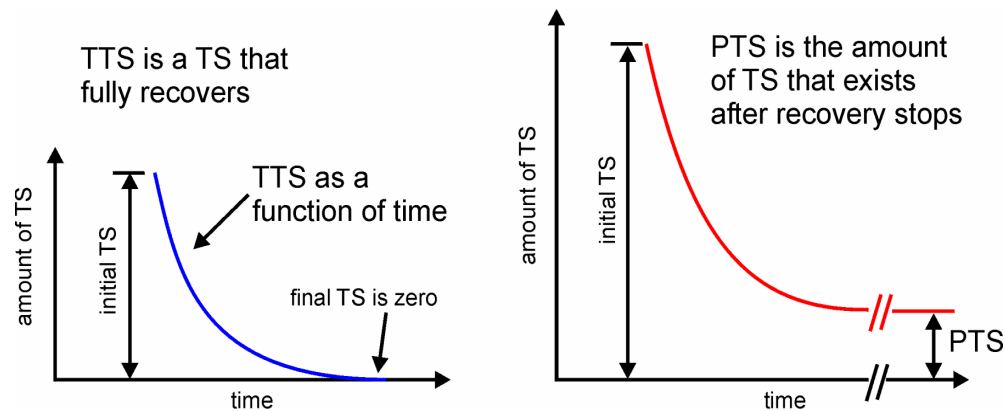
#### 3.8.3.1.1.2 Hearing Loss

Exposure to intense sound may result in noise-induced hearing loss that persists after cessation of the noise exposure. The specific amount of hearing loss, and whether the loss is temporary or permanent, depend on factors such as the exposure frequency, received sound pressure level, temporal pattern, and duration. The frequencies affected by hearing loss will vary depending on the frequency of the fatiguing noise, with frequencies at and above the noise frequency most strongly affected. The amount of hearing loss is highly variable and depends on the species, individual, and contextual factors.

Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on hearing loss and the framework used to analyze this potential impact. Hearing loss has only been studied in a few species of marine mammals, although hearing studies with terrestrial mammals are also informative.

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 called a temporary threshold shift (TTS). If the threshold shift does not completely recover (the threshold remains elevated compared to the pre-exposure value), the remaining threshold shift is called a permanent threshold shift (PTS). Figure 3.8-3 shows two hypothetical threshold shifts: one that completely recovers, a TTS, and one that does not completely recover, leaving some PTS. By definition, TTS is a function of the recovery time, therefore comparing the severity of noise exposures based on the amount of induced TTS can only be done if the recovery times are also taken into account. For example, a 20 dB TTS measured 24 hours post-exposure indicates a more hazardous exposure than one producing 20 dB of TTS measured only two minutes after exposure; if the TTS is 20 dB after 24 hours, the TTS measured after two minutes would have likely been much higher. Conversely, if 20 dB of TTS was measured after two minutes, the TTS measured after 24 hours would likely have been much smaller.

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 using electro-physiological methods, 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, that a TTS in AEP of up to approximately 50 dB, measured 24 hours post-exposure, resulted in neural degeneration. These studies demonstrate that PTS should not be used as the sole indicator of auditory injury, since exposures producing high levels of TTS (40 to 50 dB measured 24 hours after exposure)—but no PTS—may result in auditory injury.



Notes: TTS = Temporary Threshold Shift, TS = Threshold Shift, PTS = Permanent Threshold Shift

**Figure 3.8-3: Two Hypothetical Threshold Shifts**

There are no simple functional relationships between TTS and the occurrence of PTS or other auditory injury (e.g., neural degeneration). However, TTS and PTS are, by definition, mutually exclusive: an exposure that produces TTS cannot also produce PTS within the same frequency band in the same individual (Reichmuth et al., 2019); conversely, if an initial threshold shift only partially recovers, resulting in some amount of PTS, the difference between the initial threshold shift and the PTS is not called TTS. As TTS increases, the likelihood that additional exposure SPL or duration will result in PTS or other injury also increases (with the exception that researchers might not be able to observe gradual growth of TTS with increased sound exposure levels (SELs) before onset of PTS (Reichmuth et al., 2019)). Exposure thresholds for the occurrence of PTS or other auditory injury can therefore be defined based on a specific amount of TTS; that is, we assume that any additional exposure may result in some PTS or other injury. The specific upper limit of TTS is based on experimental data showing amount of TTS that have not resulted in PTS or injury. In other words, we do not need to know the exact functional relationship between TTS and PTS or other injury, we only need to know the upper limit for TTS before some PTS or injury is possible.

A variety of human and terrestrial mammal data indicate that threshold shifts up to 40 dB may be induced without PTS, and that 40 dB is a reasonable upper limit for allowable threshold shift to prevent PTS (e.g., Kryter et al., 1965; Miller et al., 1963; Ward, 1960; Ward et al., 1958; Ward et al., 1959). It is reasonable to assume the same relationship would hold for marine mammals, since there are many similarities between the inner ears of marine and terrestrial mammals, and experiments with marine mammals have revealed similarities to terrestrial mammals for features such as TTS, age-related hearing loss, drug-induced hearing loss, masking, and frequency selectivity (Finneran, 2015; Finneran et al., 2005a; Ketten, 2000). Therefore, we assume that sound exposures sufficient to produce 40 dB of TTS measured approximately four minutes after exposure represent the limit of a non-injurious exposure (i.e., higher level exposures have the potential to cause auditory injury). Exposures sufficient to produce a TTS of 40 dB, measured approximately four minutes after exposure, therefore represent the threshold for auditory injury. The predicted injury could consist of either hair cell damage/loss resulting in PTS or other auditory injury, such as the delayed neural degeneration identified by Kujawa and Liberman (2009) and Lin et al. (2011) that may not result in PTS.

Numerous studies have directly examined noise-induced hearing loss in marine mammals (see Finneran, 2015). In these studies, hearing thresholds were measured in marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds



was then used to determine the amount of TTS at various post-exposure times. The major findings from these studies include the following:

- The method used to test hearing may affect the resulting amount of measured TTS, with neurophysiological (i.e., AEP) measures producing larger amounts of TTS compared to psychophysical (i.e., behavioral) measures (Finneran, 2015; Finneran et al., 2007).
- The amount of TTS usually varies with the hearing test frequency. As the exposure SPL increases, the frequency at which the maximum TTS occurs also increases (Kastelein et al., 2020a; Kastelein et al., 2014a). For high level exposures, the maximum TTS typically occurs one-half to one octave above the exposure frequency (Finneran et al., 2007; Kastelein et al., 2020a; Kastelein et al., 2019d; Kastelein et al., 2019f; Mooney et al., 2009a; Nachtigall et al., 2004; Popov et al., 2013; Popov et al., 2011; Reichmuth et al., 2019; Schlundt et al., 2000). The overall spread of TTS from tonal exposures can therefore extend over a large frequency range (i.e., narrowband exposures can produce broadband [greater than one octave] TTS).
- The amount of TTS increases with exposure SPL and duration, and is correlated with SEL, especially if the range of exposure durations is relatively small (Kastak et al., 2007; Kastelein et al., 2014a; Popov et al., 2014). As the exposure duration increases, however, the relationship between TTS and SEL begins to break down. Specifically, duration has a more significant effect on TTS than would be predicted on the basis of SEL alone (Finneran et al., 2010b; Kastak et al., 2005; Mooney et al., 2009a). This means if two exposures have the same SEL but different durations, the exposure with the longer duration (thus lower SPL) will tend to produce more TTS than the exposure with the higher SPL and shorter duration. In most acoustic impact assessments, the scenarios of interest involve shorter duration exposures than the marine mammal experimental data from which impact thresholds are derived; therefore, use of SEL tends to over-estimate the amount of TTS. Despite this, SEL continues to be used in many situations because it is relatively simple, more accurate than SPL alone, and lends itself easily to scenarios involving multiple exposures with different SPL.
- Gradual increases of TTS may not be directly observable with increasing exposure levels, before the onset of PTS (Reichmuth et al., 2019). Similarly, PTS can occur without measurable behavioral modifications (Reichmuth et al., 2019).
- The amount of TTS depends on the exposure frequency. Sounds at low frequencies, well below the region of best sensitivity, are less hazardous than those at higher frequencies, near the region of best sensitivity (Finneran & Schlundt, 2013). The onset of TTS—defined as the exposure level at which a threshold shift of 6 dB is measured approximately four minutes after exposure (i.e., clearly above the typical variation in threshold measurements)—also varies with exposure frequency. At low frequencies TTS onset exposure levels are higher compared to those in the region of best sensitivity. For example, for harbor porpoises exposed to one-sixth octave noise bands at 16 kHz (Kastelein et al., 2019f), 32 kHz (Kastelein et al., 2019d), and 63 kHz (Kastelein et al., 2020a), less susceptibility to TTS was found as frequency increased, whereas exposure frequencies below ~6.5 kHz showed an increase in TTS susceptibility as frequency increased and approached the region of best sensitivity.
- TTS can accumulate across multiple exposures, but the resulting TTS will be less than the TTS from a single, continuous exposure with the same SEL (Finneran et al., 2010b; Kastelein et al., 2015b; Kastelein et al., 2014a; Mooney et al., 2009b). This means that TTS predictions based on the total, cumulative SEL will overestimate the amount of TTS from intermittent exposures such as sonars and impulsive sources.

- The amount of observed TTS tends to decrease with increasing time following the exposure; however, the relationship is not monotonic (i.e., increasing exposure does not always increase TTS). The time required for complete recovery of hearing depends on the magnitude of the initial shift; for relatively small shifts recovery may be complete in a few minutes, while large shifts (e.g., approximately 40 dB) may require several days for recovery. Recovery times are consistent for similar-magnitude TTS, regardless of the type of fatiguing sound exposure (impulsive, continuous noise band, or sinusoidal wave; (Kastelein et al., 2019e)). Under many circumstances TTS recovers linearly with the logarithm of time (Finneran et al., 2010a, 2010b; Finneran & Schlundt, 2013; Kastelein et al., 2012a; Kastelein et al., 2012b; Kastelein et al., 2013a; Kastelein et al., 2014a; Kastelein et al., 2014b; Kastelein et al., 2014c; Popov et al., 2014; Popov et al., 2013; Popov et al., 2011). This means that for each doubling of recovery time, the amount of TTS will decrease by the same amount (e.g., 6 dB recovery per doubling of time).

Several recent studies have shown that certain odontocete cetaceans (toothed whales) may learn to reduce their hearing sensitivity (presumably to protect their hearing) when warned of an impending intense sound exposure (Finneran, 2018; Nachtigall & Supin, 2013, 2014, 2015; Nachtigall et al., 2015; Nachtigall et al., 2016a, 2018; Nachtigall et al., 2016b). The effect was first demonstrated in a false killer whale (*Pseudorca crassidens*) by Nachtigall and Supin (2013). Subsequent experiments, using similar methods, demonstrated similar conditioned hearing changes in a bottlenose dolphin (*Tursiops truncatus*), (Nachtigall & Supin, 2014, 2015; Nachtigall et al., 2016b), beluga (*Delphinapterus leucas*) (Nachtigall et al., 2015), and harbor porpoises (*Phocoena phocoena*) (Nachtigall et al., 2016a). Using slightly different methods, Finneran (2018) measured the time course and frequency patterns of conditioned hearing changes in two dolphins. Based on these experimental measurements with captive odontocetes, it is likely that wild odontocetes would also suppress their hearing if they could anticipate an impending, intense sound, or during a prolonged exposure (even if not anticipated). Based on the time course and duration of the conditioned hearing reduction, odontocetes participating in some previous TTS experiments could have been protecting their hearing during exposures (Finneran, 2018). A better understanding of the mechanisms responsible for the observed hearing changes is needed for proper interpretation of some existing temporary threshold shift data, particularly for considering TTS due to short duration, unpredictable exposures. No modification of analysis of auditory impacts is currently suggested, as the Phase III auditory impact thresholds are based on best available data for both impulsive and non-impulsive exposures to marine mammals.

Due to the higher exposure levels or longer exposure durations required to induce hearing loss, only a few types of human-made sound sources have the potential to cause a threshold shift to a marine mammal in the wild. Along with some sonars and other transducers, these include impulsive sound sources such as air guns and impact pile driving, neither of which will be used as part of the training activities being covered in this Supplement.

Southall et al. (2019c) evaluated Southall et al. (2007) and used updated scientific information to propose revised noise exposure criteria to predict onset of auditory effects in marine mammals (i.e., PTS and TTS onset). Southall et al. (2019c) note that the quantitative processes described and the resulting exposure criteria (i.e., thresholds and auditory weighting functions) are largely identical to those in (U.S. Department of the Navy, 2017a) and NMFS (2016h, 2018a). However, they differ in that the Southall et al. (2019c) exposure criteria are more broadly applicable as they include all marine mammal species (rather than those only under NMFS jurisdiction) for all noise exposures (both in air and underwater for amphibious species), and that while the hearing group compositions are identical they

renamed the hearing groups. The thresholds discussed in the paper (TTS/PTS only) are the same as Navy's criteria and NMFS criteria.

### **Threshold Shift due to Sonar and Other Transducers**

Temporary Threshold Shift in mid-frequency cetaceans exposed to non-impulsive sound has been investigated in multiple studies (Finneran et al., 2010a; Finneran et al., 2005b; Finneran & Schlundt, 2013; Mooney et al., 2009a; Mooney et al., 2009b; Nachtigall et al., 2003; Nachtigall et al., 2004; Popov et al., 2014; Popov et al., 2013; Schlundt et al., 2000) from two species, bottlenose dolphins and beluga whales. Two high-frequency cetacean species have been studied for TTS due to non-impulsive sources: the harbor porpoise (Kastelein et al., 2013a; Kastelein et al., 2015b; Kastelein et al., 2017a; Kastelein et al., 2014a; Kastelein et al., 2014b) and the finless porpoise (*Neophocaena phocaenoides*) (Popov et al., 2011). Temporary Threshold Shift from non-impulsive sounds has also been investigated in three pinniped species: harbor seal (*Phoca vitulina*), California sea lion (*Zalophus californianus*), and Northern elephant seal (*Mirounga angustirostris*) (e.g., Kastak et al., 2005; Kastelein et al., 2012a). These data are reviewed in detail in Finneran (2015) as well as the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* technical report (U.S. Department of the Navy, 2017a), and the major findings are summarized above.

Several studies of threshold shift in marine mammals exposed to non-impulsive sounds have been published since development of the technical report. Kastelein et al. (2017a) examined threshold shift in harbor porpoises (high-frequency cetaceans) exposed to 3.5–4.1 kHz sonar playbacks. Small amounts of TTS (5–6 dB) were observed after exposures with cumulative, weighted SELs of ~156–162 dB SEL, (~3–9 dB above the TTS onset threshold). The data are therefore consistent with the Phase III thresholds. Popov et al. (2017) measured AEPs at 45 kHz in a beluga (a mid-frequency cetacean) before and after 10-minute exposure to half-octave noise centered at 32 kHz with SPL 170 dB re 1  $\mu$ Pa (weighted SEL = 198 decibels referenced to 1 micropascal squared seconds [dB re 1  $\mu$ Pa<sup>2</sup>s]). After exposure, AEP amplitude vs. stimulus SPL functions were shifted to the right, but returned to baseline values over time. Maximum threshold shift was 23–25 dB, five minutes post-exposure. For these exposures, Phase III criteria over-estimate the observed effects (i.e., Phase III criteria predict 40 dB of TTS for SEL of 198 dB re 1  $\mu$ Pa<sup>2</sup>s). Kastelein et al. (2020c) showed a much higher onset of TTS for a 88.4 kHz exposure as compared to lower exposure frequencies (i.e., 16 kHz (Kastelein et al., 2019f) 1.5 kHz and 6.5 kHz (Kastelein et al., 2020b). For the 88.4 kHz test frequency, a 185 dB re 1  $\mu$ Pa<sup>2</sup>s exposure resulted in 3.6 dB of TTS, and a 191 dB re 1  $\mu$ Pa<sup>2</sup>s exposure produced 5.2 dB of TTS at 100 kHz and 5.4 dB of TTS at 125 kHz. Together, these new studies demonstrate that the criteria for HF cetacean auditory impacts is likely to be conservative.

Kastelein et al. (2019b) measured behavioral hearing thresholds for simulated sonar signals (helicopter long range active sonar at 1.3–1.4 kHz) in two captive harbor seals. Thresholds reported in this study (mean of 51 dB re 1  $\mu$ Pa) are slightly lower than those observed in a prior study of harbor seal behavioral hearing thresholds for tones (Kastelein et al., 2009). The authors suggest this small difference may be due to characteristics of the helicopter long range active sonar signal (duration and/or harmonics) or changes in the test animals' performance over time. The data in this study would not affect the conclusions for acoustic impacts to marine mammals.

Additionally, Kastelein et al. (2019e) exposed two captive harbor seals to 6.5 kHz continuous, sinusoidal sound for one hour in water, resulting in a cumulative SEL between 159 and 195 dB re 1  $\mu$ Pa<sup>2</sup>s, then measured TTS using behavioral hearing thresholds. The highest TTSs were produced in the one-half octave band above the exposure frequency, but individual seals showed variation in the magnitude of

TTS produced. Both seals recovered within one to two hours for up to 6 dB of threshold shift. One seal showed 19 dB of TTS after a 195 dB re 1  $\mu\text{Pa}^2\text{s}$  exposure and recovered within 24 hours. Similarly, Kastelein et al. (2020b) exposed the same seals to 32 kHz, continuous, band-limited noise for one hour, resulting in a cumulative SEL between 128 and 188 dB re 1  $\mu\text{Pa}^2\text{s}$ , and measured less than 6 dB of threshold shift at 32 kHz, which recovered within one hour. At a post-exposure test frequency of 45 kHz (a half-octave above the exposure frequency), the maximum TTS observed in this study were after a ~188 and ~191 dB re 1  $\mu\text{Pa}^2\text{s}$  exposure, which resulted in approximately 34 and 45 dB of TTS, respectively. Recovery occurred over four days for both TTSs. Recovery was gradual for the 34 dB shift, but recovery from the 45 dB shift was not observed until between 4 and 24 hours post-exposure. No TTS was observed at a test frequency of 63 kHz for any sound exposure level. Overall, these studies, combined with previous work, showed that for harbor seals, times to recovery are consistent for similar-magnitude TTS, regardless of the type of sound exposure (impulsive, continuous noise band, or sinusoidal (Kastelein et al., 2020d). However, recovery patterns may be less gradual for higher-magnitude TTS (above 45 dB). Overall, this study, combined with previous work, showed that for harbor seals, recovery times are consistent for similar-magnitude TTS, regardless of the type of sound exposure (impulsive, continuous noise band, or sinusoidal).

A longitudinal study tracked the hearing of a single harbor seal over more than ten years (Reichmuth et al., 2019). The harbor seal was originally exposed to a 4.1 kHz tone, which increased incrementally in SPL and duration over time, and was tested at 5.8 kHz. No reliable TTS was observed until the harbor seal was exposed to 60 s of the tone at 181 dB re 1  $\mu\text{Pa}$ , which resulted in a large threshold shift (> 47 dB). The harbor seal's hearing at 4.1 kHz recovered within two days, but his hearing at one-half (5.8 kHz) and one (8.2 kHz) octave above the frequency of the noise resulted in PTS (8-11 dB) for over 10 and 2 years, respectively. This study contradicts common assumptions about the relationship of TTS and PTS: there was no gradual growth of TTS with increased levels of SEL before onset of PTS, and there were no behavioral fluctuations to indicate that damage to hair cells had occurred. As a result, researchers might not be able to observe gradual TTS with increasing exposure levels, and it is possible for permanent hearing damage to occur without measurable behavioral changes.

### **Threshold Shift due to Impulsive Sound Sources**

Cetacean TTS data from impulsive sources are limited to two studies with measured TTS of 6 dB or more. Finneran et al. (2002) reported behaviorally measured TTSs of 6 and 7 dB in a beluga exposed to single impulses from a seismic water gun and Lucke et al. (2009) reported AEP-measured TTS of 7 to 20 dB in a harbor porpoise exposed to single impulses from a seismic air gun.

In addition to these studies, a number of impulsive noise exposure studies have been conducted without behaviorally measurable TTS of 6 dB or more. The results of these studies are either consistent with the Navy Phase III criteria and thresholds (e.g., exposure levels were below those predicted to cause TTS, and TTS did not occur) or suggest that the Phase III thresholds over-estimate the potential for impact (e.g., exposure levels were above Navy Phase III TTS threshold, but TTS did not occur). The individual studies are summarized below:

- Finneran et al. (2000) exposed dolphins and belugas to single impulses from an “explosion simulator” and Finneran et al. (2015) exposed three dolphins to sequences of 10 impulses from a seismic air gun (maximum cumulative SEL = 193 to 195 dB re 1  $\mu\text{Pa}^2\text{s}$ , peak SPL = 196 to 210 dB re 1  $\mu\text{Pa}$ ) without measurable TTS. Finneran et al. (2003b) exposed two sea lions to single impulses from an arc-gap transducer with no measurable TTS (maximum unweighted SEL = 163 dB re 1  $\mu\text{Pa}^2\text{s}$ , peak SPL = 183 dB re 1  $\mu\text{Pa}$ ).

- Kastelein et al. (2015a) behaviorally measured mean TTS of 4 dB at 8 kHz and 2 dB at 4 kHz after a harbor porpoise was exposed to simulated impact pile driving sound. The cumulative SEL was approximately 180 dB re 1  $\mu\text{Pa}^2\text{s}$  (weighted SEL  $\sim 144$  dB re 1  $\mu\text{Pa}^2\text{s}$ , 4 dB above the TTS onset threshold). Using similar, simulated pile driving noise, but varying total exposure duration from 15 to 360 minutes, Kastelein et al. (2016) found only small amounts of TTS ( $< 6$  dB) in two harbor porpoises. The maximum weighted, cumulative SEL was 156 dB SEL (16 dB above Phase III threshold), but resulted in only  $\sim 5$  dB of TTS.
- Reichmuth et al. (2016) measured behavioral hearing thresholds in two spotted seals and two ringed seals before/after exposure to single air gun impulses and found no TTS. The maximum weighted SEL was  $\sim 156$  dB re 1  $\mu\text{Pa}^2\text{s}$  (14 dB below TTS-onset) and the maximum peak-to-peak SPL was  $\sim 204$  dB re 1  $\mu\text{Pa}$  ( $\sim 8$  dB below TTS onset).
- Kastelein et al. (2017c) measured TTS in a harbor porpoise after exposure to multiple air gun impulses. Either a single or double air gun arrangement was used. Maximum exposure peak pressure was 194/199 dB re 1  $\mu\text{Pa}$  for single/double air guns. Maximum cumulative, weighted SEL was 127/130 dB re 1  $\mu\text{Pa}^2\text{s}$ . Maximum TTS occurred at 4 kHz and was 3 dB/4 dB for single/double air guns. Kastelein et al. (2020e) exposed the same harbor porpoise again to multiple airgun sounds; however, no TTS was found, despite higher single-shot and cumulative sound exposure levels. These studies demonstrate that TTS can be context-dependent and may not be consistent within the same animal exposed to similar sounds.
- Kastelein et al. (2018a) measured TTS in two harbor seals after exposure to playbacks of impact pile-driving recordings. The maximum weighted cumulative SEL is estimated to be  $\sim 182$  dB re 1  $\mu\text{Pa}^2\text{s}$  ( $\sim 12$  dB above Navy Phase III threshold). Maximum peak pressure is estimated to be 176 dB re 1  $\mu\text{Pa}$ ,  $\sim 36$  dB below the Navy Phase III threshold. Small amounts (4 dB maximum) of TTS were observed at 4 kHz after the maximum exposure. Use of Navy Phase III criteria and thresholds would have over-estimated measured effects.
- Kastelein et al. (2019e) found that when two harbor seals were exposed to a 6.5 kHz center frequency fatiguing sound in water, the frequency at which maximum TTS occurred depended on the sound exposure level. For lower sound exposure levels ( $\sim 179$  dB re 1  $\mu\text{Pa}^2\text{s}$  and below), maximum TTS occurred at the center frequency of the fatiguing sound, and was between 0 and 5 dB. For  $\sim 183 - 195$  dB SEL exposures, maximum TTS occurred at a frequency half an octave above the center frequency of the fatiguing sound (9.2 kHz), and was between 4 and 19 dB. Seals recovered at different rates, but TTS of up to 6 dB recovered within one to two hours and TTS of up to 19 dB recovered within 24 hours.

#### 3.8.3.1.1.3 Physiological Stress

The growing field of conservation physiology relies in part on the ability to monitor stress hormones in populations of animals, particularly those that are threatened or endangered. The ability to make predictions from stress hormones about impacts on individuals and populations exposed to various forms of stressors, natural and human-caused, relies on understanding the linkages between changes in stress hormones and resulting physiological impacts. At this time, the sound characteristics that correlate with specific stress responses in marine mammals are poorly understood, as are the ultimate consequences due to these changes. Navy-funded efforts are underway to try to improve the understanding of and the ability to predict how stressors ultimately affect marine mammal populations (e.g., King et al., 2015; New et al., 2013a; Pirotta et al., 2015a). With respect to acoustically induced stress, this includes not only determining how and to what degree various types of anthropogenic sound cause stress in marine mammals, but what factors can mitigate those responses. Factors potentially

affecting an animal's response to a stressor include the mammal's life history stage, sex, age, reproductive status, overall physiological and behavioral plasticity, and whether they are naïve or experienced with the sound (e.g., prior experience with a stressor may result in a reduced response due to habituation (Finneran & Branstetter, 2013; St. Aubin & Dierauf, 2001)). Because there are many unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, the Navy assumes in its effect analysis that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

Marine mammals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to disease and naturally occurring toxins, lack of prey availability, and interactions with predators all contribute to the stress a marine mammal experiences (Atkinson et al., 2015). Breeding cycles, periods of fasting, social interactions with members of the same species, and molting (for pinnipeds) are also stressors, although they are natural components of an animal's life history. Anthropogenic activities have the potential to provide additional stressors beyond those that occur naturally (Fair et al., 2014; Meissner et al., 2015; Rolland et al., 2012). Anthropogenic stressors potentially include such things as fishery interactions, pollution, tourism, and ocean noise.

The stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor (Moberg & Mench, 2000). Over short periods (i.e., hours/days), stress responses can provide access to energetic resources that can be beneficial in life-threatening situations. 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). 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. However, 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. Chronic stressors can occur over the course of weeks or months. Rolland et al. (2017) compared acute (death by ship strike) to chronic (entanglement or live-stranding) stressors in North Atlantic right whales, and found that whales subject to chronic stressors had higher levels of glucocorticoid stress hormones (cortisol and corticosterone) than either healthy whales or those killed by ships. Authors presume that whales subject to acute stress here may have died too quickly for increases in fecal glucocorticoids to be detected.

What is known about the function of the various stress hormones is based largely upon observations of the stress response in terrestrial mammals. The endocrine response of marine mammals to stress may not be the same as that of terrestrial mammals because of the selective pressures marine mammals faced during their evolution in an ocean environment (Atkinson et al., 2015). For example, due to the necessity of breath-holding while diving and foraging at depth, the physiological role of epinephrine and norepinephrine (the catecholamines) might be different in marine versus other mammals. Catecholamines increase during breath-hold diving in seals, co-occurring with a reduction in heart rate, peripheral vasoconstriction (constriction of blood vessels), and an increased reliance on anaerobic metabolism during extended dives (Hance et al., 1982; Hochachka et al., 1995; Hurford et al., 1996); the

catecholamine increase is not associated with an increased heart rate, glycemic release, and increased oxygen consumption typical of terrestrial mammals. Other hormone functions may also be different, such as aldosterone, which has been speculated to not only contribute to electrolyte balance, but possibly also the maintenance of blood pressure during periods of vasoconstriction (Houser et al., 2011). In marine mammals, aldosterone is thought to play a particular role in stress mediation because of its noted response to handling stress (St. Aubin & Dierauf, 2001; St. Aubin & Geraci, 1989).

Relatively little information exists on the linkage between anthropogenic sound exposure and stress in marine mammals, and even less information exists on the ultimate consequences of sound-induced stress responses (either acute or chronic). Most studies to date have focused on acute responses to sound either by measuring catecholamines or by measuring heart rate as an assumed proxy for an acute stress response. Belugas demonstrated no catecholamine response to the playback of oil drilling sounds (Thomas et al., 1990b) but showed a small but statistically significant increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al., 2004). A bottlenose dolphin exposed to the same seismic water gun signals did not demonstrate a catecholamine response, but did demonstrate a statistically significant elevation in aldosterone (Romano et al., 2004), albeit the increase was within the normal daily variation observed in this species (St. Aubin et al., 1996) and was likely of little biological significance with respect to mitigating stress. Increases in heart rate were observed in bottlenose dolphins to which known calls of other dolphins were played, although no increase in heart rate was observed when background tank noise was played back (Miksis et al., 2001). Unfortunately, in this study, it cannot be determined whether the increase in heart rate was due to stress or an anticipation of being reunited with the dolphin to which the vocalization belonged. Similarly, a young beluga's heart rate was observed to increase during exposure to noise, with increases dependent upon the frequency band of noise and duration of exposure, and with a sharp decrease to normal or below normal levels upon cessation of the exposure (Lyamin et al., 2011). Spectral analysis of heart rate variability corroborated direct measures of heart rate (Bakhchina et al., 2017). This response might have been in part due to the conditions during testing, the young age of the animal, and the novelty of the exposure; a year later the exposure was repeated at a slightly higher received level and there was no heart rate response, indicating the beluga whale had potentially habituated to the noise exposure. Kvadsheim et al. (2010a) measured the heart rate of captive hooded seals during exposure to sonar signals and found an increase in the heart rate of the seals during exposure periods versus control periods when the animals were at the surface. When the animals dove, the normal dive-related bradycardia (decrease in heart rate) was not impacted by the sonar exposure. Similarly, Thompson et al. (1998) observed a rapid but short-lived decrease in heart rates in harbor and grey seals exposed to seismic air guns (cited in Gordon et al., 2003). Williams et al. (2017) recently monitored the heart rates of narwhals released from capture and found that a profound dive bradycardia persisted, even though exercise effort increased dramatically as part of their escape response following release. Thus, although some limited evidence suggests that tachycardia might occur as part of the acute stress response of animals that are at the surface, the bradycardia typical of diving in marine mammals appears to be dominant to any stress-related tachycardia and might even be enhanced in response to an acute stressor. Houser et al. (2020) measured cortisol and epinephrine obtained from 30 bottlenose dolphins exposed to simulated U.S. Navy mid-frequency sonar, and found no correlation between sound pressure level and stress hormone levels. In the same experiment (Houser et al., 2013b), behavioral responses were shown to increase in severity with increasing received sound pressure levels. These results suggest that behavioral reactions to sonar signals are not necessarily indicative of a hormonal stress response.

Whereas a limited amount of work has addressed the potential for acute sound exposures to produce a stress response, almost nothing is known about how chronic exposure to acoustic stressors affects stress hormones in marine mammals, particularly as it relates to survival or reproduction. In what is probably the only study of chronic noise exposure in marine mammals associating changes in a stress hormone with changes in anthropogenic noise, Rolland et al. (2012) compared the levels of cortisol metabolites in North Atlantic right whale feces collected before and after September 11, 2001. Following the events of September 11, shipping was significantly reduced in the region where fecal collections were made, and regional ocean background noise declined. Fecal cortisol metabolites significantly decreased during the period of reduced ship traffic and ocean noise (Rolland et al., 2012). Considerably more work has been conducted in an attempt to determine the potential effect of boating on smaller cetaceans, particularly killer whales (Bain, 2002; Erbe, 2002; Lusseau, 2006; Noren et al., 2009; Pirodda et al., 2015b; Read et al., 2014; Rolland et al., 2012; Williams et al., 2009; Williams et al., 2014b; Williams et al., 2014c; Williams et al., 2006). Most of these efforts focused primarily on estimates of metabolic costs associated with altered behavior or inferred consequences of boat presence and noise, but did not directly measure stress hormones. However, Ayres et al. (2012) investigated Southern Resident killer whale fecal thyroid hormone and cortisol metabolites to assess two potential threats to the species' recovery: lack of prey (salmon) and impacts from exposure to the physical presence of vessel traffic (but without measuring vessel traffic noise). Ayres et al. (2012) concluded from these stress hormone measures that the lack of prey overshadowed any population-level physiological impacts on Southern Resident killer whales due to vessel traffic. Collectively, these studies indicate the difficulty in teasing out factors that are dominant in exerting influence on the secretion of stress hormones, including the separate and additive effects of vessel presence and vessel noise. Nevertheless, although the reduced presence of the ships themselves cannot be ruled out as potentially contributing to the reduction in fecal cortisol metabolites in North Atlantic right whales, and there are potential issues in pseudoreplication and study design, the work of Rolland et al. (2012) represents the most provocative link between ocean noise and cortisol in cetaceans to date.

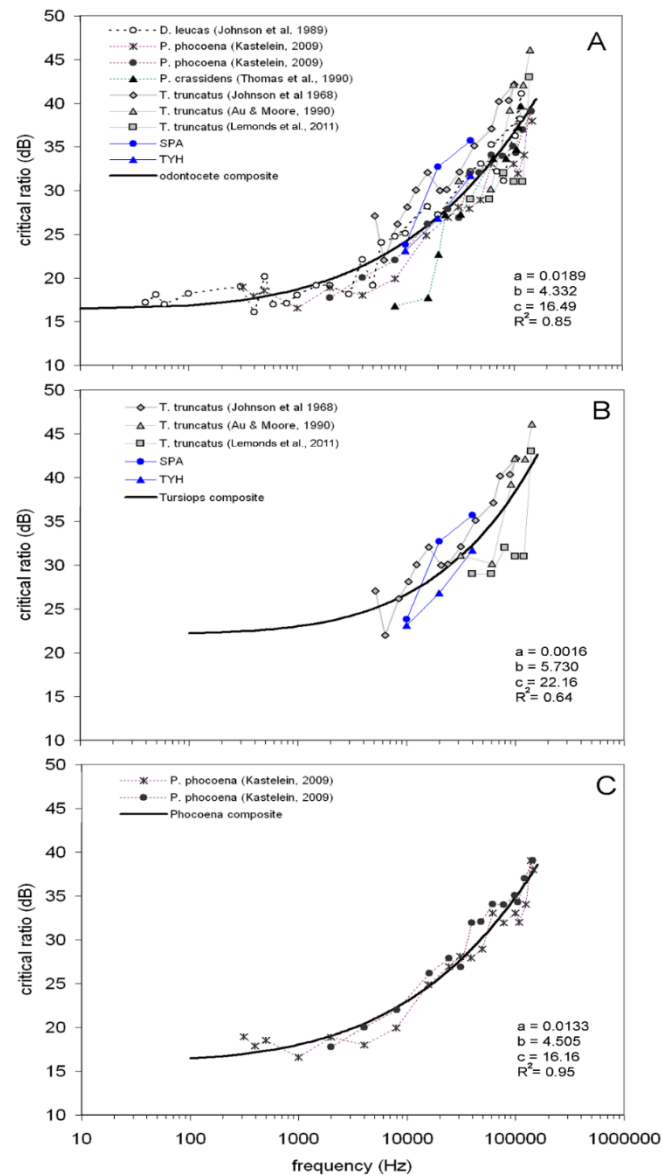
Navy-funded efforts are underway to try and improve our understanding and ability to predict how stressors ultimately affect marine mammal populations (e.g., King et al., 2015; New et al., 2013a; Pirodda et al., 2015a), and to determine whether a marine mammal being naïve or experienced with the sound (e.g., prior experience with a stressor) may result in a reduced response due to habituation (St. Aubin & Dierauf, 2001).

#### **3.8.3.1.1.4 Masking**

Masking occurs when one sound (i.e., noise) interferes with the detection, discrimination, or recognition of another sound (i.e., signal). The quantitative definition of masking is the amount in dB an auditory detection, discrimination, or recognition threshold is raised in the presence of a masker (Erbe et al., 2016). As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), 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 (with the potential exception of reverberations from impulsive noise). Masking can lead to vocal changes such as the Lombard effect (increasing amplitude), other noise-induced vocal modifications such as changing frequency (Hotchkiss & Parks, 2013), and behavioral changes (e.g., cessation of foraging, leaving an area) to both signalers and receivers, in an attempt to compensate for noise levels (Erbe et al., 2016).



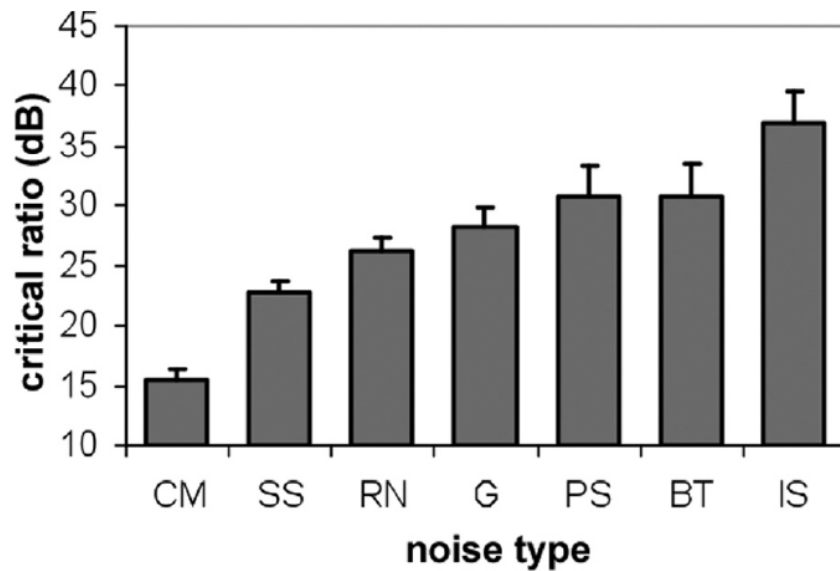
Critical ratios are the lowest signal-to-noise ratio in which detection under masking conditions occurs (Finneran & Branstetter, 2013; Johnson et al., 1989; Southall et al., 2000). When expressed in dB, critical ratios can easily be calculated by subtracting the noise level (in dB re 1  $\mu\text{Pa}^2/\text{Hz}$ ) from the signal level (in dB re 1  $\mu\text{Pa}$ ) at threshold. Critical ratios have been measured for pinnipeds (Southall et al., 2000, 2003), odontocetes (Au & Moore, 1990; Branstetter et al., 2017b; Johnson et al., 1989; Kastelein & Wensveen, 2008; Lemonds et al., 2011; Thomas et al., 1990a), and sea otters (Ghoul & Reichmuth, 2014b). Critical ratios increase as a function of signal frequency (Au & Moore, 1990; Lemonds et al., 2011). Higher frequency noise is more effective at masking higher frequency signals. Composite critical ratio functions have been estimated for odontocetes (Figure 3.8-4), which allow predictions of masking if the spectral density of noise is known (Branstetter et al., 2017b). Although critical ratios are typically estimated in controlled laboratory conditions using Gaussian (white) noise, critical ratios can vary considerably (see Figure 3.8-5) depending on the noise type (Branstetter et al., 2013; Trickey et al., 2010). Signal type (e.g., whistles, burst-pulse, sonar clicks) and spectral characteristics (e.g., frequency modulation and/or harmonics) may further influence masked detection thresholds (Branstetter et al., 2016; Branstetter & Finneran, 2008; Branstetter et al., 2013; Cunningham et al., 2014).



Source: Branstetter et al. (2017b)

Notes: (1) Odontocete critical ratios and composite model:  $CR = a[\log_{10}(f)]^b + c$ , where  $a$ ,  $b$ , and  $c$  are model coefficients and  $f$  is the signal frequency in Hz. Equation 1 was fit to aggregate data for all odontocetes. (2) *T. truncatus*. critical ratios and composite model. (3) *P. phocoena*. critical ratios and composite model. Parameter values for composite models are displayed in the lower right of each panel.

**Figure 3.8-4: Odontocete Critical Ratios**



Source: Branstetter et al. (2013)

Notes: CM = comodulated, SS = snapping shrimp, RN = rain noise, G = Gaussian, PS = pile saw, BT = boat engine noise, and IS = ice squeaks

**Figure 3.8-5: Critical Ratios for Different Noise Types**

Clark et al. (2009) developed a model for estimating masking effects on communication signals for low-frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, the model estimates that a right whale's optimal communication space (around 20 km) is decreased by 84 percent when two commercial ships pass through it. Similarly, Aguilar de Soto et al. (2006) found that a 15 dB increase in background noise due to vessels led to a communication range of only 18 percent of its normal value for foraging beaked whales. This method relies on empirical data on source levels of calls (which is unknown for many species) and requires many assumptions such as pre-industrial ambient noise conditions and simplifications of animal hearing and behavior, but it is an important step in determining the impact of anthropogenic noise on animal communication. Erbe (2016) developed a model with a noise source-centered view of masking to examine how a call may be masked from a receiver by a noise as a function of caller, receiver, and noise-source location, distance relative to each other, and received level of the call.

Vocal changes in response to anthropogenic noise can occur across the repertoire of sound production modes used by marine mammals, such as whistling, echolocation click production, calling, and singing. Vocalization changes 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 (Hotchkiss & Parks, 2013). In cetaceans, vocalization changes were reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying (Gordon et al., 2003; Holt et al., 2011; Holt et al., 2008; Lesage et al., 1999; McDonald et al., 2009; Rolland et al., 2012) as well as changes in the natural acoustic environment (Caruso et al., 2020; Dunlop et al., 2014; Helble et al., 2020). Vocal changes can be temporary, or can be persistent, as seen in the increase in starting frequency for the North Atlantic right whale upcall over the last 50 years (Tenneissen & Parks, 2016). Model simulation suggests that the frequency shift resulted in increased detection ranges between right whales; the

frequency shift, coupled with an increase in call intensity by 20 dB, led to a call detectability range of less than 3 km to over 9 km (Tennessen & Parks, 2016). In some cases, these vocal changes may have fitness consequences, such as an increase in metabolic rates and oxygen consumption, as was found for bottlenose dolphins when increasing their call amplitude (Holt et al., 2015). A switch from vocal communication to physical, surface-generated sounds such as pectoral fin slapping or breaching was observed for humpback whales in the presence of increasing natural background noise levels, indicating that adaptations to masking may not be limited to vocal modifications (Dunlop et al., 2010). These changes all represent possible tactics by the sound-producing animal to reduce the impact of masking. The receiving animal can also reduce masking by using active listening strategies such as orienting to the sound source, moving to a quieter location, or reducing self-noise from hydrodynamic flow by remaining still.

### **Spatial Release from Masking**

Spatial release from masking (SRM) will occur when a noise and signal are separated in space, resulting in a reduction or elimination of masking (Holt & Schusterman, 2007; Popov et al., 2020). The relative position of sound sources can act as one of the most salient cues that allow the listener to segregate multiple sounds in a complex auditory scene. Many sounds are emitted from a directional source that is spatially separated from biologically relevant signals. Under such conditions, minimal masking will occur, and existing models of auditory masking will overestimate the amount of actual masking. Marine mammals have excellent sound source localization capabilities (Branstetter & Mercado, 2006; Byl et al., 2019; Renaud & Popper, 1975) and a directional receiving beam pattern (see Section 3.8.2.1.4, Hearing and Vocalization), which likely combine to aid in separating auditory events, thus improving detection performance.

Spatial release from masking has been empirically demonstrated using behavioral methods in a harbor seal and a California sea lion for 1, 8, and 16 kHz tones in air (Holt & Schusterman, 2007), where maximal SRM was 19 and 12 dB for each species respectively. Byl et al. (2019) used psychophysical methods to test the horizontal underwater sound-localization acuity of harbor seals for two noise bands (8–16 kHz and 14–16 kHz). When compared to sound-localization results for tonal stimuli in the same subjects (Byl et al., 2016), these results show better sound localization for stimuli with more spectral information.

Popov et al. (2020) measured the auditory-evoked potential (AEP) in a single bottlenose dolphin and observed 32 dB of masking when there was no separation between a 64 kHz signal and noise presented directly in front of the animal. Spatial release from masking occurred when the masker was moved 30 degrees or more off-axis, but smaller angular separations between signal and noise were not tested. Approximately 16–24 dB of SRM was observed, but thresholds did not return to baseline even when the masker was 90 degrees to the left or right of center. While these results are pertinent, some of the brain structures that produce the AEP receive information from both ears, which might reduce the ability of this method (as opposed to behavioral methods) to fully describe SRM.

### **Informational Masking**

Much emphasis has been placed on signal detection in noise and, as a result, most masking studies and communication space models have focused on masked detection thresholds. However, from a fitness perspective, signal detection is almost meaningless without the ability to determine the sound source location and recognize “what” is producing the sound. Marine mammals use sound to recognize conspecifics, prey, predators, or other biologically significant sources (Branstetter et al., 2016). Masked

recognition thresholds (often called informational masking) for whistle-like sounds, have been measured for bottlenose dolphins (Branstetter et al., 2016) and are approximately 4 dB above detection thresholds (energetic masking) for the same signals. It should be noted that the term “threshold” typically refers to the listener’s ability to detect or recognize a signal 50 percent of the time. For example, human speech communication, where only 50 percent of the words are recognized, would result in poor communication (Branstetter et al., 2016). Likewise, recognition of a conspecific call or the acoustic signature of a predator at only the 50 percent level could have severe negative impacts. If “quality communication” is arbitrarily set at 90 percent recognition (which may be more appropriately related to animal fitness), the output of communication space models (which are based on 50 percent detection) would likely result in a significant decrease in communication range (Branstetter et al., 2016).

Marine mammals use sound to recognize predators (Allen et al., 2014; Cummings & Thompson, 1971; Curé et al., 2015; Fish & Vania, 1971). Auditory recognition may be reduced in the presence of a masking noise, particularly if it occurs in the same frequency band. Therefore, the occurrence of masking may prevent marine mammals from responding to the acoustic cues produced by their predators. Whether this is a possibility depends on the duration of the masking and the likelihood of encountering a predator during the time that detection and recognition of predator cues are impeded. For example, harbor seals that reside in the coastal waters off British Columbia are frequently targeted by mammal-eating killer whales. The seals acoustically discriminate between the calls of mammal-eating and fish-eating killer whales (Deecke et al., 2002), a capability that should increase survivorship while reducing the energy required to attend to all killer whale calls. Similarly, sperm whales (Curé et al., 2016; Isojunno et al., 2016), long-finned pilot whales (Visser et al., 2016), and humpback whales (Curé et al., 2015) changed their behavior in response to killer whale vocalization playbacks; these findings indicating that some recognition of predator cues could be missed if the killer whale vocalizations were masked.

#### **Masking by Sonar and Other Transducers**

Masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Because traditional military sonars typically have low duty cycles, relatively short duration, and narrow bandwidth that does not overlap with vocalizations for most marine mammal species, the effects of such masking would be limited when compared with continuous sources (e.g., vessel noise). Dolphin whistles and mid-frequency active sonar are similar in frequency, so masking is possible but less likely due to the low-duty cycle of most sonars. Low-frequency active sonar could also overlap with mysticete vocalizations (e.g., minke and humpback whales). For example, in the presence of low-frequency active sonar, humpback whales were observed to increase the length of their songs (Fristrup et al., 2003; Miller et al., 2000), possibly due to the overlap in frequencies between the whale song and the low-frequency active sonar.

Newer high-duty cycle or continuous active sonars have more potential to mask vocalizations, particularly for delphinids and other mid-frequency cetaceans. These sonars transmit more frequently (greater than 80 percent duty cycle) than traditional sonars, but at a substantially lower source level. Similarly, high-frequency acoustic sources such as pingers that operate at higher repetition rates (e.g., 2–10 kHz with harmonics up to 19 kHz, 76–77 pings per minute (Culik et al., 2001)), also operate at lower source levels. While the lower source levels limit the range of impact compared to traditional systems, animals close to the sonar source are likely to experience masking on a much longer time scale than those exposed to traditional sonars. The frequency range at which high-duty cycle systems operate overlaps the vocalization frequency of many mid-frequency cetaceans. Continuous noise at the same frequency of communicative vocalizations may cause disruptions to communication, social interactions,

and acoustically mediated cooperative behaviors such as foraging or reproductive activities. Similarly, because the systems are mid-frequency, there is the potential for the sonar signals to mask important environmental cues like predator vocalizations (e.g., killer whales), possibly affecting survivorship for targeted animals. While there are currently no available studies of the impacts of high-duty cycle sonars on marine mammals, masking due to these systems is likely analogous to masking produced by other continuous sources (e.g., vessel noise and low-frequency cetaceans), and will likely have similar short-term consequences, though longer in duration due to the duration of the masking noise. These may include changes to vocalization amplitude and frequency (Brumm & Slabbekoorn, 2005; Hotchkin & Parks, 2013) and behavioral impacts such as avoidance of the area and interruptions to foraging or other essential behaviors (Gordon et al., 2003). 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 & Slabbekoorn, 2005), a potential decrease in survivorship if predator vocalizations are masked (Brumm & Slabbekoorn, 2005), and a potential decrease in recruitment if masking interferes with reproductive activities or mother-calf communication (Gordon et al., 2003).

### **Masking by Vessel Noise**

Masking is more likely to occur in the presence of broadband, relatively continuous noise sources such as vessels. For example, right whales were observed to shift the frequency content of their calls upward while reducing the rate of calling in areas of increased anthropogenic noise (Parks et al., 2007) as well as increasing the amplitude (intensity) of their calls (Parks, 2009; Parks et al., 2011). Right whales also had their communication space reduced by up to 84 percent in the presence of vessels (Clark et al., 2009). Cholewiak et al. (2018) found that right whale gunshot calls had the lowest loss of communication space in Stellwagen National Sanctuary (5 percent), while fin and humpback whales lost up to 99 percent of their communication space with increased ambient noise and shipping noise combined. Although humpback whales off Australia did not change the frequency or duration of their vocalizations in the presence of ship noise, their source levels were lower than expected based on source level changes to wind noise, potentially indicating some signal masking (Dunlop, 2016). Vessel noise decreased the 4 km of humpback whale modeled communication space (with wind noise up to 100 dB re 1  $\mu$ Pa) to 3 km at the same received level, and at 105 dB re 1  $\mu$ Pa of noise communication space decreased again to 2 km for low-frequency signals and 1 km for high-frequency signals (Dunlop, 2019). Communication space loss due to vessels in Glacier Bay National Park was estimated to be lower for singing humpback whales than for calling whales and was highest for roaring harbor seals, but synchronizing the arrival and departure times of ships into the park restored some of that communication space for the calling whales and seals (Gabriele et al., 2018). Fournet et al. (2018) found humpback whales increase their call source levels by 0.8 dB and decrease the probability of calling by 9 percent for every 1 dB increase in ambient sound, which included vessel noise.

Multiple delphinid species have also been shown to increase the minimum or maximum frequencies of their whistles in the presence of anthropogenic noise (Papale et al., 2015). More specifically, Williams et al. (2014b) found that in median noise conditions in Haro Strait, killer whales lose 62 percent of their acoustic communication space in the frequency band of their social calls (1.5–3.5 kHz) out to 8 km due to vessel traffic noise, and in peak traffic hours lose up to 97 percent of that space; however, when looking at a smaller area or higher frequency bands, less communication space is lost. In fact, at the higher frequency band of their echolocation clicks (18–30 kHz), no communication space was lost out to 2 km. Holt et al. (2011; 2008) showed that Southern Resident killer whales in the waters surrounding

the San Juan Islands increased their call source level as vessel noise increased. In the presence of boats off the Southern end of Vancouver, Southern Resident killer whales changed the duration of 16 out of 21 discrete call types (Wieland et al., 2010). Most of those call types (n=14) increased mean duration, while 2 call types decreased in duration. Hermannsen et al. (2014) estimated that broadband vessel noise could extend up to 160 kHz at ranges from 60 to 1,200 m, and that the higher frequency portion of that noise might mask harbor porpoise clicks. However, this may not be an issue as harbor porpoises may avoid vessels and may not be close enough to have their clicks masked (Dyndo et al., 2015; Polacheck & Thorpe, 1990; Sairanen, 2014). Furthermore, Hermannsen et al. (2014) estimated that a 6 dB elevation in noise would decrease the hearing range of a harbor porpoise by 50 percent, and a 20 dB increase in noise would decrease the hearing range by 90 percent. Gervaise et al. (2012) estimated that beluga whales in the St. Lawrence Marine Park had their communication space reduced to 30 percent during average vessel traffic. During peak traffic, communication space was further reduced to 15 percent. Lesage et al. (1999) found belugas in the St. Lawrence River estuary reduced overall call rates but increased the production of certain call types when ferry and small outboard motor boats were approaching. Furthermore, these belugas increased the vocalization frequency band when vessels were in close proximity. Liu et al. (2017) found that broadband shipping noise could cause masking of humpback dolphin whistles within 1.5–3 km, and masking of echolocation clicks within 0.5–1.5 km.

### **Masking by Impulsive Sound**

Potential masking from weapon noise is likely to be similar to masking studied for other impulsive sounds, such as air guns. Masking could occur in mysticetes due to the overlap between their low-frequency vocalizations and the dominant frequencies of impulsive sources, however, masking in odontocetes or pinnipeds is less likely unless the activity is in close range when the pulses are more broadband. For example, differential vocal responses in marine mammals were documented in the presence of seismic survey noise. An overall decrease in vocalizations during active surveying was noted in large marine mammal groups (Potter et al., 2007), while blue whale feeding/social calls increased when seismic exploration was underway (Di Lorio & Clark, 2010), indicative of a possible compensatory response to the increased noise level. Furthermore, in the presence of biological interference from conspecific echolocation clicks (i.e., sonar jamming), cetaceans exhibit compensatory behaviors. Kloepper and Branstetter (2019) showed that individual bottlenose dolphins responded to jamming signals by omitting clicks (i.e., utilized a temporal response) or increasing click bandwidth (i.e., utilized a spectral response). Bowhead whales were found to increase call rates in the presence of seismic air gun noise at lower received levels (below 100 dB re: 1  $\mu\text{Pa}^2\text{s}$  cumulative SEL), but once the received level rose above 127 dB re 1  $\mu\text{Pa}^2\text{s}$  cumulative SEL the call rate began decreasing, and stopped altogether once received levels reached 170 dB re 1  $\mu\text{Pa}^2\text{s}$  cumulative SEL (Blackwell et al., 2015). Nieukirk et al. (2012) recorded both seismic surveys and fin whale 20 Hz calls at various locations around the mid-Atlantic Ocean, and hypothesized that distant seismic noise could mask those calls thereby decreasing the communication range of fin whales, whose vocalizations may propagate over 400 km to reach conspecifics (Spiesberger & Fristrup, 1990). Two captive seals (one spotted and one ringed) were exposed to seismic air gun sounds recorded within 1 km and 30 km of an air gun survey conducted in shallow (<40 m) water. They were then tested on their ability to detect a 500-millisecond upswEEP centered at 100 Hz at different points in the air gun pulse (start, middle, and end). Based on these results, a 100 Hz vocalization with a source level of 130 dB re 1  $\mu\text{Pa}$  would not be detected above a seismic survey 1 km away unless the animal was within 1–5 m, and would not be detected above a survey 30 km away beyond 46 m (Sills et al., 2017).

#### 3.8.3.1.1.5 Behavioral Reactions

As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), any stimulus in the environment can cause a behavioral response in marine mammals. These stimuli include noise from anthropogenic sources such as vessels, sonar, or aircraft, but could also include the physical presence of a vessel or aircraft. However, stimuli such as the presence of predators, prey, or conspecifics could also influence how or if a marine mammal responds to a sound. Furthermore, the response of a marine mammal to an anthropogenic sound may depend on the frequency, duration, temporal pattern and amplitude of the sound as well as the animal's prior experience with the sound and their behavioral state (i.e., what the animal is doing and their energetic needs at the time of the exposure) (Ellison et al., 2011). 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).

For marine mammals, a review of responses to anthropogenic sound was first conducted by Richardson et al. (1995b). Other reviews (Nowacek et al., 2007; Southall et al., 2007) addressed studies conducted since 1995 and focused on observations where the received sound level of the exposed marine mammal(s) was known or could be estimated, and also examined the role of context. Southall et al. (2007) synthesized data from many past behavioral studies and observations to determine the likelihood of behavioral reactions at specific sound levels, and 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). Ellison et al. (2011) outlined an approach to assessing the effects of sound on marine mammals that incorporates these contextual-based factors. They recommend considering not just the received level of sound, but also in what activity the animal is engaged, the nature and novelty of the sound (i.e., is this a new sound from the animal's perspective), and the distance between the sound source and the animal. They submit that this "exposure context," as described, greatly influences the type of behavioral response exhibited by the animal (see technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a)). Forney et al. (2017) also point out that an apparent lack of response (e.g., no displacement or avoidance of a sound source) may not necessarily mean there is no cost to the individual or population, as some resources or habitats may be of such high value that animals may choose to stay, even when experiencing stress or hearing loss. Forney et al. (2017) recommend considering both the costs of remaining in an area of noise exposure such as TTS, PTS, or masking, which could lead to an increased risk of predation or other threats or a decreased capability to forage, and the costs of displacement, including potential increased risk of vessel strike or bycatch, increased risks of predation or competition for resources, or decreased habitat suitable for foraging, resting, or socializing.

Behavioral reactions could result from a variety of sound sources such as sonar and other transducers (e.g., pingers), vessel noise, and aircraft noise. There are data on the reactions of some species in different behavioral states, providing evidence on the importance of context in gauging a behavioral response. However, for most species, little or no data exist on behavioral responses to any sound source, and so all species have been grouped into broad taxonomic groups from which general response information can be inferred (see technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a)).



### **Behavioral Reactions to Sonar and Other Transducers**

Sonar and other transducers can range in frequency from less than 1 kHz (e.g., low-frequency active sonar) to over 200 kHz (e.g., fish finders), with duty cycles that range from one ping per minute to an almost continuous sound. Although very high-frequency sonars are out of the hearing range of most marine mammals, some of these sources may contain artifacts at lower frequencies that could be detected (Deng et al., 2014; Hastie et al., 2014). High-duty cycle sonar systems operate at lower source levels, but with a more continuous sound output. These sources can be stationary, or on a moving platform, and there can be more than one source present at a time. Guan et al. (2017) also found that sound levels in the mid-frequency sonar bandwidth remained elevated at least 5 dB above background levels for the first 7–15 seconds (within 2 km) after the emission of a sonar ping; depending on the length of the sonar ping and the inter-ping interval, this reverberation could increase cumulative SEL estimates during periods of active sonar. 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. As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) and Section 3.8.3.1.1.5 (Behavioral Reactions), 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.

In order to explore this complex question, behavioral response studies have been conducted through the collaboration of various research and government organizations in Bahamian, United States (off Southern California), Mediterranean, Australian, and Norwegian waters. These studies have attempted to define and measure responses of beaked whales and other cetaceans to controlled exposures of sonar and other sounds to understand better their potential impacts. While controlling for as many variables as possible (e.g., the distance and movement of the source), these studies also introduce additional variables that do not normally occur in a real Navy training activity, including the tagging of whales, following the tagged animals with multiple vessels, and continually approaching the animal to create a dose escalation. In addition, distances of the sound source from the whales during behavioral response studies were always within 1–8 km. Some of these studies have suggested that ramping up a source from a lower source level would act as a mitigation measure to protect against higher order (e.g., TTS or PTS) impacts of some active sonar sources; however, this practice may only be effective for more responsive animals, and for short durations (e.g., five minutes) of ramp-up (von Benda-Beckmann et al., 2014; von Benda-Beckmann et al., 2016; Wensveen et al., 2017). Therefore, while these studies have provided the most information to date on behavioral responses of marine mammals to sonar, there are still many contextual factors to be teased apart, and determining what might produce a significant behavioral response is not a trivial task. Additional information about active sonar ramp-up procedures, including why the Navy will not implement them as mitigation under the Proposed Action, is provided in Section 5.5.1 (Active Sonar).

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 training activity and associated sources to assess behavioral responses (Deakos & Richlen, 2015; Henderson et al., 2016; Manzano-Roth et al., 2016; Martin et al., 2015; McCarthy et al., 2011; Mobley & Deakos, 2015; Moretti et al., 2014; Tyack et al., 2011). In addition, extensive aerial, visual, and passive acoustic monitoring have been conducted before, during, and after training events to watch for behavioral

responses during training and look for injured or stranded animals after training (Falcone et al., 2017; Farak et al., 2011; Henderson et al., 2016; Manzano-Roth et al., 2016; Mobley, 2011; Norris et al., 2012a; Norris et al., 2012b; Smultea & Mobley, 2009; Smultea et al., 2009; Trickey et al., 2015; U.S. Department of the Navy, 2011c, 2013b, 2014b, 2015). During all of these monitoring efforts, very few behavioral responses were observed, and no injured or dead animal was observed that was directly related to a training event (some dead animals were observed but typically before the event or appeared to have been deceased prior to the event; e.g., Smultea et al., 2011). While 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, these study types have the benefit of occurring in the absence of some of the added contextual variables in the controlled exposure studies. Furthermore, when visual and passive acoustic data collected during a training event are combined with ship movements and sonar use, and with tagged animal data when possible, they provide a unique and realistic scenario for analysis, as in Falcone et al. (2017), Manzano-Roth et al. (2016), or Baird et al. (2017). In addition to these types of observational behavioral response studies, Harris and Thomas (2015) highlighted additional research approaches that may provide further information on behavioral responses to sonar and other transducers beyond behavior response type studies or passive acoustic monitoring, including conducting controlled exposures on captive animals with scaled (smaller sized and deployed at closer proximity) sources, on wild animals with both scaled and real but directed sources, and predator playback studies, all of which will be discussed below.

The above behavioral response studies and observations have been conducted on a number of mysticete and odontocete species, which can be extrapolated to other similar species in these taxonomic groups. No field studies of pinniped behavioral responses to sonar have been conducted; however, there are several captive studies on some pinniped and odontocete species that can provide insight into how these animals may respond in the wild. The captive studies typically represent a more controlled approach, which allow researchers to better estimate the direct impact of the received level of sound leading to behavioral responses, and to potentially link behavioral to physiological responses. However, there are still contextual factors that must be acknowledged, including previous training to complete tasks and the presence of food rewards upon completion. There are no corresponding captive studies on mysticete whales; therefore, some of the responses to higher-level exposures must be extrapolated from odontocetes.

### **Mysticetes**

The responses of mysticetes to sonar and other duty-cycled tonal sounds are highly dependent upon the characteristics of the signal, the behavioral state of the animal, the particular sensitivity and previous experience of an individual, and other contextual factors including distance of the source, movement of the source, and the physical presence of vessels in addition to the sonar (Goldbogen et al., 2013; Harris et al., 2015; Martin et al., 2015; Sivle et al., 2015). Behavioral response studies have been conducted over a variety of contextual and behavioral states, helping to identify which contextual factors may lead to a response beyond just the received level of the sound. Observed reactions during behavioral response studies have not been consistent across individuals based on received sound levels alone, and likely were the result of complex interactions between these contextual factors.

Surface feeding blue whales did not show a change in behavior in response to mid-frequency simulated and real sonar sources with received levels between 90 and 179 dB re 1  $\mu$ Pa, 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. The behavioral

responses they observed were generally brief, of low to moderate severity, and highly dependent on exposure context (behavioral state, source-to-whale horizontal range, and prey availability) (DeRuiter et al., 2017; Goldbogen et al., 2013; Sivle et al., 2015; Southall et al., 2019c). Similarly, while the rates of foraging lunges decreased in humpback whales due to sonar exposure, there was variability in the response across individuals, with one animal ceasing to forage completely and another animal starting to forage during the exposure (Sivle et al., 2016). In addition, lunges decreased (although not significantly) during a no-sonar control vessel approach prior to the sonar exposure, and lunges decreased less during a second sonar approach than during the initial approach, possibly indicating some response to the vessel and some habituation to the sonar and vessel after repeated approaches. In the same experiment, most of the non-foraging humpback whales did not respond to any of the approaches (Sivle et al., 2016). These humpback whales also showed variable avoidance responses, with some animals avoiding the sonar vessel during the first exposure but not the second, while others avoided the sonar during the second exposure, and only one avoided both. In addition, almost half of the animals that avoided were foraging before the exposure but the others were not; the animals that avoided while not feeding responded at a slightly lower received level and greater distance than those that were feeding (Wensveen 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. In fact, when the prey field was mapped and used as a covariate in similar models looking for a response in the same blue whales, the response in deep-feeding behavior by blue whales was even more apparent, reinforcing the need for contextual variables to be included when assessing behavioral responses (Friedlaender et al., 2016). Further, it was found that the probability of a moderate behavioral response increased when the range to source was closer for these foraging blue whales, although there was a high degree of uncertainty in that relationship (Southall et al., 2019b). However, even when responses did occur the animals quickly returned to their previous behavior after the sound exposure ended (Goldbogen et al., 2013; Sivle et al., 2015). In another study, humpback whales exposed to a 3 kHz pinger meant to act as a net alarm to prevent entanglement did not respond or change course, even when within 500 m (Harcourt et al., 2014). However, five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives; in this case, the alarm was comprised of a mixture of signals with frequencies from 500 to 4,500 Hz, was long in duration (lasting several minutes), and was purposely designed to elicit a reaction from the animals as a prospective means to protect them from ship strikes (Nowacek et al., 2004). Although the animals' received SPL was similar in the latter two studies (133–150 dB re 1  $\mu\text{Pa}^2\text{s}$ ), the frequency, duration, and temporal pattern of signal presentation were different. Harris et al. (2019a) suggest that differences in responses between species may be due to contextual factors such as location, time of year, sound source characteristics, or exposure context through the comparison of differences in changes in lunge feeding between blue, fin, and humpback whales observed during sonar controlled exposure experiments.

Humpback whales in another behavioral response experiment in Australia also responded to a 2 kHz tone stimulus by changing their course during migration to move more offshore and surfaced more frequently, but otherwise did not respond (Dunlop et al., 2013). Humpback whales in the Norwegian behavioral response study may have habituated slightly between the first and second sonar exposure (Sivle et al., 2015), and actually responded more severely to killer whale vocalization playbacks than they did to the sonar playbacks. Several humpback whales have been observed during aerial or visual surveys during Navy training events involving sonar; no avoidance or other behavioral responses were ever noted, even when the whales were observed within 5 km of a vessel with active (or possibly active) sonar and maximum received levels were estimated to be between 135 and 161 dB re 1  $\mu\text{Pa}$  (Mobley,

2011; Mobley & Milette, 2010; Mobley & Pacini, 2012; Mobley et al., 2012; Smultea et al., 2009). In fact, one group of humpback whales approached a vessel with active sonar so closely that the sonar was shut down and the vessel slowed; the animals continued approaching and swam under the bow of the vessel (U.S. Department of the Navy, 2011b). Another group of humpback whales continued heading towards a vessel with active sonar as the vessel was moving away for almost 30 minutes, with an estimated median received level of 143 dB re 1  $\mu$ Pa. This group was observed producing surface active behaviors such as pec slaps, tail slaps, and breaches; however, these are very common behaviors in competitive pods during the breeding season and were not considered to have occurred in response to the sonar (Mobley et al., 2012). In addition, Henderson et al. (2019) examined the dive and movement behavior of humpback whales tagged at the U.S. Navy's Pacific Missile Range Facility, including whales incidentally exposed to sonar during Navy training activities. Tracking data showed that individual humpbacks spent limited time, no more than a few days, in the vicinity of Kaua'i. Potential behavioral responses to sonar exposure were limited and may have been influenced by engagement in breeding and social behaviors.

The strongest baleen whale response in any behavioral response study was observed in a minke whale in the 3S2 study, which responded at 146 dB re 1  $\mu$ Pa by strongly avoiding the sound source (Kvadsheim et al., 2017; Sivle et al., 2015). Although the minke whale increased its swim speed, directional movement, and respiration rate, none of these were greater than rates observed in baseline behavior, and its dive behavior remained similar to baseline dives. A minke whale tagged in the Southern California behavioral response study also responded by increasing its directional movement, but maintained its speed and dive patterns, and so did not demonstrate as strong of a response (Kvadsheim et al., 2017). In addition, the 3S2 minke whale demonstrated some of the same avoidance behavior during the controlled ship approach with no sonar, indicating at least some of the response was to the vessel (Kvadsheim et al., 2017). Martin et al. (2015) found that the density of calling minke whales was reduced during periods of Navy training involving sonar relative to the periods before training, and increased again in the days after training was completed. The responses of individual whales could not be assessed, so in this case it is unknown whether the decrease in calling animals indicated that the animals left the range, or simply ceased calling. Similarly, minke whale detections made using Marine Acoustic Recording Instruments off Jacksonville, FL, were reduced or ceased altogether during periods of sonar use (Norris et al., 2012b; U.S. Department of the Navy, 2013b), especially with an increased ping rate (Charif et al., 2015). Harris et al. (2019b) utilized acoustically generated minke whale tracks at the U.S. Navy's Pacific Missile Range Facility to statistically demonstrate changes in the spatial distribution of minke whale acoustic presence Before, During, and After surface ship mid-frequency active sonar training. The spatial distribution of probability of acoustic presence was different in the During phase compared to the Before phase, and the probability of presence at the center of ship activity for the During phase was close to zero for both years. The After phases for both years retained lower probabilities of presence suggesting the return to baseline conditions may take more than five days. The results show a clear spatial redistribution of calling minke whales during surface ship mid-frequency active sonar training, however a limitation of passive acoustic monitoring is that one cannot conclude if the whales moved away, went silent, or a combination of the two. Two minke whales also stranded in shallow water after the U.S. Navy training event in the Bahamas in 2000, although these animals were successfully returned to deep water with no physical examinations; therefore, no final conclusions were drawn on whether the sonar led to their stranding (Filadelfo et al., 2009a; Filadelfo et al., 2009b; U.S. Department of Commerce & U.S. Department of the Navy, 2001).

Baleen whales have also been exposed to lower and much higher frequency sonars, with the hypothesis that these whales may react more strongly to lower frequency sounds that overlap with their

vocalization range. One series of studies was undertaken in 1997–1998 pursuant to the Navy’s Low-Frequency Sound Scientific Research Program. The frequency bands of the low-frequency sonars used were between 100 and 500 Hz, with received levels between 115 and 150 dB re 1  $\mu$ Pa, and the source was always stationary. Fin and blue whales were targeted on foraging grounds, singing humpback whales were exposed on breeding grounds, and gray whales were exposed during migratory behavior. These studies found only short-term responses to low-frequency sound by some fin and humpback whales, including changes in vocal activity and avoidance of the source vessel, while other fin, humpback, and blue whales did not respond at all. When the source was in the path of migrating gray whales they changed course up to 2 km to avoid the sound, but when the source was outside their path, little response was observed although received levels were similar (Clark & Fristrup, 2001; Croll et al., 2001; Fristrup et al., 2003; Miller et al., 2000; Nowacek et al., 2007). Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were also not found to affect dive times of humpback whales in Hawaiian waters (Frankel & Clark, 2000). Frankel and Stein (2020) exposed migrating gray whales to moored-source IMAPS sonar transmissions in the 21–25 kHz frequency band (estimated RL = 148 dB re 1  $\mu$ Pa<sup>2</sup>) and showed that whales changed their path and moved closer to the shore when the vessel range was 1–2 km during sonar transmissions.

Opportunistic passive acoustic based studies have also detected behavioral responses to sonar, although definitive conclusions are harder to draw. 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 110–120 dB re 1  $\mu$ Pa (Melcón et al., 2012); however, without visual observations it is unknown whether there was another factor that contributed to the reduction in foraging calls, such as the presence of conspecifics. In another example, Risch et al. (2012, 2014) determined that humpback whale song produced in the Stellwagen Bank National Marine Sanctuary was reduced, and since the timing was concurrent with an Ocean Acoustic Waveguide Remote Sensing experiment occurring 200 km away, they concluded that the reduced song was a result of the Ocean Acoustic Waveguide Remote Sensing. However, Gong et al. (2014) analyzed the same data set while also looking at the presence of herring in the region, and found that the singing humpbacks were actually located on nearby Georges Bank and not on Stellwagen, and that the song rate in their data did not change in response to Ocean Acoustic Waveguide Remote Sensing, but could be explained by natural causes.

Although some strong responses have been observed in mysticetes to sonar and other transducers (e.g., the single minke whale), for the most part mysticete responses appear to be fairly moderate across all received levels. While some responses such as cessation of foraging or changes in dive behavior could carry short-term impacts, in all cases behavior returned to normal after the signal stopped. Mysticete responses also seem to be highly mediated by behavioral state, with no responses occurring in some behavioral states, and contextual factors and signal characteristics having more impact than received level alone. Many of the contextual factors resulting from the behavioral response studies (e.g., close approaches by multiple vessels or tagging) would never be introduced in real Navy training scenarios. While data are lacking 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 are likely to have similar responses to high-duty cycle sonars. Therefore, mysticete behavioral responses to Navy sonar will likely be a result of the animal’s behavioral state and prior experience rather than external variables such as ship proximity; thus, if significant behavioral responses occur they will likely be short term. In fact, no significant behavioral responses such as panic, stranding,

or other severe reactions have been observed during monitoring of actual training exercises (Smultea et al., 2009; U.S. Department of the Navy, 2011c, 2014a; Watwood et al., 2012).

### **Odontocetes**

Behavioral response studies have been conducted on odontocete species since 2007, with a focus on beaked whale responses to active sonar transmissions or controlled exposure playback of simulated sonar on various military ranges (Claridge et al., 2009; Defence Science and Technology Laboratory, 2007; Falcone et al., 2017; Henderson et al., 2015; Henderson et al., 2016; Isojunno et al., 2020; Manzano-Roth et al., 2016; Manzano-Roth et al., 2013; McCarthy et al., 2011; Moretti et al., 2009; Southall et al., 2014; Southall et al., 2013; Southall et al., 2015; Southall et al., 2012a; Southall et al., 2011; Southall et al., 2012b; Tyack et al., 2011). Through analyses of these behavioral response studies, a preliminary overarching effect of greater sensitivity to most anthropogenic exposures was seen in beaked whales compared to the other odontocetes studied (Southall et al., 2009).

Observed reactions by Blainville's, Cuvier's, and Baird's beaked whales to mid-frequency sonar sounds have included cessation of clicking, decline in group vocal periods, termination of foraging dives, changes in direction to avoid the sound source, slower ascent rates to the surface, longer deep and shallow dive durations, and other unusual dive behavior (Boyd et al., 2008; Defence Science and Technology Laboratory, 2007; DeRuiter et al., 2013b; Miller et al., 2015; Southall et al., 2011; Stimpert et al., 2014; Tyack et al., 2011). Similar responses have been observed in northern bottlenose whales, one of which conducted the longest and deepest dive on record for that species after the sonar exposure and continued swimming away from the source for over seven hours (Miller et al., 2015; Wensveen et al., 2019). Responses have occurred at received levels between 95 and 150 dB re 1  $\mu$ Pa. Many of these exposures occurred within 1–8 km of the focal animal, within a few hours of tagging the animal, and with one or more boats within a few kilometers to observe responses and record acoustic data. One Cuvier's beaked whale was also incidentally exposed to real Navy sonar located over 100 km away, and the authors did not detect similar responses at comparable received levels. Received levels from the mid-frequency active sonar signals from the controlled and incidental exposures were calculated as 84–144 and 78–106 dB re 1  $\mu$ Pa, respectively, indicating that context of the exposures (e.g., source proximity, controlled source ramp-up) may have been a significant factor in the responses to the simulated sonars (DeRuiter et al., 2013b). However, in a remote environment where sonar exposure is rare, similar responses in northern bottlenose whales were detected in whales up to 28 km away from the source at modeled received levels estimated at 117–126 dB re 1  $\mu$ Pa with no vessel nearby (von Benda-Beckmann et al., 2019; Wensveen et al., 2019). One northern bottlenose whale did approach the ship and circle the source, then resumed foraging after the exposure, but the source level was only 122 dB re 1  $\mu$ Pa.

Falcone et al. (2017) modeled deep and shallow dive durations, surface interval durations, and inter-deep dive intervals of Cuvier's beaked whales against predictor values that included helicopter dipping, mid-power mid-frequency active sonar and hull-mounted, high-power mid-frequency active sonar along with other, non-mid-frequency active sonar predictors. They found both shallow and deep dive durations to increase as the proximity to both mid- and high-powered sources decreased, and found surface intervals and inter-deep dive intervals to also increase in the presence of both types of sonars, although surface intervals shortened during periods of no mid-frequency active sonar. The responses to the mid-power mid-frequency active sonar at closer ranges were comparable to the responses to the higher Source Level ship sonar, again highlighting the importance of proximity. This study also supports context as a response factor, as helicopter dipping sonars are shorter duration and randomly located, so

more difficult for beaked whales to predict or track and therefore potentially more likely to cause a response, especially when they occur at closer distances (6–25 km in this study). Sea floor depths and quantity of light are also important variables to consider in Cuvier's beaked whale behavioral response studies, as their foraging dive depth increased with sea floor depth up to sea floor depths of 2,000 m. The fraction of time spent at foraging depths and likely foraging was greater at night, although they spent more time near the surface during the night as well, particularly on dark nights with little moonlight, likely avoiding predation by staying deeper during periods of bright lunar illumination (Barlow et al., 2020). Sonar occurred during 10 percent of the dives studied and had little effect on the resulting dive metrics. Watwood et al. (2017) found that helicopter dipping events occurred more frequently but with shorter durations than periods of hull-mounted sonar, and also found that the longer the duration of a sonar event, the greater reduction in detected Cuvier's beaked whale group dives. Therefore, when looking at the number of detected group dives there was a greater reduction during periods of hull-mounted sonar than during helicopter dipping sonar. Similar results were found by DiMarzio et al. (2019).

Long-term tagging work has demonstrated that the longer duration dives considered a behavioral response by DeRuiter et al. (2013b) fell within the normal range of dive durations found for eight tagged Cuvier's beaked whales on the Southern California Offshore Range (Schorr et al., 2014). However, the longer inter-deep dive intervals found by DeRuiter et al. (2013b), which were among the longest found by Schorr et al. (2014) and Falcone et al. (2017), could indicate a response to sonar. In addition, Williams et al. (2017) note that in normal deep dives or during fast swim speeds, beaked whales and other marine mammals use strategies to reduce their stroke rates, including leaping or wave surfing when swimming, and interspersing glides between bouts of stroking when diving. They determined that in the post-exposure dives by the tagged Cuvier's beaked whales described in DeRuiter et al. (2013b), the whales ceased gliding and swam with almost continuous strokes. This change in swim behavior was calculated to increase metabolic costs about 30.5 percent and increase the amount of energy expending on fast swim speeds from 27 to 59 percent of their overall energy budget. This repartitioning of energy was detected in the model up to 1.7 hours after the single sonar exposure. Therefore, while the overall post-exposure dive durations were similar, the metabolic energy calculated by Williams et al. (2017) was higher. However, Southall et al. (2019a) found that prey availability was higher in the western area of the Southern California Offshore Range where Cuvier's beaked whales preferentially occurred, while prey resources were lower in the eastern area and moderate in the area just north of the Range. This high prey availability may indicate that fewer foraging dives are needed to meet metabolic energy requirements than would be needed in another area with fewer resources.

Wensveen et al. (2019) examined the roles of sound source distance and received level in northern bottlenose whales in an environment without frequent sonar activity using controlled exposure experiments. They observed behavioral avoidance of the sound source over a wide range of distances (0.8–28 km) and estimated avoidance thresholds ranging from received SPLs of 117–126 dB re 1  $\mu$ Pa. The behavioral response characteristics and avoidance thresholds were comparable to those previously observed in beaked whale studies; however, they did not observe an effect of distance on behavioral response and found that onset and intensity of behavioral response were better predicted by received SPL. Joyce et al. (2019) examined modeled received sound levels, dive data, and horizontal movement of seven satellite-tagged Blainville's beaked whales before, during, and after mid-frequency active sonar training at the Atlantic Undersea Test and Evaluation Center instrumented range. They found a decline in deep dives at the onset of the training and an increase in time spent on foraging dives as individuals moved away from the range. Predicted received levels at which presumed responses were observed

were comparable to those previously observed in beaked whale studies. Acoustic data indicated that vocal periods were detected on the range within 72 hours after training ended.

On Navy ranges, Blainville's beaked whales located on the range appear to move off-range during sonar use and return only after the sonar transmissions have stopped, sometimes taking several days to do so (Claridge et al., 2009; Henderson et al., 2015; Manzano-Roth et al., 2016; McCarthy et al., 2011; Moretti et al., 2009; Tyack et al., 2011). For example, five Blainville's beaked whales that were estimated to be within 2–29 km of the Atlantic Undersea Test and Evaluation Center range at the onset of sonar were displaced a maximum of 28–68 km from the range after moving away from the range, although one whale approached the range during the period of active sonar (Joyce et al., 2019). However, Blainville's beaked whales remain on the range to forage throughout the rest of the year (Henderson et al., 2016), possibly indicating that this a preferred foraging habitat regardless of the effects of the noise, or it could be that there are no long-term consequences of the sonar activity. Similarly, photo-identification studies in the SOCAL Range Complex have identified approximately 100 individual Cuvier's beaked whale individuals, with 40 percent having been seen in one or more prior years, with re-sightings up to seven years apart, indicating a possibly resident population on the range (Falcone & Schorr, 2014; Falcone et al., 2009).

Beaked whales may respond similarly to shipboard echosounders, commonly used for navigation, fisheries, and scientific purposes, with frequencies ranging from 12 to 400 kHz and source levels up to 230 dB re 1  $\mu$ Pa but typically a very narrow beam (Cholewiak et al., 2017). During a scientific cetacean survey, an array of echosounders was used in a one-day-on, one-day-off paradigm. Beaked whale acoustic detections occurred predominantly (96 percent) when the echosounder was off, with only 4 detections occurring when it was on. Beaked whales were sighted fairly equally when the echosounder was on or off, but sightings were further from the ship when the echosounder was on (Cholewiak et al., 2017). These findings indicate that the beaked whales may be avoiding the area and may cease foraging near the echosounder. On the other hand, Varghese et al. (2020) analyzed group vocal periods from Cuvier's beaked whales during multibeam echosounder activity recorded in the Southern California Antisubmarine Warfare Range and failed to find any clear evidence of behavioral response due to the echosounder survey. The whales did not leave the range or cease foraging, and in fact group vocal periods increased during and after multibeam echosounder surveys. Since echosounders are highly directional and the sound doesn't propagate horizontally, the difference in these results may be due to the locations of beaked whales relative to the echosounder; in fact one of the surveys by Varghese et al. (2020) was largely conducted on a portion of the range little used by Cuvier's beaked whales.

Tyack et al. (2011) hypothesized that beaked whale responses to sonar may represent an anti-predator response. To test this idea, vocalizations of a potential predator—a killer whale—were also played back to a Blainville's beaked whale. This exposure resulted in a similar but more pronounced reaction than that elicited by sonar playback, which included longer inter-dive intervals and a sustained straight-line departure of more than 20 km from the area (Allen et al., 2014; Tyack et al., 2011). De Soto et al. (2020) hypothesized that the high degree of vocal synchrony in beaked whales during their deep foraging dives, coupled with their silent, low-angled ascents, have evolved as an anti-predator response to killer whales. Since killer whales do not dive deep when foraging and so may be waiting at the surface for animals to finish a dive, these authors speculated that by diving in spatial and vocal cohesion with all members of their group, and by surfacing silently and up to a km away from where they were vocally active during the dive, they minimize the ability of killer whales to locate them when at the surface. This may lead to a trade-off for the larger, more fit animals that could conduct longer foraging dives, such that all members



of the group remain together and are better protected by this behavior. The authors further speculate that this may explain the long, slow, silent, and shallow ascents that beaked whales make when sonar occurs during a deep foraging dive. However, these hypotheses are based only on the dive behavior of tagged beaked whales, with no observations of predation attempts by killer whales, and need to be tested further to be validated. This anti-predator hypothesis was also tested by playing back killer whale vocalizations to pilot whales, sperm whales, and even other killer whales, to determine responses by both potential prey and conspecifics (Miller, 2012; Miller et al., 2011). Results varied, from no response by killer whales to an increase in group size and attraction to the source in pilot whales (Curé et al., 2012). Gotz et al. (2020) tested startle responses in bottlenose dolphins and found that these responses can occur at moderate received levels and mid-frequencies, and that the relationship between rise time and startle response was more gradual than expected in an odontocete. They therefore hypothesize that the extreme responses of beaked whales to sonar could be a form of startle response, rather than an anti-predator response.

While there has been a focus on beaked whale responses to sonar, other species have been studied during behavioral response studies as well, including pilot whales, killer whales, and sperm whales. Responses by these species have also included horizontal avoidance, reduced breathing rates, changes in behavioral state, and changes in dive behavior (Antunes et al., 2014; Isojunno et al., 2018; Isojunno et al., 2017; Isojunno et al., 2020; Miller, 2012; Miller et al., 2011; Miller et al., 2014). Additionally, separation of a killer whale calf from its group during exposure to mid-frequency sonar playback was observed (Miller et al., 2011). Received level thresholds at the onset of avoidance behavior were generally higher for pilot whales (mean 150 dB re 1  $\mu$ Pa) and sperm whales (mean 140 dB re 1  $\mu$ Pa) than killer whales (mean 129 dB re 1  $\mu$ Pa) (Antunes et al., 2014; Miller, 2012; Miller et al., 2014). A close examination of the tag data from the Norwegian killer whales indicated that responses were mediated by behavior, signal frequency, or received sound energy. For example, killer whales only changed their dive behavior when doing deep dives at the onset of 1–2 kHz sonar (sweeping across frequencies), but did not change their dive behavior if they were deep-diving during 6–7 kHz sonar (sweeping across frequencies). Nor did they change their dive behavior if they were conducting shallow dives at the onset of either type of sonar. Similarly, pilot whales and sperm whales performed normal deep dives during 6–7 kHz sonar (and more deep foraging dives than during baseline for the pilot whales), while during 1–2 kHz sonar the pilot whales conducted fewer deep dives and the sperm whales performed shorter and shallower dives (Sivle et al., 2012). In addition, pilot whales were also more likely to respond to lower received levels when non-feeding than feeding during 6–7 kHz sonar exposures, but were more likely to respond at higher received levels when non-feeding during 1–2 kHz sonar exposures. Foraging time in pilot whales was reduced during the initial sonar exposure (both mid-frequency active sonar and low-frequency active sonar), with a concurrent increase in travel behavior; however, foraging increased again during subsequent exposures, potentially indicating some habituation (Isojunno et al., 2017). No reduction in foraging was observed during killer whale playbacks. Cessation of foraging appeared to occur at a lower received level of 145–150 dB re 1  $\mu$ Pa than had been observed previously for avoidance behavior (around 170 dB re 1  $\mu$ Pa; Antunes et al., 2014). Pilot whales also exhibited reduced breathing rates relative to their diving behavior when the low frequency active sonar levels were high (reaching 180 dB re 1  $\mu$ Pa), but only on the first sonar exposure; on subsequent exposures their breathing rates increased (Isojunno et al., 2018) indicating a change in response tactic with additional exposures. Furthermore, pilot whales exposed to a 38 kHz downward-facing echosounder did not change their dive and foraging behavior during exposure periods, although the animals' heading variance increased and fewer deep dives were conducted (Quick et al., 2017). In contrast, killer whales were more likely to

respond to either sonar type when non-feeding than when feeding (Harris et al., 2015). Sperm whales were exposed to pulsed active sonar (1-2 kHz) at moderate source levels and high source levels, as well as continuously active sonar at moderate levels for which the summed energy (SEL) equaled the summed energy of the high source level pulsed sonar (Isojunno et al., 2020). Foraging behavior did not change during exposures to moderate source level sonar, but non-foraging behavior increased during exposures to high source level sonar and to the continuous sonar, indicating that the energy of the sound (the sound exposure level) was a better predictor of response than SPL. However, the time of day of the exposure was also an important covariate in determining the amount of non-foraging behavior, as were order effects (e.g. the SEL of the previous exposure). These results again demonstrate that the behavioral state and environment of the animal mediates the likelihood of a behavioral response, as do the characteristics (e.g., frequency, energy level) of the sound source itself. Further, the highly flexible activity time budgets observed for pilot whales, with a large amount of time spent resting at the surface, may indicate context-dependency on some behaviors, such as the presence of prey driving periods of foraging. Therefore, that time may be more easily re-allocated to missed foraging opportunities, leading to less severe population consequences of periods of reduced foraging (Isojunno et al., 2017).

Other responses during behavioral response studies included the synchronization of pilot whale surfacings with sonar pulses during one exposure, possibly as a means of mitigating the sound (Wensveen et al., 2015), and mimicry of the sonar with whistles by pilot whales (Alves et al., 2014), false killer whales (DeRuiter et al., 2013a) and Risso's dolphins (Smultea et al., 2012). In contrast, in another study melon-headed whales had "minor transient silencing" (a brief, non-lasting period of silence) after each 6–7 kHz signal, and (in a different oceanographic region) pilot whales had no apparent response (DeRuiter et al., 2013a). The probability of detecting delphinid vocalizations (whistles, clicks, and buzzes) increased during periods of sonar relative to the period prior to sonar in a passive acoustic study using Marine Autonomous Recording Units in the Jacksonville Range Complex, while there was no impact of sonar to the probability of detecting sperm whale clicks (Charif et al., 2015; U.S. Department of the Navy, 2013a).

In addition, killer whale sighting data from the same region in Norway as the behavioral response study were used to compare the presence or absence of whales from other years against the period with sonar. The authors found a strong relationship between the presence of whales and the abundance of herring, and only a weak relationship between the whales and sonar activity (Kuningas et al., 2013). Baird et al. (2014; 2017; 2013) also tagged four shallow-diving odontocete species (rough-toothed dolphins, pilot whales, bottlenose dolphins, and false killer whales) in Hawaii off the Pacific Missile Range Facility before Navy training events. 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  $\mu$ Pa and distances from sonar sources ranged between 3.2 and 94.4 km. However, one pilot whale did have reduced dive rates (from 2.6 dives per hour before to 1.6 dives per hour during) and deeper dives (from a mean of 124 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 behavior-driven, and can be species and even exposure specific.

Other opportunistic observations of behavioral responses to sonar have occurred as well, although in those cases it is difficult to attribute observed responses directly to the sonar exposure, or to know exactly what form the response took. For example, both sperm and pilot whales potentially ceased sound production during the Heard Island feasibility test, with transmissions centered at 57 Hz and up to 220 dB re 1  $\mu$ Pa (Bowles et al., 1994), although it could not be determined whether the animals ceased sound production or left the area. In May 2003, killer whales in Haro Strait, Washington, exhibited what were believed by some observers to be aberrant behaviors, during which time the USS Shoup was in the vicinity and engaged in mid-frequency active sonar operations. Sound fields modeled for the USS Shoup transmissions (Fromm, 2009; National Marine Fisheries Service, 2005; U.S. Department of the Navy, 2004) estimated a mean received SPL of approximately 169 dB re 1  $\mu$ Pa at the location of the killer whales at the closest point of approach between the animals and the vessel (estimated SPLs ranged from 150 to 180 dB re 1  $\mu$ Pa). However, attributing the observed behaviors to any one cause is problematic given there were six nearby whale watch vessels surrounding the pod, and subsequent research has demonstrated that “Southern Residents modify their behavior by increasing surface activity (breaches, tail slaps, and pectoral fin slaps) and swimming in more erratic paths when vessels are close” (National Oceanic and Atmospheric Administration, 2014). 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. 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; Watkins & Schevill, 1975). The authors did not report received levels from these exposures and reported similar reactions from noise generated by banging on their boat hull; therefore, it was unclear if the sperm whales were reacting to the sonar signal itself or to a potentially new unknown sound in general.

During aerial and visual monitoring of Navy training events involving sonar, rough-toothed dolphins and unidentified dolphins were observed approaching the vessel with active sonar as if to bow ride, while spotted dolphins were observed nearby but did not avoid or approach the vessel (Mobley, 2011; U.S. Department of the Navy, 2011b; Watwood et al., 2012). During small boat surveys near the Southern California Offshore Range in southern California, more dolphins were encountered in June compared to a similar survey conducted the previous November after seven days of mid-frequency sonar activity; it was not investigated if this change was due to the sonar activity or was due to the poor weather conditions in November that may have prevented animals from being seen (Campbell et al., 2010). There were also fewer passive acoustic dolphin detections during and after longer sonar activities in the Mariana Islands Range Complex, with the post-activity absence lasting longer than the mean dolphin absence of two days when sonar was not present (Munger et al., 2014; Munger et al., 2015).

Acoustic harassment devices and acoustic deterrent devices, which transmit sound into the acoustic environment similar to Navy sources, have been used to deter marine mammals from fishing gear both to prevent entanglement and to reduce depredation (taking fish). These devices have been used successfully to deter harbor porpoises and beaked whales from getting entangled in fishing nets. For example, Kyhn et al. (2015) tested two types of pingers, one with a 10 kHz tone and one with a broadband 30–160 kHz sweep. Porpoise detection rates were reduced by 65 percent for the sweep and

40 percent for the tone, and while there was some gradual habituation after the first two to four exposures, longer term exposures (over 28 days) showed no evidence of additional habituation. Omeyer et al. (2020) also tested a 50–120 kHz pinger near harbor porpoise and found a 37 percent reduction in detections at the recorder near the pinger, but only a 9 percent reduction at a recorder 100 m away, indicating a response only occurred in relatively close proximity to the pinger. While clicking returned to normal levels as soon as the pinger was shut off (implying no long-term displacement), the response to the active pinger remained consistent over the nine-month study period, indicating no habituation occurred and the pingers remained an effective deterrent. Similarly, Kindt-Larsen et al. (2019) tested two pinger types in four configurations, and found that while both pingers effectively deterred harbor porpoises, their effect decreased with increasing distance (although their effective distance was limited to a few hundred m). In addition, a species' habituation to a pinger may occur with single tones but is less likely with a mixture of signals. Additionally, sperm whales in the Caribbean stopped vocalizing when presented with sounds from nearby acoustic pingers (Watkins & Schevill, 1975). However, acoustic harassment devices used to deter marine mammals from depredating long lines or aquaculture enclosures have proven less successful. For example, Tixier et al. (2014) used a 6.5 kHz pinger with a source level of 195 dB re 1  $\mu$ Pa on a longline to prevent depredation by killer whales, and although two groups of killer whales fled over 700 m away during the first exposure, they began depredating again after the third and seventh exposures, indicating rapid habituation. In a review of marine mammal deterrents, Schakner & Blumstein (2013) point out that both the characteristics of deterrents and the motivation of the animal play a role in the effectiveness of acoustic harassment devices. Deterrents that are strongly aversive or simulate a predator or are otherwise predictive of a threat are more likely to be effective, unless the animal habituates to the signal or learns that there is no true threat associated with the signal. In some cases net pingers may create a "dinner bell effect," where marine mammals have learned to associate the signal with the availability of prey (Jefferson & Curry, 1996; Schakner & Blumstein, 2013). This may be why net pingers have been more successful at reducing entanglements for harbor porpoise and beaked whales since these species are not depredating from the nets but are getting entangled when foraging in the area and are unable to detect the net (Carretta et al., 2008; Schakner & Blumstein, 2013). Niu et al. (2020; 2012) exposed captive dolphins to pulsed and continuous tonal signals to investigate acoustic deterrence. For all test frequencies, the dolphins increased surfacing distance relative to transducer, surfaced more often, and reduced clicks compared to baseline. Although some acclimatization was observed during daily tests, no habituation was observed over the full duration of the studies. Bowles and Anderson (2012) exposed a variety of species in captivity to novel objects, including a fishing net and anchor with line, both with and without a gillnet pinger. Responses varied broadly by species, with three species of pinniped showing mild avoidance of the net with the pinger. In contrast, the Pacific white-sided dolphin approached the gillnet without a pinger but avoided it completely when the pinger was added, and Commerson's dolphins demonstrated strong behavioral responses to the pinger including high speed swimming and other high energy behavior, increased use of a refuge pool, and increased rates of vocalizations. In further trials meant to test habituation, the Commerson's dolphins appeared to sensitize to the pinger instead, with even stronger aversive behavior.

Similarly, a 12 kHz acoustic harassment device intended to scare seals was ineffective at deterring seals but effectively caused avoidance in harbor porpoises out to over 500 m from the source, highlighting different species- and device-specific responses (Mikkelsen et al., 2017). Likewise, in a long term study of killer whale occurrence in inland waters off British Columbia, a region that had been used regularly from 1985 to 1993 showed a significant decrease in killer whale occurrence from 1993 to 1999 when

four acoustic deterrent devices were deployed on seal farms; during the same time frame there was no evidence in a reduction in seals in the same area, although they were the intended targets of the devices (Morton & Symonds, 2002). During the same time period, no reduction in killer whale occurrence was detected at an adjacent location, leading to the conclusion that the killer whales were avoiding the area ensounded by the deterrent devices. Once the devices were removed, the killer whales returned to the affected area in similar numbers as had previously occurred. Additional behavioral studies have been conducted with captive harbor porpoises using acoustic alarms, such as those used on fishing nets to help deter marine mammals from becoming caught or entangled (Kastelein et al., 2006; Kastelein et al., 2001). These studies have found that high-frequency sources with varied duration, interval, and sweep characteristics can prove to be effective deterrents for harbor porpoises (Kastelein et al., 2017d). Van Beest et al. (2017) modeled the long-term, population-level impacts of fisheries bycatch, pinger deterrents, and time-area closures on a population of harbor porpoises. They found that when pingers were used alone (in the absence of gillnets or time-area closures), the animals were deterred from the area often enough to cause a population-level reduction of 21 percent, greater even than the modeled level of current bycatch impacts. However, when the pingers were coupled with gillnets in the model, and time-area closures were also used (allowing a net- and pinger-free area for the porpoises to move into while foraging), the population only experienced a 0.8 percent decline even with current gillnet use levels. This demonstrates that, when used correctly, pingers can successfully deter porpoises from gillnets without leading to any negative impacts.

Controlled experiments have also been conducted on captive animals to estimate received levels at which behavioral responses occur. In one study, bottlenose dolphin behavioral responses were recorded when exposed to 3 kHz sonar-like tones between 115 and 185 dB re 1  $\mu$ Pa (Houser et al., 2013a), and in another study bottlenose dolphins and beluga whales were presented with one-second tones up to 203 dB re 1  $\mu$ Pa to measure TTS (Finneran et al., 2003a; Finneran et al., 2001; Finneran et al., 2005b; Finneran & Schlundt, 2004; Schlundt et al., 2000). During these studies, responses included changes in respiration rate, fluke slaps, and a refusal to participate or return to the location of the sound stimulus. This refusal included what appeared to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al., 2002; Schlundt et al., 2000). In the behavioral response study, bottlenose dolphins demonstrated a 50 percent probability of response at 172 dB re 1  $\mu$ Pa over 10 trials. In the TTS experiment, bottlenose dolphins exposed to one-second intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1  $\mu$ Pa; beluga whales did so at received levels of 180 to 196 dB re 1  $\mu$ Pa and above. In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway et al., 1997; Schlundt et al., 2000). While animals were commonly reinforced with food during these studies, the controlled environment and ability to measure received levels provide insight on received levels at which animals will behaviorally respond to noise sources.

Behavioral responses to a variety of sound sources have been studied in captive harbor porpoises, including acoustic alarms (Kastelein et al., 2006; Kastelein et al., 2001), emissions for underwater data transmission (Kastelein et al., 2005b), and tones, including 1–2 kHz and 6–7 kHz sweeps with and without harmonics (Kastelein et al., 2014c), 25 kHz with and without sidebands (Kastelein et al., 2015f; Kastelein et al., 2015g), and mid-frequency sonar tones at 3.5–4.1 kHz at 2.7 percent and 96 percent duty cycles (e.g., one tone per minute versus a continuous tone for almost a minute) (Kastelein et al., 2018b). Responses include increased respiration rates, more jumping, or swimming further from the source, but responses were different depending on the source. For example, harbor porpoises

responded to the 1–2 kHz upsweep at 123 dB re 1  $\mu$ Pa, but not to the downsweep or the 6–7 kHz tonal at the same level (Kastelein et al., 2014c). When measuring the same sweeps for a startle response, the 50 percent response threshold was 133 and 101 dB re 1  $\mu$ Pa for 1–2 kHz and 6–7 kHz sweeps, respectively, when no harmonics were present, and decreased to 90 dB re 1  $\mu$ Pa for 1–2 kHz sweeps with harmonics present (Kastelein et al., 2014c). Harbor porpoises did not respond to the low-duty cycle mid-frequency tones at any received level, but one did respond to the high-duty cycle signal with more jumping and increased respiration rates (Kastelein et al., 2018b). Harbor porpoises responded to seal scarers with broadband signals up to 44 kHz with a slight respiration response at 117 dB re 1  $\mu$ Pa and an avoidance response at 139 dB re 1  $\mu$ Pa, but another scarer with a fundamental (strongest) frequency of 18 kHz did not have an avoidance response until 151 dB re 1  $\mu$ Pa (Kastelein et al., 2015e). Exposure of the same acoustic pinger to a striped dolphin under the same conditions did not elicit a response (Kastelein et al., 2006), again highlighting the importance in understanding species differences in the tolerance of underwater noise, although sample sizes in these studies was small so these could reflect individual differences as well. Lastly, Kastelein et al. (2019a) examined the potential masking effect of high sea state ambient noise on captive harbor porpoise perception of and response to high duty cycle playbacks of AN/SQS-53C sonar signals by observing their respiration rates. Results indicated that sonar signals were not masked by the high sea state noise, and received levels at which responses were observed were similar to those observed in prior studies of harbor porpoise behavior.

Behavioral responses by odontocetes to sonar and other transducers appear to range from no response at all to responses that could potentially lead to long-term consequences for individual animals (e.g., mother-calf separation). This is likely in part due to the fact that this taxonomic group is so broad and includes some of the most sensitive species (e.g., beaked whales and harbor porpoise) as well as some of the least sensitive species (e.g., bottlenose dolphins). This is also the only group for which both field behavioral response studies and captive controlled exposure experiments have been conducted, leading to the assessment of both contextually driven responses as well as dose-based responses. This wide range in both exposure situations and individual- and species-sensitivities makes reaching general conclusions difficult. However, it does appear as though exposures in close proximity, with multiple vessels that approach the animal lead to higher-level responses in most odontocete species regardless of received level or behavioral state. In contrast, in more “real-world” exposure situations, with distant sources moving in variable directions, behavioral responses appear to be driven by behavioral state, individual experience or species-level sensitivities. These responses may also occur more in-line with received level such that the likelihood of a response would increase with increased received levels. However, these “real-world” responses are more likely to be short term, lasting the duration of the exposure or even shorter as the animal assesses the sound and (based on prior experience or contextual cues) determines a threat is unlikely. Therefore, while odontocete behavioral responses to Navy sonar will vary across species, populations, and individuals, they are not likely to lead to long-term consequences or population-level effects.

### **Pinnipeds**

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 & Janik, 2010). Captive seals received food (reinforcement) during sound playback, while wild seals were exposed opportunistically. These results indicate that motivational state (e.g., reinforcement via food acquisition) can be a factor in whether or not an animal tolerates or habituates to novel or unpleasant sounds. Another study found that captive hooded seals

reacted to 1–7 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  $\mu$ Pa (Kvadsheim et al., 2010b); however, the animals adapted to the sound and did not show the same avoidance behavior upon subsequent exposures. Captive harbor seals responded differently to three signals at 25 kHz with different waveform characteristics and duty cycles. The seals responded to the frequency modulated signal at received levels over 137 dB re 1  $\mu$ Pa 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  $\mu$ Pa) (Kastelein et al., 2015d). Captive California sea lions were exposed to mid-frequency sonar at various received levels (125–185 dB re 1  $\mu$ Pa) during a repetitive task (Houser et al., 2013a). Behavioral responses included a refusal to participate, hauling out, an increase in respiration rate, and an increase in the time spent submerged. Young animals (less than two years old) were more likely to respond than older animals. Dose-response curves were developed both including and excluding those young animals. The majority of responses below 155 dB re 1  $\mu$ Pa were changes in respiration, whereas over 170 dB re 1  $\mu$ Pa 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  $\mu$ Pa, 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  $\mu$ Pa and demonstrated minor responses by occasionally hauling out at 128–138 dB re 1  $\mu$ Pa (Kastelein et al., 2015c). 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 & 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  $\mu$ Pa at 1 m, but the 8 kHz tone and 1–4 kHz sweep at source levels of 165 dB re 1  $\mu$ Pa caused the sea lions to haul out (Akamatsu et al., 1996).

Similar to the other taxonomic groups assessed, pinniped behavioral responses to sonar and other transducers seem to be mediated by the contextual factors of the exposure, including the proximity of the source, the characteristics of the signal, and the behavioral state of the animal. However, all pinniped behavioral response studies have been conducted in captivity, so while these results may be broadly applied to real-world exposure situations, it must be done with caution. Based on exposures to other sound sources in the wild (e.g., impulsive sounds and vessels), pinnipeds are not likely to respond strongly to Navy sonar that is not in close proximity to the animal or approaching the animal.

### **Sea Otters**

There is no research on the effects of sonar on sea otters. Sea otters spend approximately 80 percent of their time on the surface of the water (Curland, 1997) with their heads above the surface, which reduces their exposure to underwater sounds. They may show similar reactions to those of pinnipeds which are also amphibious hearers. However, underwater hearing sensitivities are significantly reduced in sea otters when compared to pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), so any reactions may have lower overall severity. Pinnipeds may haul out, swim faster, or increase their respiration rate in response to sonar (Houser et al., 2013a; Kastelein et al., 2015d). Pinnipeds also showed that they may avoid an

area temporarily, but may habituate to sounds quickly (Kvadsheim et al., 2010a; Kvadsheim et al., 2010b). Deviations from pinniped behavior could be a result of sea otter dives being energetically costly (i.e., requiring twice the metabolic energy that phocid seals need to dive). Therefore, sea otters may not dive or travel far in response to disturbance, as they already require long periods of rest at the surface to counterbalance the high metabolic cost of foraging at sea (Yeates et al., 2007). Sea otters may also habituate to sonar signals. However, the typical sea otter habitat (water less than 100 m in depth) is far inshore of the TMAA and the location for most Navy activities and so sea otters are unlikely be exposed to or impacted by Navy use of sonar or other transducers.

### **Behavioral Reactions to Vessel Noise**

Sound emitted from large vessels, such as cargo ships, is the principal source of low-frequency noise in the ocean today, and marine mammals are known to react to or be affected by that noise (Erbe et al., 2019; Hatch & Wright, 2007; Hildebrand, 2005; Richardson et al., 1995b). For example, Erbe et al. (2012) estimated the maximum annual underwater SEL from vessel traffic near Seattle was 215 dB re 1  $\mu\text{Pa}^2\text{s}$ , and Bassett et al. (2010) measured mean SPLs at Admiralty Inlet from commercial shipping at 117 dB re 1  $\mu\text{Pa}$  with a maximum exceeding 135 dB re 1  $\mu\text{Pa}$  on some occasions. Similarly, Veirs et al. (2015) found average broadband noise levels in Haro Strait to be 110 dB re 1  $\mu\text{Pa}$  that extended up to 40 kHz, well into the hearing range of odontocetes.

Many studies of behavioral responses by marine mammals to vessels have been focused on the short-and long-term impacts of whale watching vessels. In short-term studies, researchers noted changes in resting and surface behavior states of cetaceans to whale watching vessels (Acevedo, 1991; Aguilar de Soto et al., 2006; Arcangeli & Crosti, 2009; Au & Green, 2000; Christiansen et al., 2010; Erbe, 2002; Noren et al., 2009; Stockin et al., 2008; Williams et al., 2009). Received levels were often not reported so it is difficult to distinguish responses to the presence of the vessel from responses to the vessel noise. Most studies examined the short-term response to vessel sound and vessel traffic (Magalhães et al., 2002; Richardson et al., 1995b; Watkins, 1981), with behavioral and vocal responses occurring when received levels were over 20 dB greater than ambient noise levels. Other research has attempted to quantify the effects of whale watching using focused experiments (Meissner et al., 2015; Pirodda et al., 2015b).

The impact of vessel noise has received increased consideration, particularly as whale watching and shipping traffic has risen (McKenna et al., 2012; Pirodda et al., 2015b; Veirs et al., 2015). Odontocetes and mysticetes in particular have received increased attention relative to vessel noise and vessel traffic, with pinnipeds and sea otters less so. The impacts of ship noise on marine mammals also appear to be largely context- and species-dependent (Erbe et al., 2019). Still, not all species in all taxonomic groups have been studied, and so results do have to be extrapolated across these broad categories in order to assess potential impacts.

### **Mysticetes**

Baleen whales 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 (Baker et al., 1983; Fiori et al., 2019; Gende et al., 2011; Watkins, 1981). Other common responses include changes in vocalizations, call rate, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au & Green, 2000; Dunlop, 2019; Fournet et al., 2018; Machernis et al., 2018; Richter et al., 2003; Williams et al., 2002a).



The likelihood of response may be driven by the distance, speed, approach, or noise level of the vessel, the animal's behavioral state, or by the prior experience of the individual or population. For example, in one study fin and humpback whales largely ignored vessels that remained 100 m or more away (Watkins, 1981). In another study, minke whales in the Antarctic did not show any apparent response to a survey vessel moving at normal cruising speeds (about 12 knots) at a distance of 5.5 NM. However, when the vessel drifted or moved at very slow speeds (about 1 knot), many whales approached it (Leatherwood et al., 1982). Similarly, Bernasconi et al. (2012) observed the reactions of six individual baleen whales of unknown species at distances of 50–400 m from a fishing vessel conducting an acoustic survey of pelagic fisheries, with only a slight change in swim direction when the vessel began moving around the whales. Gray whales were likely to continue feeding when approached by a vessel in areas with high motorized vessel traffic, but in areas with less motorized vessel traffic they were more likely to change behaviors, either indicating habituation to vessels in high traffic area, or indicating possible startle reactions to close-approaching non-motorized vessels (e.g., kayaks) in quieter areas (Sullivan & Torres, 2018). Changes in behavior of humpback whales when vessels came within 500 m were also dependent on behavioral state such that they would keep feeding but were more likely to start traveling if they were surface active when approached (Di Clemente et al., 2018). Changes in humpback whale behavior were also affected by time of day, season, or the type of vessel approach (Di Clemente et al., 2018; Fiori et al., 2019). Avoidance responses occurred most often after “J” type vessel approaches (i.e., traveling parallel to the whales' direction of travel, then overtaking the whales by turning in front of the group) compared to parallel or direct approaches; mother humpbacks were particularly sensitive to direct and J type approaches and spent significantly more time diving in response (Fiori et al., 2019). Humpback whales changed their acoustic and social behavior when vessels were present; their communication area was reduced by half in average vessel-dominated noise (105 dB re 1  $\mu$ Pa), but the physical presence of vessels was the major contributing factor to decreased social interactions (Dunlop, 2019). In contrast, for resting humpback whale mother-calf pairs, the presence of a passing vessel did not change their behavior, but fast vessels with louder low-frequency weighted source levels of 173 dB re 1  $\mu$ Pa, equating to weighted received levels of 133 dB re 1  $\mu$ Pa at an average distance of 100 m, led to a decrease in resting behavior and increase in dives, swim speeds, and respiration rates (Sprogis et al., 2020). Migrating humpback whales reacted similarly to vessels towing seismic air gun arrays, regardless of whether the air guns were active or not; this indicates that it was the presence of ships (rather than the active air guns) that reduced social interactions between males and mother-calf pairs (Dunlop et al., 2020).

In response to an approaching large commercial vessel in an area of high ambient noise levels (125–130 dB re 1  $\mu$ Pa), a tagged female blue whale turned around mid-ascent and descended perpendicular to the ship's path (Szesciorka et al., 2019). The whale did not respond until the ship's closest point of approach (100 m distance, 135 dB re 1  $\mu$ Pa), which was only 10 dB above the ambient noise levels. After the ship passed, the whale ascended to the surface again with a three-minute delay. However, other species of mysticete have demonstrated their lack of reaction to vessel noise. Sei whales have been observed ignoring the presence of vessels entirely and even passing close to the vessel (Reeves et al., 1998), and North Atlantic right whales tend not to respond to the sounds of oncoming vessels and continue to use habitats in high vessel traffic areas (Nowacek et al., 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves. This lack of response may be due to habituation to the presence and associated noise of vessels in right whale habitat, or may be due to propagation effects that may attenuate vessel noise near the surface (Nowacek et al., 2004; Terhune & Verboom, 1999).

When baleen whales do respond to vessels, responses can be as minor as a change in breathing patterns (e.g., Baker et al., 1983; Jahoda et al., 2003), or can be evidenced by a decrease in overall presence, as was observed during a construction project in the United Kingdom, when fewer minke whales were observed as vessel traffic increased (Anderwald et al., 2013). Avoidance responses can be as simple as an alteration in swim patterns or direction by increasing speed and heading away from the vessel (Jahoda et al., 2003), or by increasing swim speed, changing direction to avoid, and staying submerged for longer periods of time (Au & Green, 2000). For example, in the presence of approaching vessels, blue whales perform shallower dives accompanied by more frequent surfacing but otherwise do not exhibit strong reactions (Calambokidis et al., 2009). In another study in Hawaii, humpback whales exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 2,000 m and 4,000 m away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were less than 2,000 m away (Baker et al., 1983). Similarly, humpback whales in Australia demonstrated variable responses to whale watching vessels, including both horizontal avoidance, approaching, and changes in dive and surface behavior (Stamation et al., 2010). Humpback whales demonstrated similar responses to tourist vessels in Alaska, with increased respiration rates when the time spent near vessels increased, increased swim speeds and more non-linear movement (Schuler et al., 2019). In addition, while foraging and traveling behavior states were likely to be maintained in the presence of tourist vessels, surface active behavior was more likely to transition to traveling behavior. Humpback whales avoided a Navy vessel by increasing their dive times and decreasing respiration rates at the surface (Smultea et al., 2009). Williamson et al. (2016) specifically looked at close approaches to humpback whales by small research boats for the purposes of tagging. They found that while dive behavior did not change for any groups, some groups did increase their speed and change their course during or right after the approach, but resumed pre-approach speed and heading shortly thereafter. Only mother-calf groups were found to increase their speed during the approach and maintain the increased speed for longer after the approach, but these groups too resumed normal swim speeds after about 40 minutes. It should be noted that there were no responses by any groups that were approached closely but with no attempts at tagging, indicating that the responses were not due to the vessel presence but to the tagging attempt. In addition, none of the observed changes in behavior were outside the normal range of swim speeds or headings for these migrating whales.

Mysticetes have been shown to both increase and decrease calling behavior in the presence of vessel noise. Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcón et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. While humpback whale call repetition and rate has increased in association with high vessel noise (Doyle et al., 2008), a recent study with stringent inclusion criteria found that the probability of humpback whale calls decreased as vessel noise increased (Fournet et al., 2018). The amplitude of humpback whale calls did not change in the absence or presence of vessel noise. However, feeding calls increased amplitude with higher levels of any (i.e., weather or vessel) ambient noise (Fournet et al., 2018). Boat traffic has been a cause of decreased humpback song activity near Brazil (Sousa-Lima & Clark, 2008), and decreased frequency parameters of fin whale calls (Castellote et al., 2012). Bowhead whales avoided the area around icebreaker ship noise and increased their time at the surface and number of blows (Richardson et al., 1995a). Right whales increase the amplitude or frequency of their vocalizations or call at a lower rate in the presence of increased vessel noise (Parks et al., 2007; Parks et al., 2011), and these vocalization changes may persist over long periods if background noise levels remained elevated. Humpback whales increase the source levels of their calls with increased ambient noise levels that

include vessel noise, but the probability of calling is also decreased when vessel noise was part of the soundscape (Fournet et al., 2018).

The long-term consequences of vessel noise are not well understood (see Section 3.8.3.1.1.7, Long-Term Consequences). In a short-term study, minke whales on feeding grounds in Iceland responded to increased whale watching vessel traffic with a decrease in foraging, both during deep dives and at the surface (Christiansen et al., 2013). They also increased their avoidance of the boats while decreasing their respiration rates, likely leading to an increase in their metabolic rates. Christiansen and Lusseau (2015) and Christiansen et al. (2014) followed up this study by modeling the cumulative impacts of whale watching boats on minke whales, but found that although the boats cause temporary feeding disruptions, there were not likely to be long-term consequences as a result. This suggests that short-term responses may not lead to long-term consequences and that over time animals may habituate to the presence of vessel traffic. However, in an area of high whale watch activity, vessels were within 2,000 m of blue whales 70 percent of the time, with a maximum of 8 vessels observed within 400 m of one whale at the same time. This study found reduced surface time, fewer breaths at the surfaced, and shorter dive times when vessels were within 400 m (Lesage et al., 2017). Since blue whales in this area forage 68 percent of the time, and their foraging dive depths are constrained by the location of prey patches, these reduced dive durations may indicate reduced time spent foraging by over 36 percent. In the short term this reduction may be compensated for, but prolonged exposure to vessel traffic could lead to long-term consequences. Using historical records, Watkins (1986) showed that the reactions of four species of mysticetes to vessel traffic and whale watching activities in Cape Cod had changed over the 25-year period examined (1957–1982). Reactions of minke whales changed from initially more positive reactions, such as coming towards the boat or research equipment to investigate, to more uninterested reactions towards the end of the study. Fin whales, the most numerous species in the area, showed a trend from initially more negative reactions, such as swimming away from the boat with limited surfacing, to more uninterested reactions (ignoring) allowing boats to approach within 30 m. Right whales showed little change over the study period, with a roughly equal number of reactions judged to be negative and uninterested; no right whales were noted as having positive reactions to vessels. Humpback whales showed a trend from negative to positive reactions with vessels during the study period. The author concluded that the whales had habituated to the human activities over time (Watkins, 1986).

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. When 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 fact, in many cases the 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. Although a lack of response in the presence of a vessel may minimize potential disturbance from passing ships, it does increase the whales' vulnerability to vessel strike, which may be of greater concern for baleen whales than vessel noise.

### **Odontocetes**

Most odontocetes react neutrally to vessels, although both avoidance and attraction behavior have been observed (Hewitt, 1985; Würsig et al., 1998). Würsig et al. (1998) found that Kogia whales and beaked whales were the most sensitive species to vessels, and reacted by avoiding marine mammal survey vessels in 73 percent of sightings, more than any other odontocetes. Avoidance reactions include

a decrease in resting behavior or change in travel direction (Bejder et al., 2006a). Incidents of attraction include common, rough-toothed, and bottlenose dolphins bow riding and jumping in the wake of a vessel (Norris & Prescott, 1961; Ritter, 2002; Shane et al., 1986; Würsig et al., 1998). A study of vessel reactions by dolphin communities in the eastern tropical Pacific found that populations that were often the target of tuna purse-seine fisheries (spotted, spinner, and common dolphins) show evasive behavior when approached; however, populations that live closer to shore (within 100 NM; coastal spotted and bottlenose dolphins) that are not set on by purse-seine fisheries tend to be attracted to vessels (Archer et al., 2010). The presence of vessels has also been shown to interrupt feeding behavior in delphinids (Meissner et al., 2015; Pirotta et al., 2015b).

Short-term displacement of dolphins due to tourist boat presence has been documented (Carrera et al., 2008), while longer term or repetitive/chronic displacement for some dolphin groups due to chronic vessel noise has been noted (Haviland-Howell et al., 2007). Delphinid behavioral states also change in the presence of tourist boats that often approach animals, with travel and/or resting increasing and foraging and social behavior decreasing (Cecchetti et al., 2017; Clarkson et al., 2020; Kassamali-Fox et al., 2020; Meissner et al., 2015). Most studies of the behavioral reactions to vessel traffic of bottlenose dolphins have documented at least short-term changes in behavior, activities, or vocalization patterns when vessels are near, although the distinction between vessel noise and vessel movement has not been made clear (Acevedo, 1991; Arcangeli & Crosti, 2009; Berrow & Holmes, 1999; Fumagalli et al., 2018; Gregory & Rowden, 2001; Janik & Thompson, 1996; Lusseau, 2004; Marega et al., 2018; Mattson et al., 2005; Scarpaci et al., 2000). Steckenreuter (2011) found bottlenose dolphin groups to feed less, become more tightly clustered, and have more directed movement when approached to 50 m than groups approached to 150 m or approached in a controlled manner. Guerra et al. (2014) demonstrated that bottlenose dolphins subjected to chronic noise from tour boats responded to boat noise by alterations in group structure and in vocal behavior but also found the dolphins' reactions varied depending on whether the observing research vessel was approaching or moving away from the animals being observed. This demonstrates that the influence of the sound exposure cannot be decoupled from the physical presence of a surface vessel, thus complicating interpretations of the relative contribution of each stimulus to the response. Indeed, the presence of surface vessels, their approach, and speed of approach, seemed to be significant factors in the response of the Indo-Pacific humpback dolphins (Ng & Leung, 2003).

The effects of tourism and whale watching have highly impacted killer whales, such as the Northern and Southern Resident populations. These animals are targeted by numerous small whale watching vessels in the Pacific Northwest and, from 1998 to 2012 during the viewing season, have had an annual monthly average of nearly 20 vessels of various types within 0.5 miles of their location during daytime hours (Clark, 2015; Eisenhardt, 2014; Erbe et al., 2014). These vessels have source levels that ranged from 145 to 169 dB re 1  $\mu$ Pa and produce broadband noise up to 96 kHz. While new regulations on the distance boats had to maintain were implemented, there did not seem to be a concurrent reduction in the received levels of vessel noise, and noise levels were found to increase with more vessels and faster moving vessels (Holt et al., 2017). These noise levels have the potential to result in behavioral disturbance, interfere with communication, and affect the killer whales' hearing capabilities via masking (Erbe, 2002; Veirs et al., 2015). Killer whales foraged significantly less and traveled significantly more when boats were within 100 m of the whales (Kruse, 1991; Lusseau et al., 2009; Trites & Bain, 2000; Williams et al., 2002a; Williams et al., 2009; Williams et al., 2002b). These short-term feeding activity disruptions may have important long-term population-level effects (Lusseau et al., 2009; Noren et al., 2009). As with other delphinids, the reaction of the killer whales to whale watching vessels may be in

response to the vessel pursuing them rather than to the noise of the vessel itself, or to the number of vessels in their proximity. Williams et al. (2014b) modeled behavioral responses of killer whales to vessel traffic by looking at their surface behavior relative to the received level of three large classes of ships. The authors found that the severity of the response was largely dependent on seasonal data (e.g., year and month) as well as the animal's prior experience with vessels (e.g., age and sex), and the number of other vessels present, rather than the received level of the larger ships (Williams et al., 2014b).

Sperm whales generally react only to vessels approaching within several hundred m; however, some individuals may display avoidance behavior, such as quick diving (Magalhães et al., 2002; Würsig et al., 1998) or a decrease in time spent at the surface (Isojunno & Miller, 2015). One study showed that after diving, sperm whales showed a reduced timeframe before they emitted the first click than prior to a vessel interaction (Richter et al., 2006). Smaller whale watching and research vessels generate more noise in higher frequency bands and are more likely to approach odontocetes directly, and to spend more time near an individual whale. 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. It is unknown whether the whales left the area, ceased to click, or surfaced during this period. However, some of the reduction in click detections may be due to masking of the clicks by the vessel noise, particularly during the closest point of approach.

Little information is available on the behavioral impacts of vessels or vessel noise on beaked whales (Cox et al., 2006), although it seems most beaked whales react negatively to vessels by quick diving and other avoidance maneuvers (Würsig et al., 1998). Limited evidence suggests that beaked whales respond to vessel noise, anthropogenic noise in general, and mid-frequency sonar at similar sound levels (Aguilar de Soto et al., 2006; Tyack et al., 2011; Tyack, 2009). An observation of vocal disruption of a foraging dive by a Cuvier's beaked whale when a large, noisy vessel passed suggests that some types of vessel traffic may disturb foraging beaked whales (Aguilar de Soto et al., 2006). Tyack et al. (2011) noted the result of a controlled exposure to pseudorandom noise suggests that beaked whales would respond to vessel noise at similar received levels to those noted previously for mid-frequency sonar. Pirotta et al. (2012) found that while the distance to a vessel did not change the duration of a foraging dive, the proximity of the vessel may have restricted the movement of the group. The maximum distance at which this change was significant was 5.2 km, with an estimated received level of 135 dB re 1  $\mu$ Pa.

Small dolphins and porpoises may also be more sensitive to vessel noise. Both finless porpoises (Li et al., 2008) and harbor porpoises (Polacheck & Thorpe, 1990) routinely avoid and swim away from large motorized vessels, and harbor porpoises may click less when near large ships (Sairanen, 2014). A resident population of harbor porpoise in Swansea Bay are regularly near vessel traffic, but only 2 percent of observed vessels had interactions with porpoises in one study (Oakley et al., 2017). Of these, 74 percent of the interactions were neutral (no response by the porpoises) while vessels were 10 m–1 km away. Of the 26 percent of interactions in which there was an avoidance response, most were observed in groups of 1–2 animals to fast-moving or steady plane-hulling motorized vessels. Larger groups reacted less often, and few responses were observed to non-motorized or stationary vessels. Another study found that when vessels were within 50 m, harbor porpoises had an 80 percent probability of changing their swimming direction when vessels were fast moving; this dropped to 40 percent probability when vessels were beyond 400 m (Akkaya Bas et al., 2017). These porpoises also demonstrated a reduced proportion of feeding and shorter behavioral bout durations in general, if vessels were in close proximity, 62 percent of the time. Although most vessel noise is constrained to lower frequencies below 1 kHz, at close range vessel noise can extend into mid- and high-frequencies

(into the tens of kHz) (Hermannsen et al., 2014; Li et al., 2015); these frequencies are what harbor porpoises are likely responding to, at M-weighted received SPLs with a mean of 123 dB re 1  $\mu$ Pa (Dyndo et al., 2015). Foraging harbor porpoises also have fewer prey capture attempts and have disrupted foraging when vessels pass closely and noise levels are higher (Wisniewska et al., 2018). Hermannsen et al. (2019) estimated that noise in the 16 kHz frequency band resulting from small recreational vessels not equipped with an Automatic Identification System and therefore not included in most vessel noise impact models could be elevated up to 124 dB re 1  $\mu$ Pa and raise ambient levels up to 51 dB; these higher levels were associated with vessel speed and range. Using the threshold levels found by Dyndo et al. (2015) and Wisniewska et al. (2018), these authors determined that recreational vessel noise in the 16 kHz band could cause behavioral responses in harbor porpoises, and that those thresholds were exceeded by 49–85 percent of high noise events.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado & Wartzok, 2008), with whistle frequency increasing in the presence of low-frequency noise and whistle frequency decreasing in the presence of high-frequency noise (Gospić & Picciulin, 2016). For example, bottlenose dolphins in Portuguese waters decrease their call rates and change the frequency parameters of whistles in the presence of boats (Luís et al., 2014), while dolphin groups with calves increase their whistle rates when tourist boats are within 200 m and when the boats increase their speed (Guerra et al., 2014). Likewise, modification of multiple vocalization parameters was shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al., 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al., 2005). Killer whales are also known to modify their calls during increased noise. For example, the source level of killer whale vocalizations was shown to increase with higher background noise levels associated with vessel traffic (the Lombard effect) (Holt et al., 2008). In addition, calls with a high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al., 2011). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. This type of change has been observed in killer whales off the northwestern coast of the United States between 1973 and 2003. This population increased the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which is suggested as being a long-term response to increased masking noise produced by the vessels (Foote et al., 2004).

The long-term and cumulative implications of ship sound on odontocetes is largely unknown (National Academies of Sciences Engineering and Medicine, 2017; National Marine Fisheries Service, 2007a), although some long-term consequences have been reported (Lusseau & Bejder, 2007). Repeated exposure to acoustic and other anthropogenic stimuli has been studied in several cases, especially as related to vessel traffic and whale watching. Common dolphins in New Zealand responded to dolphin-watching vessels by interrupting foraging and resting bouts, and took longer to resume behaviors in the presence of the vessel (Stockin et al., 2008). The authors speculated that repeated interruptions of the dolphins' foraging behaviors could lead to long-term implications for the population. Bejder et al. (2006a) studied responses of bottlenose dolphins to vessel approaches and found stronger and longer lasting reactions in populations of animals that were exposed to lower levels of vessel traffic overall. The authors indicated that lesser reactions in populations of dolphins regularly subjected to high levels of

vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity.

Similar to mysticetes, odontocete responses to vessel noise are varied, although many odontocete species seem to be more sensitive to vessel presence and vessel noise, and these two factors are difficult to tease apart. Some species, in particular killer whales and porpoises, may be sensitized to vessels and respond at further distances and lower received levels than other delphinids. In contrast, many odontocete species also approach vessels to bow ride, indicating either that these species are less sensitive to vessels, or that the behavioral drive to bow ride supersedes any impact of the associated noise. With these broad and disparate responses, it is difficult to assess the impacts of vessel noise on odontocetes.

### **Pinnipeds**

Pinniped reactions to vessels are variable and reports include a wide spectrum of possibilities from avoidance and alert, to cases where animals in the water are attracted, and cases on land where there is lack of significant reaction suggesting habituation to or tolerance of vessels (Richardson et al., 1995b). Specific case reports in Richardson et al. (1995b) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007), pinniped responses to vessels are affected by the context of the situation and by the animal's experience.

Anderwald et al. (2013) investigated grey seal reactions to an increase in vessel traffic off Ireland's coast in association with construction activities, and their data suggest the number of vessels had an indeterminate effect on the seals' presence. Harbor seals haul out on tidewater glaciers in Alaska, and most haul outs occur during pupping season. Blundell & Pendleton (2015) found that the presence of any vessel reduces haul out time, but cruise ships and other large vessels in particular shorten haul out times. Another study of reactions of harbor seals hauled out on ice to cruise ship approaches in Disenchantment Bay, Alaska, revealed that animals are more likely to flush and enter the water when cruise ships approach within 500 m and four times more likely when the cruise ship approaches within 100 m (Jansen et al., 2010). Karpovich et al. (2015) also found that harbor seal heart rates increased when vessels were present during haul out periods, and increased further when vessels approached and animals re-entered the water. Harbor seals responded more to vessels passing by haul out sites in areas with less overall vessel activity, and the model best predicting their flushing behavior included the number of boats, type of boats, and distance to boats. More flushing occurred to non-motorized vessels (e.g., kayaks), likely because they tended to occur in groups rather than as single vessels, and tended to pass closer (25–184 m) to the haul out sites than motorized vessels (55–591 m) (Cates & Acevedo-Gutiérrez, 2017). Jones et al. (2017) modeled the spatial overlap of vessel traffic and grey and harbor seals in the UK, and found most overlap to occur within 50 km of the coast, and high overlap occurring within 5 of 13 grey seal Special Areas of Conservation and within 6 of 12 harbor seal Special Areas of Conservation. They also estimated received levels of shipping noise and found maximum daily M-weighted cumulative SEL values from 170 to 189 dB, with the upper confidence intervals of those estimates sometimes exceeding TTS values. However, there was no evidence of reduced population size in any of these high overlap areas.

Mikkelsen et al. (2019) used long-term biologgers (DTAGs) on harbor seals and grey seals to opportunistically examine behaviors. The data showed that seals were exposed to vessel noise between 2.2 and 20.5 percent of their time in water. Potential responses to vessels included interruption of resting and foraging behaviors.

### **Sea Otters**

Sea otters have similar in-air hearing sensitivities as pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), and may react in a similar fashion when approached by vessels. Sea otters depend on visual acuity to forage, so while their eyes are able to focus both in air and underwater (Riedman & Estes, 1990), their underwater hearing sensitivities are significantly reduced compared to pinnipeds (Ghoul & Reichmuth, 2014a, 2014b). While reactions to underwater vessel noise may occur, they will have lower overall severity to those of pinnipeds. Sea otters in Monterey, CA that were living in areas of disturbance from human activity such as recreational boating spent more time engaged in travel than resting (Curland, 1997). Sea otters in undisturbed areas spent 5 percent of their time travelling; otters in areas of disturbance due to vessels were shown to spend 13 percent of their time travelling (Curland, 1997). While this may not appear to be a large change in behavior, sea otter dives are very costly and require twice the metabolic energy that phocid seals need to dive; therefore sea otters may not dive or travel far in response to disturbance, as they already require long periods of rest at the surface to counterbalance the high cost of foraging at sea (Yeates et al., 2007). For example, when a single air gun vessel passed a large raft of otters, several otters were mildly alarmed (e.g., rolled over on their sides or bellies and looked intently at the vessel as it approached) but did not leave the raft. However, they reacted to the vessel every time it passed, even though the air gun was only operational for two of the four passes. This indicates that otters were either responding to the loud airborne sounds of the boat engines and compressor, or to the close approach of the vessel itself, rather than the seismic sounds (Reidman, 1983). However, sea otters may habituate quickly. Even when purposefully harassed in an effort to cause a behavioral response, sea otters generally moved only a short distance (100 to 200 m) before resuming normal activity, and nearby boats, nets, and floating oil containment booms were sometimes an attractant (Davis et al., 1988). Although Barrett (2019) found that sea otters have high metabolic rate and are at risk of increased energetic costs when disturbed, there was less than a 10 percent chance of disturbance when small vessels were more 54 m away from sea otters.

### **Behavioral Reactions to Aircraft Noise**

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 rotary-wing aircraft (i.e., helicopters), as well as unmanned aerial systems. Thorough reviews of the subject and available information is presented in Richardson et al. (1995b) and elsewhere (e.g., Efroymsen et al., 2001; Holst et al., 2011; Luksenburg & Parsons, 2009; 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 (Holst et al., 2011; Mancini et al., 1988). Richardson et al. (1995b) noted that marine mammal reactions to aircraft overflight largely consisted of opportunistic and anecdotal observations lacking clear distinction between reactions potentially caused by the noise of the aircraft and the visual cue an aircraft presents. In addition, it was suggested that variations in the responses noted were due to generally other undocumented factors associated with overflights (Richardson et al., 1995b). These factors could include aircraft type (single engine, multi-engine, jet turbine), flight path (altitude, centered on the animal, off to one side, circling, level and slow), environmental factors (e.g., wind speed, sea state, cloud cover), and locations where native subsistence hunting continues and animals are more sensitive to anthropogenic impacts, including the noise from aircraft. Erbe et al. (2018) measured airplane noise levels underwater at sites about 1 and 10 km from an airport runway and found median noise levels up to 117 dB re 1  $\mu$ Pa and 10 kHz at the close site, and up to 91 dB re 1  $\mu$ Pa and 2 kHz at the more distant site; both would be audible to a number of marine



mammals at those levels and frequencies. Christiansen et al. (2016b) measured the in-air and underwater noise levels of two unmanned aerial vehicles, and found that in air, the broadband source levels were around 80 dB re 20  $\mu$ Pa, while at a meter underwater received levels were 95–100 dB re 1  $\mu$ Pa when the vehicle was only 5–10 m above the surface, and were not quantifiable above ambient noise levels when the vehicle was higher. Therefore, if an animal is near the surface and the unmanned aerial vehicle is low, it may be detected, but in most cases these vehicles are operated at much higher altitudes (e.g., over 30 m) and so are not likely to be heard.

The impact of aircraft overflights is one of the least well-known sources of potential behavioral response by any species or taxonomic group, and so many generalities must be made based on the little data available. There are some data for each taxonomic group; taken together it appears that in general, marine mammals have varying levels of sensitivity to overflights depending on the species and context.

### **Mysticetes**

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Koski et al., 1998). Richardson (1985; 1995b) found no evidence that single or occasional aircraft flying above mysticetes causes long-term displacement of these mammals.

Bowhead whales in the Beaufort Sea exhibited a transient behavioral response to fixed-wing aircraft and vessels. Reactions were frequently observed at less than 1,000 ft. above sea level, infrequently observed at 1,500 ft., and not observed at all at 2,000 ft. (Richardson et al., 1985). Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 150 m or higher. The bowheads exhibited fewer behavioral changes than did the odontocetes in the same area (Patenaude et al., 2002). It should be noted that bowhead whales in this study may have more acute responses to anthropogenic activity than many other marine mammals since these animals were presented with restricted egress due to limited open water between ice floes. Additionally, these animals are hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

A pilot study was conducted on the use of unmanned aerial systems to observe bowhead whales; flying at altitudes between 120 and 210 m above the surface, no behavioral responses were observed in any animals (Koski et al., 2015; Koski et al., 1998). Similarly, Christiansen et al. (2016a) did not observe any responses to an unmanned aerial vehicle flown 30–120 m above the water when taking photos of humpback whales to conduct photogrammetry and assess fitness. In a follow-on study, Christiansen et al. (2020) also did not observe any behavioral response in the form of changes in swim speeds, respiration rates, turning angles, or interbreath intervals to an unmanned aerial vehicle flown over 10 southern right whale mother-calf pairs. In addition, some of the animals were equipped with DTAGs to measure the sound of the unmanned aerial vehicle; the received levels in the 100–1,500 Hz band were  $86 \pm 4$  dB re 1  $\mu$ Pa, very similar to ambient noise levels measured at  $81 \pm 7$  dB in the same frequency band. Acevedo-Whitehouse et al. (2010) successfully maneuvered a remote controlled helicopter over large baleen whales to collect samples of their blows, with no more avoidance behavior than noted for typical photo-identification vessel approaches. These vehicles are much smaller and quieter than typical aircraft and so are less likely to cause a behavioral response, although they may fly at much lower altitudes (Smith et al., 2016).

### **Odontocetes**

Variable responses to aircraft have been observed in toothed whales, though overall little change in behavior has been observed during flyovers. Some toothed whales dove, slapped the water with their flukes or flippers, or swam away from the direction of the aircraft during overflights; others did not visibly react (Richardson et al., 1995b). Würsig et al. (1998) found that beaked whales were the most sensitive cetacean and reacted by avoiding marine mammal survey aircraft in 89 percent of sightings and at more than twice the rate as Kogia whales, which was the next most reactive of the odontocetes in 39 percent of sightings; these are the same species that were sensitive to vessel traffic.

During standard marine mammal surveys at an altitude of 750 ft., some sperm whales remained on or near the surface the entire time the aircraft was in the vicinity, while others dove immediately or a few minutes after being sighted. Other authors have corroborated the variability in sperm whales' reactions to fixed-wing aircraft or helicopters (Green et al., 1992; Richter et al., 2006; Richter et al., 2003; Smultea et al., 2008; Würsig et al., 1998). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al., 1995b). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 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 (Würsig et al., 1998). The same species that show strong avoidance behavior to vessel traffic (Kogia species and beaked whales) show similar reactions to aircraft (Würsig 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 (State of Hawaii, 2015). No changes in group cohesion or orientation behavior were observed for groups of Risso's dolphins, common dolphins, or killer whales when a survey airplane flew at altitudes of 213–610 m, but this may be due to the plane maintaining lateral distances greater than 500 m in all (Smultea & Lomac-MacNair, 2016).

Much like mysticetes, odontocetes have demonstrated no responses to unmanned aerial systems. For example, Durban et al. (2015) conducted photogrammetry studies of killer whales using a small helicopter flown 35–40 m above the animals with no disturbance noted. However, it is possible that odontocete responses could increase with use at reduced altitudes, due either to noise or the shadows created by the vehicle (Smith et al., 2016). Bottlenose dolphins responded to a small portion of unmanned aerial vehicles by briefly orienting when the vehicle was relatively close (10–30 m high), but in most cases did not respond at all (Ramos et al., 2018).

### **Pinnipeds**

Richardson et al. (1995b) noted that responsiveness to aircraft overflights generally was dependent on the altitude of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). In general pinnipeds are unresponsive to overflights, and may startle, orient towards the sound source or increase vigilance, or may briefly re-enter the water, but typically remain

hauled out or immediately return to their haul out location (Blackwell et al., 2004; Gjertz & Børset, 1992). Adult females, calves and juveniles are more likely to enter the water than males, and stampedes resulting in mortality to pups (by separation or crushing) can occur when disturbance is severe, although 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 (Bester et al., 2002; Gjertz & Børset, 1992), although they have been known to elicit behavioral reactions such as fleeing (Hoover, 1988). For California sea lions and Steller sea lions at a rocky haulout off Crescent City in northern California, helicopter approaches to landing sites typically caused the most severe response of diving into the water (National Oceanic and Atmospheric Administration, 2010). Responses were also dependent on the species, with Steller sea lions being more sensitive and California sea lions more tolerant. Depending on the time between subsequent approaches, animals hauled out in between and fewer animals reacted upon subsequent exposures (National Oceanic and Atmospheric Administration, 2010).

Pinniped reactions to rocket launches and overflight at San Nicolas Island were studied from August 2001 to October 2008 (Holst et al., 2011). California sea lions startled and increased vigilance for up to two minutes after a rocket overflight, with some individuals moving down the beach or returning to the water. Northern elephant seals showed little reaction to any overflight. Harbor seals had the most pronounced reactions of the three species observed with most animals within approximately 4 km of the rocket trajectory leaving their haulout sites for the water and not returning for several hours. The authors concluded that the effects of the rocket launches were minor with no effects on local populations evidenced by the growing populations of pinnipeds on San Nicolas Island (Holst et al., 2011).

Pinnipeds may be more sensitive to unmanned aerial systems, especially those flying at low altitudes, due to their possible resemblance to predatorial birds (Smith et al., 2016), which could lead to flushing behavior (Olson, 2013). Responses may also vary by species, age class, behavior, and habituation to other anthropogenic noise, as well as by the type, size, and configuration of unmanned aerial vehicle used (Pomeroy et al., 2015). However, in general pinnipeds have demonstrated little to no response to unmanned aerial systems, with some orienting towards the vehicle, other alerting behavior, or short-term flushing possible (Moreland et al., 2015; Sweeney et al., 2015).

### **Sea Otters**

Sea otters spend approximately 80 percent of their time on the surface of the water (Curland, 1997) with their heads above the surface. Recordings of underwater noise produced by helicopter overflights did not appear to affect sea otter foraging behavior, foraging success, or daily activity patterns when projected underwater 1–1.5 km from a group of otters in Lobos Cove (Reidman, 1983). Sea otters have similar in-air hearing sensitivities as pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), and may react in a similar fashion when exposed to aircraft noise. Pinnipeds in general are unresponsive but may react depending on the altitude of the aircraft or the abruptness of the associated sound (Richardson et al., 1995b), with reactions ranging from unresponsiveness to flushing into the water location (Blackwell et al., 2004; Gjertz & Børset, 1992). Sea otters may dive below the surface of the water or flush into the water to avoid aircraft noise. However, sea otter dives are very costly and require twice the metabolic energy that phocid seals need to dive; therefore sea otters may not dive or travel so readily in response

to disturbance, as they already require long periods of rest at the surface to counterbalance the high cost of foraging at sea (Yeates et al., 2007). So far, there has been no evidence that any aircraft has had adverse effects on a well-monitored translocated colony of sea otters at San Nicolas Island, which has a landing field operated by the U.S. Navy (U.S. Fish and Wildlife Service, 2012, 2015).

### **Behavioral Reactions to Impulsive Noise**

Impulsive signals (i.e., weapon noise and explosions), 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. However, 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. Behavioral reactions from explosive sounds are likely to be similar to reactions studied for other impulsive sounds, such as those produced by air guns and impact pile driving. Data on behavioral responses to impulsive sound sources are limited across all marine mammal groups, with only a few studies available for mysticetes, odontocetes, pinnipeds, and sea otters. Most data have come from seismic surveys that occur over long durations (e.g., on the order of days to weeks) and typically utilize large multi-air gun arrays that fire repeatedly. While seismic data provide the best available science for assessing behavioral responses to impulsive sounds by marine mammals, it is likely that these responses represent a worst-case scenario as compared to responses to explosives used in Navy activities, which would typically consist of single impulses or a cluster of impulses, rather than long-duration, repeated impulses.

### **Mysticetes**

Baleen whales have shown a variety of responses to impulsive sound sources, including avoidance, attraction to the source, reduced surface intervals, altered swimming behavior, and changes in vocalization rates (Gordon et al., 2003; McCauley et al., 2000; Richardson et al., 1985; Southall et al., 2007). Studies have been conducted on many baleen whale species, including gray, humpback, blue, fin and bowhead whales; it is assumed that these responses are representative of all baleen whale species. The behavioral state of the whale seems to be an integral part of whether or not the animal responds and how they respond, as does the location and movement of the sound source, more than the received level of the sound.

Migratory behavior seems to lead to a higher likelihood of response, with some species demonstrating more sensitivity than others do. For example, migrating gray whales showed avoidance responses to seismic vessels at received levels between 164 and 190 dB re 1  $\mu$ Pa (Malme et al., 1986, 1988). Similarly, migrating humpback whales showed avoidance behavior at ranges of 5–8 km from a seismic array during observational studies and controlled exposure experiments in one Australian study (McCauley et al., 1998), and in another Australian study decreased their dive times and reduced their swimming speeds (Dunlop et al., 2015). However, when comparing received levels and behavioral responses using ramp-up versus a constant noise level of air guns, humpback whales did not change their dive behavior but did deviate from their predicted heading and decreased their swim speeds (Dunlop et al., 2016). In addition, the whales demonstrated more course deviation during the constant source trials but reduced travel speeds more in the ramp-up trials; in either case there was no dose-response relationship with the received level of the air gun noise, and similar responses were observed in control trials with vessel movement but no air guns so some of the response was likely due to the presence of the vessel and not the received level of the air guns. When looking at the relationships between proximity, received level, and behavioral response, Dunlop et al. (2017) used responses to two different air guns and found responses occurred more towards the smaller, closer source than to the larger source at the same

received level, demonstrating the importance of proximity. Responses were found to be more likely when the source was within 3 km or above 140 dB re 1  $\mu$ Pa, 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  $\mu$ Pa peak-to-peak). Bowhead whales seem to be the most sensitive species, perhaps due to a higher overlap between bowhead whale distribution and seismic surveys in Arctic and sub-Arctic waters, as well as a recent history of being hunted. While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al., 1995b), some whales avoided vessels by more than 20 km at received levels as low as 120 dB re 1  $\mu$ Pa. Additionally, Malme et al. (1988) observed clear changes in diving and breathing patterns in bowheads at ranges up to 73 km from seismic vessels, with received levels as low as 125 dB re 1  $\mu$ Pa. Bowhead whales may also avoid the area around seismic surveys, from 6 to 8 km (Koski and Johnson 1987, as cited in Gordon et al., 2003) out to 20 or 30 km (Richardson et al., 1999). However, work by Robertson (2013) supports the idea that behavioral responses are contextually dependent, and that during seismic operations bowhead whales may be less “available” for counting due to alterations in dive behavior but that they may not have left the area after all.

In contrast, noise from seismic surveys was not found to impact feeding behavior or exhalation rates in western gray whales while resting or diving off the coast of Russia (Gailey et al., 2007; Yazvenko et al., 2007); however, the increase in vessel traffic associated with the surveys and the proximity of the vessels to the whales did affect the orientation of the whales relative to the vessels and shortened their dive-surface intervals (Gailey et al., 2016). Todd et al. (1996) found no clear short-term behavioral responses by foraging humpbacks to explosions associated with construction operations in Newfoundland but did see a trend of increased rates of net entanglement closer to the noise source, possibly indicating a reduction in net detection associated with the noise through masking or TTS. Distributions of fin and minke whales were modeled with a suite of environmental variables along with the occurrence or absence of seismic surveys, and no evidence of a decrease in sighting rates relative to seismic activity was found for either species (Vilela et al., 2016). Their distributions were driven entirely by environmental variables, particularly those linked to prey including warmer sea surface temperatures, higher chlorophyll-a values, and higher photosynthetically available radiation (a measure of primary productivity). Sighting rates based on over 8,000 hours of baleen and toothed whale survey data were compared on regular vessel surveys versus both active and passive periods of seismic surveys (Kavanagh et al., 2019). Models of sighting numbers were developed, and it was determined that baleen whale sightings were reduced by 88 and 87 percent during active and inactive phases of seismic surveys, respectively, compared to regular surveys. These results seemed to occur regardless of geographic location of the survey; however, when only comparing active versus inactive periods of seismic surveys the geographic location did seem to affect the change in sighting rates.

Vocal responses to seismic surveys have been observed in a number of baleen whale species, including a cessation of calling, a shift in frequency, increases in amplitude or call rate, or a combination of these strategies. Blue whale feeding/social calls were found to increase when seismic exploration was underway, with seismic pulses at average received SELs of 131 dB re 1  $\mu$ Pa<sup>2</sup>s (Di Lorio & Clark, 2010), a potentially compensatory response to increased noise level. Responses by fin whales to a 10-day seismic survey in the Mediterranean Sea included possible decreased 20-Hz call production and movement of animals from the area based on lower received levels and changes in bearings (Castellote et al., 2012).

However, similarly distant seismic surveys elicited no apparent vocal response from fin whales in the mid-Atlantic Ocean; instead, Nieukirk et al. (2012) hypothesized that 20-Hz calls may have been masked from the receiver by distant seismic noise. Models of humpback whale song off Angola showed significant seasonal and diel variation, but also showed a decrease in the number of singers with increasing received levels of air gun pulses (Cerchio et al., 2014). Bowhead whale calling rates decreased significantly at sites near seismic surveys (41–45 km) where median received levels were between 116 and 129 dB re 1  $\mu$ Pa, and did not decrease at sites further from the seismic surveys (greater than 104 km) where median received levels were 99–108 dB re 1  $\mu$ Pa (Blackwell et al., 2013). In fact, bowhead whale calling rates increased at the lower received levels, began decreasing at around 127 dB re 1  $\mu$ Pa<sup>2</sup>s cumulative SEL, and ceased altogether at received levels over 170 dB re 1  $\mu$ Pa<sup>2</sup>s cumulative SEL (Blackwell et al., 2015). Similar patterns were observed for bowhead vocalizations in the presence of tonal sounds associated with drilling activities, and were amplified in the presence of both the tonal sounds and air gun pulses (Blackwell et al., 2017).

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. However, responses appear to be behaviorally mediated, with most avoidance responses occurring during migration behavior and little observed response during feeding behavior. These response patterns are likely to hold true for Navy impulsive sources; however, Navy impulsive sources would largely be stationary (e.g., explosives fired at a fixed target), and short term (on the order of hours rather than days or weeks) than were found in these studies and so responses would likely occur in closer proximity or not at all.

### **Odontocetes**

Few data are available on odontocete responses to impulsive sound sources, with only a few studies on responses to seismic surveys, pile driving and construction activity available. However, odontocetes appear to be less sensitive to impulsive sound than mysticetes, with responses occurring at much closer distances. This may be due to the predominance of low-frequency sound associated with these sources that propagates long distances and overlaps with the range of best hearing for mysticetes but is below that range for odontocetes. The exception to this is the harbor porpoise, which has been shown to be highly sensitive to most sound sources, avoiding both stationary (e.g., pile driving) and moving (e.g., seismic survey vessels) impulsive sound sources out to approximately 20 km (e.g., Haelters et al., 2014; Pirodda et al., 2014). However, even this response is short term, with porpoises returning to the area within hours after the cessation of the noise.

Madsen et al. (2006) and Miller et al. (2009) tagged and monitored eight sperm whales in the Gulf of Mexico exposed to seismic air gun surveys. Sound sources were from approximately 2 to 7 NM away from the whales, and received levels were as high as 162 dB SPL re 1  $\mu$ Pa (Madsen et al., 2006). The whales showed no horizontal avoidance, however one whale rested at the water's surface for an extended period of time until air guns ceased firing (Miller et al., 2009). While the remaining whales continued to execute foraging dives throughout exposure, tag data suggested there may have been subtle effects of noise on foraging behavior (Miller et al., 2009). Similarly, Weir (2008) observed that seismic air gun surveys along the Angolan coast did not significantly reduce the encounter rate of sperm whales during the 10-month survey period, nor were avoidance behaviors to air gun impulsive sounds observed. In contrast, Atlantic spotted dolphins did show a significant, short-term avoidance response to air gun impulses within approximately 1 km of the source (Weir, 2008). The dolphins were observed at greater distances from the vessel when the air gun was in use, and when the air gun was not in use they

readily approached the vessel to bow ride. Kavanagh et al. (2019) also found that toothed whales were more adverse to active airguns, as sightings of several species of odontocetes were reduced by 53 and 29 percent during active and inactive phases of seismic surveys, respectively, compared to regular surveys.

Captive bottlenose dolphins sometimes vocalized or were reluctant to return to the test station after exposure to single impulses from a seismic water gun (Finneran et al., 2002). When exposed to multiple impulses from a seismic air gun, some dolphins turned their heads away from the sound source just before the impulse, showing that they could anticipate the timing of the impulses and perhaps reduce the received level (Finneran et al., 2015). During construction (including the blasting of old bastions) of a bridge over a waterway commonly used by the Tampa Bay, FL stock of bottlenose dolphins, the use of the area by females decreased while males displayed high site fidelity and continued using the area, perhaps indicating differential habitat uses between the sexes (Weaver, 2015).

A study was conducted on the response of harbor porpoises to a seismic survey using aerial surveys and C-PODs (an autonomous recording device that counts odontocete clicks); the animals appeared to have left the area of the survey, and decreased their foraging activity within 5–10 km, as evidenced by both a decrease in vocalizations near the survey and an increase in vocalizations at a distance (Pirrotta et al., 2014; Thompson et al., 2013). However, the animals returned within a day after the air gun operation ceased, and the decrease in occurrence over the survey period was small relative to the observed natural seasonal decrease compared to the previous year. A number of studies (Brandt et al., 2011; Dähne et al., 2014; Haelters et al., 2014; Thompson et al., 2010; Tougaard et al., 2005; Tougaard et al., 2009) also found strong avoidance responses by harbor porpoises out to 20 km during pile driving; however, all studies found that the animals returned to the area after the cessation of pile driving. When bubble curtains were deployed around pile driving, the avoidance distance appeared to be reduced to half that distance (12 km), and the response only lasted about five hours rather than a day before the animals returned to the area (Dähne et al., 2017).

However, not all harbor porpoise behavioral response studies ended in habitat displacement. Sarnocińska et al. (2020) also placed C-PODs near oil and gas platforms and control sites 15 km away and found a dose-response effect with the lowest amount of porpoise activity closest to the seismic vessel ( $SEL_{\text{single shot}} = 155 \text{ dB re } 1 \mu\text{Pa}^2\text{s}$ ) and then increasing porpoise activity out to 8–12 km, outside of which levels were similar to baseline. Distance to the seismic vessel was a better model predictor of porpoise activity than sound level. Despite these smaller-scale responses, a large-scale response was not detected, and overall porpoise activity in the seismic area was similar to the control stations; this may indicate that the porpoises were moving around the seismic area to avoid the ship, but not leaving the area entirely (Sarnocińska et al., 2020).

When exposing a captive harbor porpoise to impact pile driving sounds, Kastelein et al. (2013b) found that above 136 dB re 1  $\mu\text{Pa}$  (zero-to-peak) the animal's respiration rates increased, and at higher levels it jumped more frequently. Bergstrom et al. (2014) found that although there was a high likelihood of acoustic disturbance during wind farm construction (including pile driving), the impact was short term. Graham et al. (2017) assessed the occurrence of bottlenose dolphins and harbor porpoises over different area and time scales with and without impact and vibratory pile driving. While there were fewer hours with bottlenose dolphin detections and reduced detection durations within the pile driving area and increased detection durations outside the area, the effects sizes were small, and the reduced harbor porpoise encounter duration was attributed to seasonal changes outside the influence of the pile driving. However, received levels in this area were lower due to propagation effects than in the other

areas described above, which may have led to the lack of or reduced response. In another impulsive pile driving study, Graham et al. (2019) found that the distance at which behavioral responses were probable decreased over the course of the construction project, suggesting habituation to pile-driving noise in the local harbor porpoise population.

Odontocete behavioral responses to impulsive sound sources are likely species- and context-dependent, with most species demonstrating little to no apparent response. Responses might be expected within close proximity to a noise source, under specific behavioral conditions such as females with offspring, or for sensitive species such as harbor porpoises.

### **Pinnipeds**

A review of behavioral reactions by pinnipeds to impulsive noise can be found in Richardson et al. (1995b) and Southall et al. (2007). Blackwell et al. (2004) observed that ringed seals exhibited little or no reaction to pipe-driving noise with mean underwater levels of 157 dB re 1  $\mu$ Pa and in-air levels of 112 dB re 20  $\mu$ Pa, suggesting that the seals had habituated to the noise. In contrast, captive California sea lions avoided sounds from an underwater impulsive source at levels of 165–170 dB re 1  $\mu$ Pa (Finneran et al., 2003b). Harbor and grey seals were also observed to avoid a seismic air gun by rapidly swimming away, and ceased foraging during exposure, but returned to normal behavior afterwards (Thompson et al. 1998, cited in Gordon et al., 2003). In another study, few responses were observed by New Zealand fur seals to a towed air gun array operating at full power; rather, when responses were observed it seemed to be to the physical presence of the vessel and tow apparatus, and these only occurred when the vessel was within 200 m and sometimes as close as 5 m (Lalas & McConnell, 2016). Captive Steller sea lions were exposed to a variety of tonal, sweep, impulsive and broadband sounds to determine what might work as a deterrent from fishing nets. The impulsive sound had a source level of 120 dB re 1  $\mu$ Pa at 1 m, and caused the animals to haul out and refuse to eat fish presented in a net (Akamatsu et al., 1996). Steller sea lions exposed to in-air explosive blasts increased their activity levels and often re-entered the water when hauled out (Demarchi et al., 2012). However, these responses were short-lived and within minutes, the animals had hauled out again, and there were no lasting behavioral impacts in the days following the blasts.

Experimentally, Götz & Janik (2011) tested underwater startle responses to a startling sound (sound with a rapid rise time and a 93 dB sensation level [the level above the animal's hearing threshold at that frequency]) and a nonstartling sound (sound with the same level, but with a slower rise time) in wild-captured gray seals. The animals exposed to the startling treatment avoided a known food source, whereas animals exposed to the nonstartling treatment did not react or habituated during the exposure period. The results of this study highlight the importance of the characteristics of the acoustic signal in an animal's response of habituation.

Pinnipeds may be the least sensitive taxonomic group to most noise sources, although some species may be more sensitive than others, and are likely to only respond to loud impulsive sound sources at close ranges by startling, jumping into the water when hauled out, or even cease foraging, but only for brief periods before returning to their previous behavior (e.g., (Southall et al., 2007)). Pinnipeds may even experience TTS (see Section 3.8.3.1.1.2, Hearing Loss) before exhibiting a behavioral response (Southall et al., 2007).

### **Sea Otters**

There are few available studies on responses of sea otters to impulsive sounds. A playback study of multiple and single air guns had no significant impact on sea otters in California. During the multiple air



gun exposures, otters rested 1 percent more and foraged 1 percent less. They were successful at obtaining prey during 84 percent of their foraging dives when the air gun vessel was 50 NM away, and success rate only decreased by 5 percent when the multiple air gun vessel moved closer (0.5 NM away). Overall, foraging and dive behaviors remained undisturbed, as did the density and distribution of sea otters in the area. This study caveats that the data were collected under rough weather conditions which could have affected the otters' perception of the seismic sounds. In addition, otters kept close to shore in relatively sheltered coves (Reidman, 1983).

During the single air gun experiment, the air gun ship approached a raft of otters (at a minimum of 730 m), and several otters were mildly alarmed (e.g., rolled over on their sides or bellies and looked intently at the vessel as it approached) but did not leave the raft. Of the four times the vessel passed the group of otters, the air gun was operational during only two of the transects. However, the otters reacted to the vessel every time it passed, indicating that otters were either responding to the loud airborne sounds of the boat engines and compressor, or to the close approach of the vessel itself, rather than the seismic sounds (Reidman, 1983).

In a follow-up study, Riedman (1984) monitored sea otter reactions to drilling platform sounds and air gun firing projected from a source vessel 0.9 to 1.6 km away from groups of sea otters. No behavioral reactions or movements were observed in 14 days of observations with 15-38 individual sea otters present on any given day. Sound pressure levels from the air gun were reported as 166 dB re 1  $\mu$ Pa at 1.1 km, which means that two otters may have been subjected to levels greater than this at ranges of 900 m on the one day the pair foraged closer to the air gun ship for one hour. Most of the otters would have been subjected to just under this level, since the majority of otters foraged 1.3–1.6 m away from the sound sources, and propagation loss due to distance and the kelp environment needs to be considered. In a survey of the local coastline, no change in numbers of sea otters was evident between just prior to the sound stimuli and on day ten of the emissions. No changes in feeding dive times or feeding success was seen during the study either.

When conducting impact and vibratory pile driving for the Parsons Slough estuarine restoration, the Elkhorn Slough National Estuarine Research Reserve (2011) recorded the abundance and behavior of sea otters in the area. Disturbances within 30 m of the pile driving site included otters raising their heads, swimming away without startling, or startle diving. Usually only single adult males with an established territory that included the construction site traveled within 30 m. Otters further away (> 180 m) were observed swimming away with startling, including mother-pup pairs. However, sea otter behavioral disturbances 30-180 m away from the pile driving site were difficult to tease apart from the impacts of pedestrian vessels and other construction activities.

Sea otters spend approximately 80 percent of their time on the surface of the water (Curland, 1997) with their heads above the surface, which reduces their exposure to underwater sounds. They require long periods of undisturbed rest at the surface to counterbalance high metabolic costs associated with foraging at sea (Yeates et al., 2007). If reactions to Navy impulsive noise were to occur, they may be similar to those of pinnipeds, which show temporary avoidance responses or cessation of foraging behavior (Thompson et al., 1998, cited in Gordon et al., 2003). However, underwater hearing sensitivities are significantly reduced in sea otters when compared to pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), so reactions may not be as strong, if they occur at all.

#### 3.8.3.1.1.6 Stranding

When a marine mammal (alive or dead) swims or floats onto shore and becomes beached or incapable of returning to sea, the event is termed a “stranding” (Geraci et al., 1999; Geraci & Lounsbury, 2005; Perrin & Geraci, 2002). A stranding can also occur away from the shore if the animal is unable to cope in its present situation (e.g., disabled by a vessel strike, out of habitat) (Geraci & Lounsbury, 2005).

Specifically, under U.S. law, a stranding is an event in the wild in which: “ (A) a marine mammal is dead and is (i) on a beach or shore of the United States; or (ii) in waters under the jurisdiction of the United States (including any navigable waters); or (B) a marine mammal is alive and is (i) on a beach or shore of the United States and is unable to return to the water; (ii) on a beach or shore of the United States and, although able to return to the water, is in need of medical attention; or (iii) in the waters under the jurisdiction of the United States (including any navigable waters), but is unable to return to its natural habitat under its own power or without assistance” (16 United States Code [U.S.C.] section 1421h).

Marine mammals are subjected to a variety of natural and anthropogenic factors, acting alone or in combination, which may cause a marine mammal to strand (Geraci et al., 1999; Geraci & Lounsbury, 2005). Natural factors related to strandings include limited food availability or following prey inshore, predation, disease, parasitism, natural toxins, echolocation disturbance, climatic influences, solar activity-based disruption of magnetoreception, and aging (Bradshaw et al., 2006; Culik, 2004; Geraci et al., 1999; Geraci & Lounsbury, 2005; Granger et al., 2020; Huggins et al., 2015; National Research Council, 2006; Perrin & Geraci, 2002; Walker et al., 2005). Anthropogenic factors include pollution (Hall et al., 2006; Jepson et al., 2005), vessel strike (Geraci & Lounsbury, 2005; Laist et al., 2001), fisheries interactions (Read et al., 2006), entanglement (Baird & Gorgone, 2005; Saez et al., 2013; Saez et al., 2012), human activities (e.g., feeding, gunshot) (Dierauf & Gulland, 2001; Geraci & Lounsbury, 2005), and noise (Cox et al., 2006; National Research Council, 2003; Richardson et al., 1995b). For some stranding events, environmental factors (e.g., ocean temperature and wind speed and geographic conditions) can be utilized in predictive models to aid in understanding why marine mammals strand in certain areas more than others (Berini et al., 2015). Decomposition, buoyancy, scavenging by other marine species, wave damage, and other oceanic conditions complicate the assessment of marine mammal carcasses (Moore et al., 2020). In most instances, even for the more thoroughly investigated strandings involving post-stranding data collection and necropsies, the cause (or causes) for strandings remains undetermined.

Along the coasts of the continental United States and Alaska between 2001 and 2009, there were on average approximately 12,545 cetacean strandings and 39,104 pinniped strandings (51,649 total) per year (National Marine Fisheries Service, 2016b). Several mass strandings (strandings that involve two or more individuals of the same species, excluding a single mother-calf pair) that have occurred over the past two decades have been associated with anthropogenic activities that introduced sound into the marine environment such as naval operations and seismic surveys. U.S. Navy sonar has been identified as a contributing factor in a small number of strandings; none of these have occurred in the Study Area. U.S. Navy sonar has been identified as a contributing factor in a small number of strandings; none of these have occurred in the Study Area.

Sonar use during exercises involving the U.S. Navy has been identified as a contributing cause or factor in five specific mass stranding events: Greece in 1996; the Bahamas in March 2000; Madeira Island, Portugal in 2000; the Canary Islands in 2002, and Spain in 2006 (Cox et al., 2006; Fernandez, 2006; U.S. Department of the Navy, 2017c), as described in the Navy’s technical report titled *Marine Mammal Strandings Associated with U.S. Navy Sonar Activities* (U.S. Department of the Navy, 2017c). These five

mass strandings have resulted in about 40 known cetacean deaths consisting mostly of beaked whales and with close linkages to mid-frequency active sonar activity. In these circumstances, exposure to non-impulsive acoustic energy was considered a potential indirect cause of death of the marine mammals (Cox et al., 2006). Factors that were associated with these beaked whales strandings included steep bathymetry, multiple hull-mounted platforms using sonar simultaneously, constricted channels, and strong surface ducts. An in-depth discussion of these strandings and these factors is in the technical report titled *Marine Mammal Strandings Associated with U.S. Navy Sonar Activities* (available at [www.goaeis.com](http://www.goaeis.com)). Strandings of other marine mammal species have not been as closely linked to sonar exposure, but rather, have typically been attributed to natural or other anthropogenic factors. The Navy has reviewed training requirements, standard operating procedures, and potential mitigation measures, and has implemented changes to reduce the potential for acoustic related strandings to occur in the future. Discussions of procedures associated with these and other training events are presented in Chapter 5 (Mitigation).

Simonis et al. (2020a) relied on substantially incomplete or inaccurate assumptions about U.S. Navy sonar use around the Mariana Islands (i.e., publicly available press releases and news reports about named Navy activities, which may or may not have involved sonar, rather than actual records of sonar use) to claim a correlation between sonar and beaked whale strandings in the Mariana Islands (outside of the MITT Study Area). Simonis et al. (2020a) found that there was a 1 percent probability of the strandings and sonar co-occurring randomly. In response to the preliminary analysis of Simonis et al. (2020a), the Navy provided additional information to the researchers indicating that the assumptions about sonar use in their analysis were incorrect or incomplete; therefore, their published findings were not valid. In discussions with NMFS following Simonis et al.'s findings, including NMFS researchers who participated in Simonis et al.'s study, the Navy agreed to examine the classified sonar record around the Mariana Islands for correlation with beaked whale strandings. The Center for Naval Analysis conducted a statistical study of correlation of beaked whale strandings around the Mariana Islands with the use of U.S. Navy sonar, finding that no statistically significant correlation exists (Center for Naval Analysis, 2020). The Center for Naval Analysis study used the complete classified record of all U.S. Navy sonar used between 2007 and 2019, including major training events, joint exercises, and unit-level training/testing. Sonar sources in this record conservatively included both hull-mounted and non-hull-mounted sources, rather than solely hull-mounted sources (which have been previously associated with a limited number of beaked whale strandings outside of this study area). The analysis also included the complete beaked whale stranding record for the Mariana Islands through 2019. Following the methods in Simonis et al. (2020), the Center for Naval Analysis conducted a Poisson distribution analysis and found no statistically significant correlation between sonar use and beaked whale strandings when considering the complete sonar use record. The unclassified summary of the Center for Naval Analysis's study was provided to NMFS and their scientists. The Navy is supporting continued efforts to gain a better understanding of beaked whale occurrence and potential effects from Navy activities in the Mariana Islands.

Multiple hypotheses regarding the relationship between non-impulsive sound exposure and stranding have been proposed (see Bernaldo de Quirós et al., 2019). These range from direct impact of the sound on the physiology of the marine mammal, to behavioral reactions contributing to altered physiology (e.g., "gas and fat embolic syndrome") (Fernandez et al., 2005; Jepson et al., 2003; Jepson et al., 2005), to behaviors directly contributing to the stranding (e.g., beaching of fleeing animals). Unfortunately, without direct observation of not only the event but also the underlying process, and given the potential for artefactual evidence (e.g., chronic condition, previous injury) to complicate conclusions from the

post-mortem analyses of stranded animals (Cox et al., 2006), it has not been possible to determine with certainty the exact mechanism underlying these strandings. Based on examination of the above sonar-associated strandings, Bernaldo de Quirós et al. (2019) list diagnostic features, the presence of all of which suggest gas and fat embolic syndrome for beaked whales stranded in association with sonar exposure. Bernaldo de Quirós et al. (2019) observed that, to date, strandings which have a confirmed association with naval exercise have exhibited all seven of the following diagnostic features:

1. Individual or multiple animals stranded within hours or a few days of an exercise in good body condition
2. Food remnants in the first gastric compartment ranging from undigested food to squid beaks
3. Abundant gas bubbles widely distributed in veins (subcutaneous, mesenteric, portal, coronary, subarachnoid veins, etc.) composed primarily of nitrogen in fresh carcasses
4. Gross subarachnoid and/or acoustic fat hemorrhages
5. Microscopic multi-organ gas and fat emboli associated with bronchopulmonary shock
6. Diffuse, mild to moderate, acute, monophasic myonecrosis (hyaline degeneration) with "disintegration" of the interstitial connective tissue and related structures, including fat deposits, and their replacement by amorphous hyaline material (degraded material) in fresh and well-preserved carcasses
7. Multi-organ microscopic hemorrhages of varying severity in lipid-rich tissues such as the central nervous system, spinal cord, and the coronary and kidney fat when present

Historically, stranding reporting and response efforts have been inconsistent, although they have improved considerably over the last 25 years. Although reporting forms have been standardized nationally, data collection methods, assessment methods, detail of reporting and procedures vary by region and are not yet standardized across the United States. Conditions such as weather, time, location, and decomposition state may also affect the ability to thoroughly examine a specimen (Carretta et al., 2016b; Moore et al., 2013). Because of this, the current ability to interpret long-term trends in marine mammal stranding is limited. While the investigation of stranded animals provides insight into the types of threats marine mammal populations face, investigations are only conducted on a small fraction of the total number of strandings that occur, limiting the understanding of the causes of strandings (Carretta et al., 2016a). Although many marine mammals likely strand due to natural or anthropogenic causes, the majority of reported type of occurrences in marine mammal strandings in the Pacific include fisheries interactions, entanglement, vessel strike, and predation (Carretta et al., 2019a; Carretta et al., 2019b; Carretta et al., 2017a; Helker et al., 2019; Helker et al., 2017; National Oceanic and Atmospheric Administration, 2018e, 2019a).

Stranded marine mammals are reported along the entire western coast of the United States each year. Marine mammals strand due to natural or anthropogenic causes, the majority of reported type of occurrences in marine mammal strandings in this region include fishery interactions, illness, predation, and vessel strikes (Carretta et al., 2017a; Helker et al., 2017; National Marine Fisheries Service, 2016g). It is important to note that the mass stranding of pinnipeds along the west coast considered part of a NMFS declared Unusual Mortality Event are still being evaluated. The likely cause of this event is the lack of available prey near rookeries due to warming ocean temperatures (National Oceanic and Atmospheric Administration, 2018a). Carretta et al. (2016b; 2013) provide additional information and data on the threats from human-related activities and the potential causes of strandings for the U.S. Pacific coast marine mammal stocks.

#### 3.8.3.1.1.7 Long-Term Consequences

Long-term consequences to a population are determined by examining changes in the population growth rate (see Section 3.0.4.3, Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). Physical effects that could lead to a reduction in the population growth rate include mortality or injury, which could remove animals from the reproductive pool, and permanent hearing impairment or chronic masking, which could impact navigation, foraging, predator avoidance, or communication. The long-term consequences due to individual behavioral reactions and short-term or chronic instances of physiological stress are especially difficult to predict because individual experience over time can create complex contingencies, especially for long-lived animals like marine mammals. For example, a lost reproductive opportunity could be a measurable cost to the individual, or for very small populations to the population as a whole; however, short-term costs may be recouped during the life of an otherwise healthy individual. These factors are taken into consideration when assessing risk of long-term consequences. It is more likely that any long-term consequences to an individual would be a result of costs accumulated over a season, year, or life stage due to multiple behavioral or stress responses resulting from exposure to many sound-producing activities over significant periods.

Marine mammals exposed to high levels of human activities may leave the area, habituate to the activity, or tolerate the disturbance and remain in the area (Wartzok et al., 2003). Highly resident or localized populations may also stay in an area of disturbance because the cost of displacement may be higher than the cost of remaining (Forney et al., 2017). Longer term displacement can lead to changes in abundance or distribution patterns of the species in the affected region (Bejder et al., 2006b; Blackwell et al., 2004; Teilmann et al., 2006). Gray whales in Baja California abandoned a historical breeding lagoon in the mid-1960s due to an increase in dredging and commercial shipping operations. However, whales did repopulate the lagoon after shipping activities had ceased for several years (Bryant et al., 1984). Mysticetes in the northeast tended to adjust to vessel traffic over a number of years, trending towards more neutral responses to passing vessels (Watkins, 1986), indicating that some animals may habituate or otherwise learn to cope with high levels of human activity. Bejder et al. (2006a) studied responses of bottlenose dolphins to vessel approaches and found that lesser reactions in populations of dolphins regularly subjected to high levels of vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity. Related population characteristics, such as if a population is open or closed, can influence the sensitivity of population disturbance as well (New et al., 2020). New et al. (2020) found that closed populations could not withstand a higher probability of disturbance, compared to open populations with no limitation on food.

Moore and Barlow (2013) noted a decline in the overall beaked whale population in a broad area of the Pacific Ocean along the U.S. West Coast. Moore and Barlow (2013) provide several hypotheses for the decline of beaked whales in those waters, one of which is anthropogenic sound including the use of sonar by the U.S. Navy; however, new data have been published raising uncertainties over whether a decline in the beaked whale population occurred off the U.S. West Coast between 1996 and 2014 (Barlow, 2016). Moore and Barlow (2017) have since incorporated information from the entire 1991 to 2014 time series, which suggests an increasing abundance trend and a reversal of the declining trend along the U.S. West Coast that had been noted in their previous (2013) analysis.

In addition, studies on the Atlantic Undersea Test and Evaluation Center instrumented range in the Bahamas have shown that some Blainville's beaked whales may be resident during all or part of the year in the area. Individuals may move off the range for several days during and following a sonar event, but

return within a few days (Joyce et al., 2019; McCarthy et al., 2011; Tyack et al., 2011).

Photo-identification studies in the SOCAL Range Complex have identified approximately 100 individual Cuvier's beaked whale individuals, with 40 percent having been seen in one or more prior years and re-sightings up to seven years apart (Falcone & Schorr, 2014; Falcone et al., 2009). These results indicate long-term residency by individuals in an intensively used Navy training area, which may suggest a lack of long-term consequences as a result of exposure to Navy training activities, but could also be indicative of high-value resources that exceed the cost of remaining in the area. Long-term residency does not mean there has been no impact on population growth rates and there are no data existing on the reproductive rates of populations inhabiting the Navy range area around San Clemente Island as opposed to beaked whales from other areas. In that regard however, recent results from photo-identifications are beginning to provide critically needed calving and weaning rate data for resident animals on the Navy's Southern California range. Three adult females that had been sighted with calves in previous years were again sighted in 2016, one of these was associated with her second calf, and a fourth female that was first identified in 2015 without a calf, was sighted in 2016 with a calf (Schorr et al., 2017). Resident females documented with and without calves from year to year will provide the data for this population that can be applied to future research questions.

Research involving three tagged Cuvier's beaked whales in the SOCAL Range Complex reported on by Falcone and Schorr (2012, 2014) has documented movements in excess of hundreds of kilometers by some of those animals. Schorr et al. (2014) reported the results for an additional eight tagged Cuvier's beaked whales in the same area. Five of these eight whales made journeys of approximately 250 km from their tag deployment location, and one of these five made an extra-regional excursion over 450 km south to Mexico and back again. Given that some beaked whales may routinely move hundreds of kilometers as part of their normal pattern (Schorr et al., 2014), temporarily leaving an area to avoid sonar or other anthropogenic activity may have little cost.

Another approach to investigating long-term consequences of anthropogenic noise exposure has been an attempt to link short-term effects to individuals from anthropogenic stressors with long-term consequences to populations using population models. Population models are well known from many fields in biology including fisheries and wildlife management. These models accept inputs for the population size and changes in vital rates of the population, such as the mean values for survival age, lifetime reproductive success, and recruitment of new individuals into the population. Unfortunately, for acoustic and explosive impacts on marine mammal populations, many of the inputs required by population models are not known. Nowacek et al. (2016) reviewed new technologies, including passive acoustic monitoring, tagging, and the use of unmanned aerial vehicles that can improve scientists' abilities to study these model inputs and link behavioral changes to individual life functions and ultimately population-level effects. The linkage between immediate behavioral or physiological effects to an individual due to a stressor such as sound, the subsequent effects on that individual's vital rates (growth, survival, and reproduction), and in turn the consequences for the population have been reviewed in National Research Council (2005).

The Population Consequences of Acoustic Disturbance model (National Research Council 2005) proposes a conceptual model for determining how changes in the vital rates of individuals (i.e., a biologically significant consequence to the individual) translates into biologically significant consequences to the population. In 2009, the U.S. Office of Naval Research set up a working group to transform the Population Consequences of Acoustic Disturbance framework into a mathematical model and include other stressors potentially causing disturbance in addition to noise. The model, now called Population

Consequences of Disturbance, has been used for case studies involving bottlenose dolphins, North Atlantic right whales, beaked whales, southern elephant seals, California sea lions, blue whales, humpback whales, and harbor porpoise (Costa et al., 2016a; Costa et al., 2016b; Harwood & King, 2014; Hatch et al., 2012; King et al., 2015; McHuron et al., 2018; New et al., 2014; New et al., 2013a; Pirodda et al., 2018a; Pirodda et al., 2018b). Currently, the Population Consequences of Disturbance model provides a theoretical framework and identifies types of data that would be needed to assess population-level impacts using this process. The process is complicated and provides a foundation for the type of data that are needed, which are currently lacking for many marine mammal species (Booth et al., 2020). Relevant data needed for improving these analytical approaches for population-level consequences resulting from disturbances will continue to be collected during projects funded by the Navy's marine species monitoring program.

Costa et al. (2016a) emphasized taking into account the size of an animal's home range, whether populations are resident and non-migratory or if they migrate over long areas and share their feeding or breeding areas with other populations. These factors, coupled with the extent, location, and duration of a disturbance can lead to markedly different impact results. For example, Costa et al. (2016a) modeled seismic surveys with different radii of impacts on the foraging grounds of Bering Sea humpback whales, West Antarctic Peninsula humpback whales, and California Current blue whales, and used data from tagged whales to determine foraging locations and effort on those grounds. They found that for the blue whales and the West Antarctic humpback whales, less than 19 percent and 16 percent (respectively) of each population would be exposed, and less than 19 percent and 6 percent (respectively) of foraging behavior would be disturbed. This was likely due to the fact that these populations forage for krill over large areas. In contrast, the Bering Sea population of humpback whales had over 90 percent of the population exposed when the disturbance zones extended beyond 50 km, but 100 percent of their foraging time would occur during an exposure when the zone was 25 km or more. These animals forage for fish over a much smaller area, thereby having a limited range for foraging that can be disturbed. Similarly, Costa et al. (2016b) placed disturbance zones in the foraging and transit areas of northern elephant seals and California sea lions. Again, the location and radius of disturbance impacted how many animals were exposed and for how long, with California sea lions disturbed for a longer period than elephant seals, which extend over a broader foraging and transit area. However, even the animals exposed for the longest periods had negligible modeled impacts on their reproduction and pup survival rates. Energetic costs were estimated for western gray whales that migrated to possible wintering grounds near China or to the Baja California wintering grounds of eastern gray whales versus the energetic costs of the shorter migration of eastern gray whales (Villegas-Amtmann et al., 2017). Researchers found that when the time spent on the breeding grounds was held constant for both populations, the energetic requirements for the western gray whales were estimated to be 11 and 15 percent greater during the migration to Baja California and China, respectively, than for the migration of eastern gray whales, and therefore this population would be more sensitive to energy lost through disturbance.

Pirodda et al. (2018b) modeled one reproductive cycle of a female North Pacific blue whale, starting with leaving the breeding grounds off Baja California to begin migrating north to feeding grounds off California, and ending with her returning to the breeding grounds, giving birth, and lactating. They modeled this scenario with no disturbance and found 95 percent calf recruitment; under a "normal" environmental perturbation (El Niño-Southern Oscillation) there was a very small reduction in recruitment, and, under an "unprecedented" environmental change, recruitment was reduced to 69 percent. An intense, localized anthropogenic disturbance was modeled (although the duration of the

event was not provided); if the animals were not allowed to leave the area, they did not forage and recruitment dropped to 63 percent. However, if animals could leave the area of the disturbance then there was almost no change to the recruitment rate. A weak but broader spatial disturbance, where foraging was reduced by 50 percent, caused only a small decrease in calf recruitment to 94 percent. Similarly, Hin et al. (2019) looked at the impacts of disturbance on long-finned pilot whales and found that the timing of the disturbance with seasonally-available resources is important. If a disturbance occurred during periods of low resource availability, the population-level consequences were greater than if the disturbance occurred during periods when resource levels were high.

Using the Population Consequences of Disturbance framework, modeling of the long-term consequences of exposure has been conducted for a variety of marine mammal species and stressors. Even when high and frequent exposure levels are included, few long-term consequences have been predicted. For example, De Silva et al. (2014) conducted a population viability analysis on the long-term impacts of pile driving and construction noise on harbor porpoises and bottlenose dolphins. Despite including the extreme and unlikely assumptions that 25 percent of animals that received PTS would die, and that behavioral displacement from an area would lead to breeding failure, the model only found short-term impacts on the population size and no long-term effects on population viability. Similarly, King et al. (2015) developed a Population Consequences of Disturbance framework using expert elicitation data on impacts from wind farms on harbor porpoises, and even under the worst case scenarios predicted less than a 0.5 percent decline in harbor porpoise populations. Nabe-Nelson et al. (2014) also modeled the impact of noise from wind farms on harbor porpoises and predicted that even when assuming a 10 percent reduction in population size if prey is impacted up to two days, the presence of ships and wind turbines did not deplete the population. In contrast, Heinis and De Jong (2015) used the Population Consequences of Disturbance framework to estimate impacts from both pile driving and seismic exploration on harbor porpoises and found a 23 percent decrease in population size over six years, with an increased risk for further reduction with additional disturbance days. These seemingly contradictory results demonstrate that refinements to models need to be investigated to improve consistency and interpretation of model results.

Recent studies have investigated the potential consequences of fasting for harbor porpoises because their high metabolic rate may leave them especially vulnerable to disturbances that prevent them from feeding. Kastelein et al. (2019c) used an opportunistic experimental approach whereby four stranded wild harbor porpoises were able to consume 85–100 percent of their daily food mass intake in a short time period with no physical problems, suggesting they can compensate for periods of missed feeding if food is available. Similarly, using a modelled approach, Booth (2019) found that harbor porpoises are capable of recovering from lost foraging opportunities, largely because of their varied diet, high foraging rates, and high prey capture success. By modeling their foraging behavior and known prey species and sizes, the porpoises' generalist feeding behavior, in most scenarios, would enable them to obtain more than 100 percent of their energetic needs through typical foraging behavior, and therefore would largely be robust to short-term disturbances to foraging.

The Population Consequences of Disturbance model developed by New et al. (2013b) predicted that beaked whales require energy dense prey and high quality habitat, and that non-lethal disturbances that displace whales from that habitat could lead to long-term impacts on fecundity and survival; however, the authors were forced to use many conservative assumptions within their model since many parameters are unknown for beaked whales. As discussed above in Schorr et al. (2014), beaked whales have been tracked roaming over distances of 250 km or more, indicating that temporary displacement



from a small area may not preclude finding energy dense prey or high quality habitat. 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. In addition, Deros et al. (2020) propose that blubber thickness, which has been used to measure cetacean energy stores and health, is not an appropriate metric because marine mammals may not use their fat stores in a similar manner to terrestrial mammals. These results may be useful in the development of future Population Consequences of Multiple Stressors and Population Consequences of Disturbance models since they should seek to qualify cetacean health in a more ecologically relevant manner.

Another Population Consequences of Disturbance model developed in New et al. (2014) predicted elephant seal populations to be relatively robust even with a greater than 50 percent reduction in foraging trips (only a 0.4 percent population decline in the following year). McHuron et al. (2018) modeled the introduction of a generalized disturbance at different times throughout the breeding cycle of California sea lions, with the behavior response being an increase in the duration of a foraging trip by the female. Very short duration disturbances or responses led to little change, particularly if the disturbance was a single event, and changes in the timing of the event in the year had little effect. However, with even relatively short disturbances or mild responses, when a disturbance was modeled as recurring there were resulting reductions in population size and pup recruitment. Often, the effects weren't noticeable for several years, as the impacts on pup recruitment did not affect the population until those pups were mature.

Population Consequences of Disturbance models can also be used to assess the impacts of multiple stressors. For example, Farmer et al. (2018) modeled the combined impacts of an oil spill and acoustic disturbance due to seismic airgun surveys. They found that the oil spill led to declines in the population over 10 years, and some models that included behavioral response to airguns found further declines. However, the amount of additional population decline due to acoustic disturbance depended on the way the dose-response of the noise levels were modeled, with a single step-function leading to higher impacts than a function with multiple steps and frequency weighting. In addition, the amount of impact from both disturbances was mediated when the metric in the model that described animal resilience was changed to increase resilience to disturbance (e.g., able to make up reserves through increased foraging).

It should be noted that, in all of these models, assumptions were made and many input variables were unknown and so were estimated using available data. It is still not possible to utilize individual short-term behavioral responses to estimate long-term or population-level effects.

The best assessment of long-term consequences from Navy training activities will be to monitor the populations over time within the Study Area. A U.S. workshop on Marine Mammals and Sound (Fitch et al., 2011) indicated a critical need for baseline biological data on marine mammal abundance, distribution, habitat, and behavior over sufficient time and space to evaluate impacts from human-generated activities on long-term population survival. The Navy has developed and implemented comprehensive monitoring plans since 2009 for protected marine mammals occurring on Navy ranges with the goal of assessing the impacts of training activities on marine species and the effectiveness of the Navy's mitigation measures. The results of this long-term monitoring are now being compiled and analyzed for trends in occurrence or abundance over time (e.g., Martin et al., 2017); preliminary results of this analysis at Pacific Missile Range Facility off Kauai, Hawaii indicate no changes in detection rates

for several species over the past decade, demonstrating that Navy activities may not be having long-term population-level impacts. This type of analysis can be expanded to the other Navy ranges, such as in the Pacific Northwest. Continued analysis of this 15-year dataset and additional monitoring efforts over time are necessary to fully understand the long-term consequences of exposure to military readiness activities.

#### **3.8.3.1.2 Impacts from Sonar and Other Transducers**

Sonar and other transducers proposed for use could be used throughout the Study Area. Sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. General categories of these systems are described in Section 3.0.4.1 (Acoustic Sources).

Sonar-induced acoustic resonance and bubble formation phenomena are very unlikely to occur under realistic conditions, as discussed in Section 3.8.3.1.1.1 (Injury). Non-auditory injury (i.e., other than PTS) and mortality from sonar and other transducers is so unlikely as to be discountable under normal conditions and is therefore not considered further in this analysis.

The most probable impacts from exposure to sonar and other transducers are PTS, TTS, behavioral reactions, masking, and physiological stress (Sections 3.8.3.1.1.2, Hearing Loss; 3.8.3.1.1.3, Physiological Stress; 3.8.3.1.1.4, Masking; and 3.8.3.1.1.5, Behavioral Reactions).

##### **3.8.3.1.2.1 Methods for Analyzing Impacts from Sonar and Other Transducers**

The Navy performed a quantitative analysis to estimate the number of times that marine mammals could be affected by sonars and other transducers used during Navy training activities. The Navy's quantitative analysis to determine impacts on marine mammals uses the Navy Acoustic Effects Model to produce initial estimates of the number of times that animals may experience these effects; these estimates are further refined by considering animal avoidance of sound-producing activities and implementation of procedural mitigation measures. The steps of this quantitative analysis are described in Section 3.0.1.2 (Navy's Quantitative Analysis to Determine Impacts to Sea Turtles and Marine Mammals), which takes into account

- criteria and thresholds used to predict impacts from sonar and other transducers (see below);
- the species density (U.S. Department of the Navy, 2020b) and spatial distribution (Watwood et al., 2018) of marine mammals; and
- the influence of environmental parameters (e.g., temperature, depth, salinity) on sound propagation when estimating the received sound level on the animals.

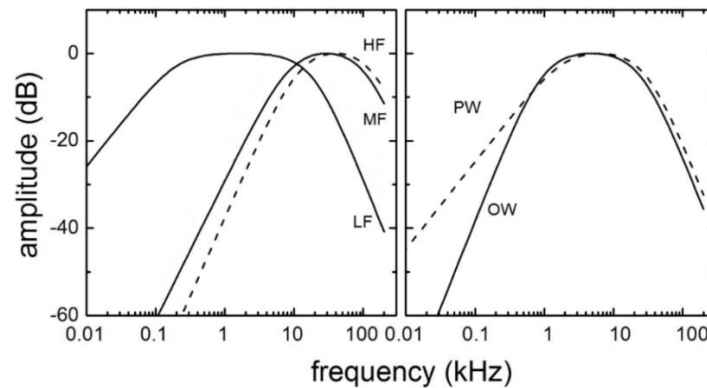
A detailed explanation of this analysis is provided in the technical report titled *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (U.S. Department of the Navy, 2018d).

#### **Criteria and Thresholds Used to Estimate Impacts from Sonar and Other Transducers**

See the technical report titled *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a) for detailed information on how the criteria and thresholds were derived. The marine mammal criteria and thresholds developed for that technical report were relied on by National Marine Fisheries Service in establishing guidance for assessing the effects of sound on marine mammal hearing (National Marine Fisheries Service, 2016h) and were re-affirmed in the 2018 revision (National Marine Fisheries Service, 2018a). In addition, these auditory impact criteria were recently published by Southall et al. (2019c).

### Auditory Weighting Functions

Animals are not equally sensitive to noise at all frequencies. To capture the frequency-dependent nature of the effects of noise, auditory weighting functions are used (Figure 3.8-6). Auditory weighting functions are mathematical functions that adjust received sound levels to emphasize ranges of best hearing and de-emphasize ranges with less or no auditory sensitivity. They are based on a generic band pass filter and incorporates species-specific hearing abilities to calculate a weighted received sound level in units SPL or SEL. Due to the band pass nature of auditory weighting functions, they resemble an inverted “U” shape with amplitude plotted as a function of frequency. The flatter portion of the plotted function, where the amplitude is closest to zero, is the emphasized frequency range (i.e., the pass-band), while the frequencies below and above this range (where amplitude declines) are de-emphasized.



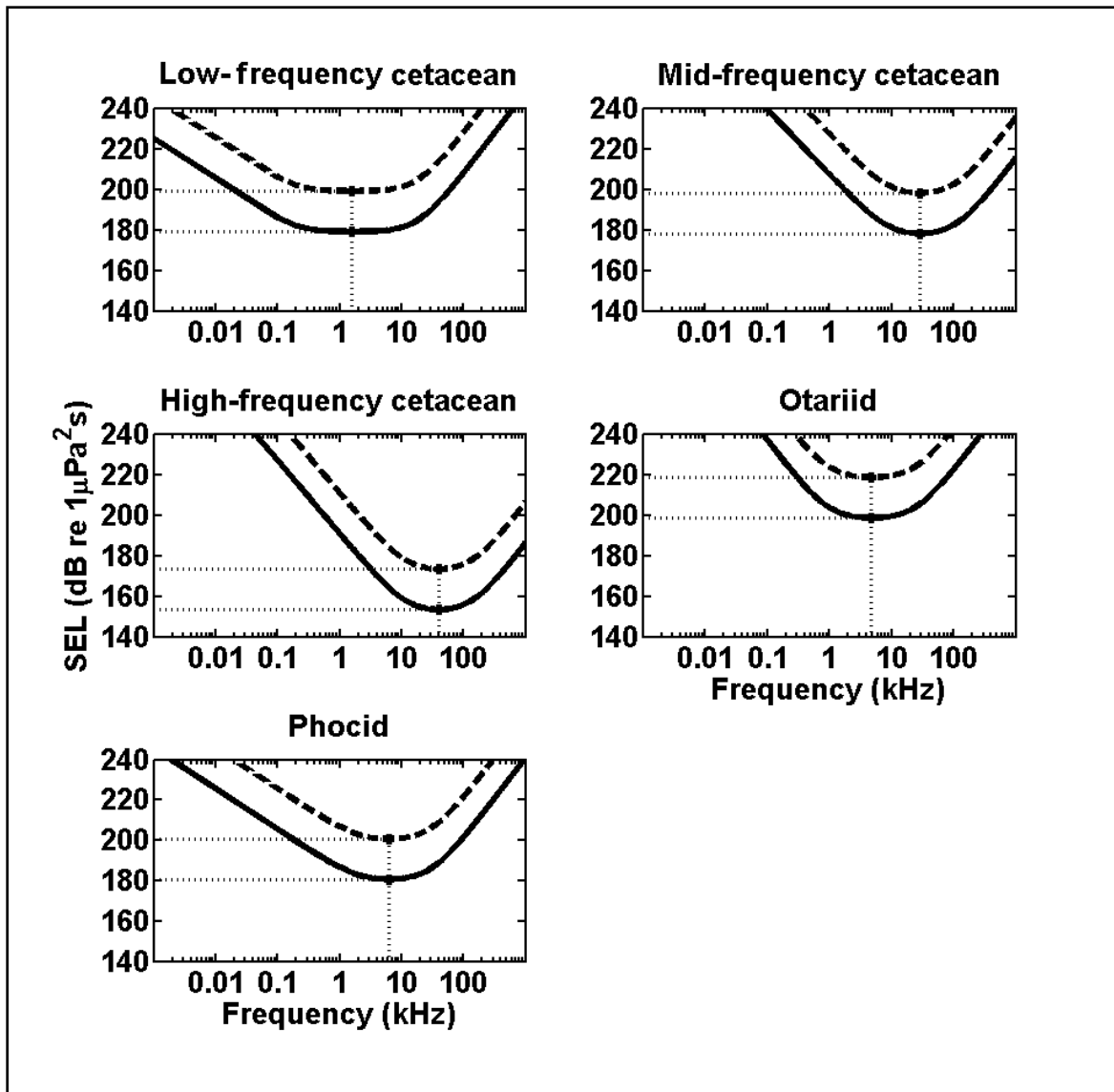
Source: For parameters used to generate the functions and more information on weighting function derivation, see the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* technical report (U.S. Department of the Navy (2017a))

Notes: HF = high-frequency cetacean, LF = low-frequency cetacean, MF = mid-frequency cetacean, PW = phocid (in-water), and OW = otariid (in-water).

**Figure 3.8-6: Navy Auditory Weighting Functions for All Species Groups**

### Hearing Loss from Sonar and Other Transducers

Defining the TTS and PTS exposure functions (Figure 3.8-7) requires identifying the weighted exposures necessary for TTS and PTS onset from sounds produced by sonar and other transducers. The criteria used to define threshold shifts from non-impulsive sources (e.g., sonar) determines TTS onset as the SEL necessary to induce 6 dB of threshold shift. An SEL 20 dB above the onset of TTS is used in all hearing groups of marine mammals underwater to define the PTS threshold (Southall et al., 2007).



Notes: The solid curve is the exposure function for TTS onset and the large dashed curve is the exposure function for PTS onset. Small dashed lines and asterisks indicate the SEL threshold for TTS and PTS onset in the frequency range of best hearing.

**Figure 3.8-7: TTS and PTS Exposure Functions for Sonar and Other Transducers**

#### Behavioral Responses from Sonar and Other Transducers

Behavioral response criteria are used to estimate the number of animals that may exhibit a behavioral response to sonar and other transducers. See the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* technical report for detailed information on how the Behavioral Response Functions were derived (U.S. Department of the Navy, 2017a). Developing the new behavioral criteria involved multiple steps. All peer-reviewed published behavioral response studies conducted both in the field and on captive animals were examined in order to understand the breadth of behavioral responses of marine mammals to sonar and other transducers.

The data from the behavioral studies were analyzed by looking for significant responses, or lack thereof, for each experimental session. The terms “significant response” or “significant behavioral response” are used in describing behavioral observations from field or captive animal research that may rise to the level of “harassment” for military readiness activities. Under the MMPA, for military readiness activities, such as Navy training, behavioral “harassment” is “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*” (16 U.S.C. section 1362(3)(18)(B)). Under the ESA, NMFS has issued interim guidance on the term “harass,” defining it as an action that “creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.”

The likelihood of injury due to disruption of normal behaviors would depend on many factors, such as the duration of the response, from what the animal is being diverted, and life history of the animal. Due to the nature of behavioral response research to date, it is not currently possible to ascertain the types of observed reactions that would lead to an abandonment or significant alteration of a natural behavior pattern. Therefore, the Navy has developed a methodology to estimate the possible significance of behavioral reactions and impacts on natural behavior patterns.

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

Moderate severity responses could become significant if sustained over a longer duration. What constitutes a long-duration response is different for each situation and species, although it is likely dependent upon the magnitude of the response and species characteristics such as age, body size, feeding strategy, and behavioral state at the time of the exposure. In general, a response could be considered “long-duration” if it lasted for tens of minutes to a few hours, or enough time to significantly disrupt an animal’s daily routine.

Moderate severity responses included:

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

For the derivation of behavioral criteria, a significant duration was defined as a response that lasted for the duration of exposure or longer, regardless of how long the exposure session may have been. This assumption was made because it was not possible to tell if the behavioral responses would have continued if the exposure had continued. The costs associated with these observed behavioral reactions were not measured so it is not possible to judge whether reactions would have risen to the level of significance as defined above, although it was conservatively assumed the case. High severity responses include those responses with immediate consequences (e.g., stranding, mother-calf separation), and were always considered significant behavioral reactions regardless of duration.

Marine mammal species were placed into behavioral criteria groups based on their known or suspected behavioral sensitivities to sound (Figure 3.8-8 through Figure 3.8-11). In most cases, these divisions are driven by taxonomic classifications (e.g., mysticetes, pinnipeds). The Odontocete group combines most of the mid- and high-frequency cetaceans, without the beaked whales or harbor porpoises, while the Pinniped group combines the otariids and phocids. These groups are combined as there are not enough data to separate them for behavioral responses.

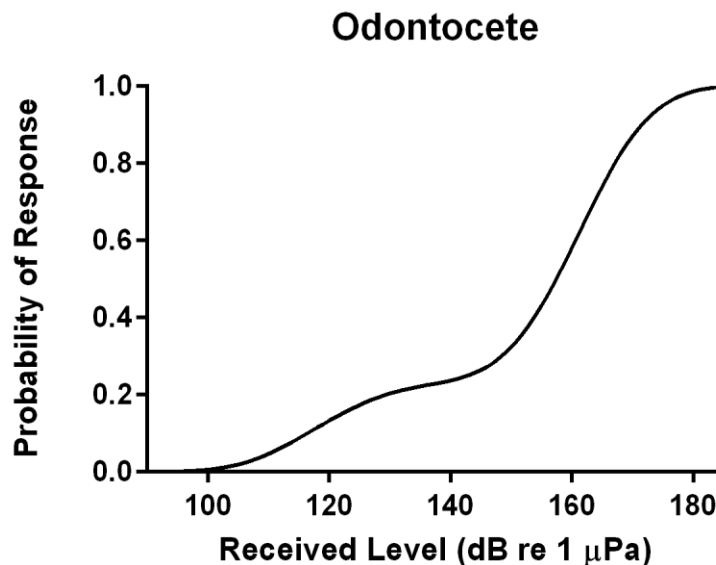


Figure 3.8-8: Behavioral Response Function for Odontocetes

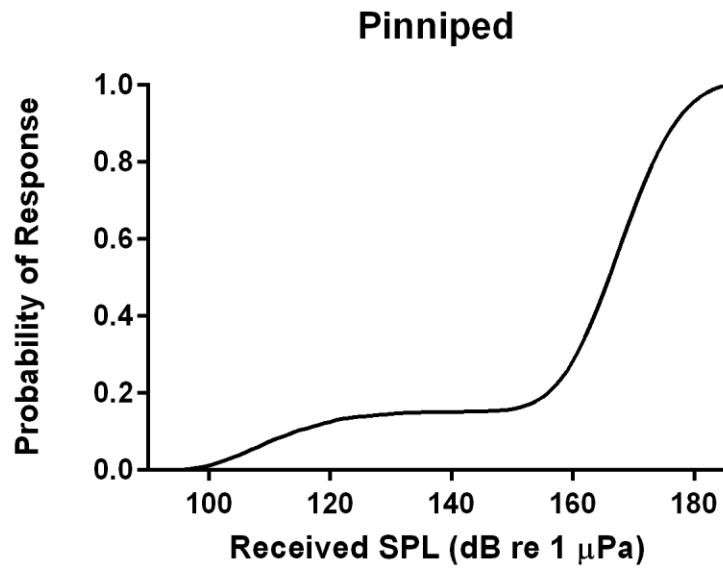


Figure 3.8-9: Behavioral Response Function for Pinnipeds

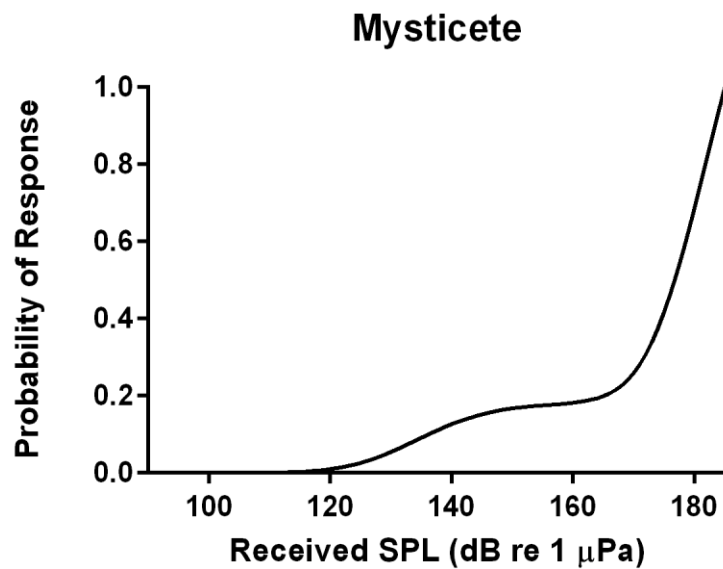
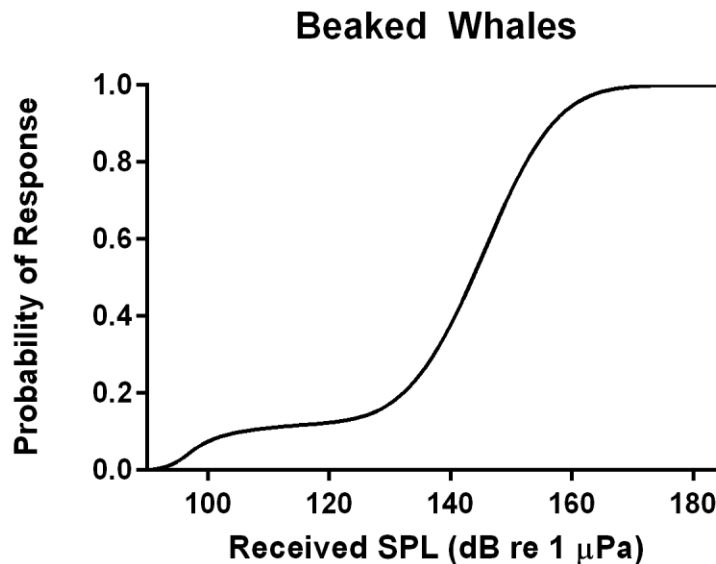


Figure 3.8-10: Behavioral Response Function for Mysticetes



**Figure 3.8-11: Behavioral Response Function for Beaked Whales**

The information currently available regarding harbor porpoises suggests a very low threshold level of response for both captive and wild animals. Threshold levels at which both captive (Kastelein et al., 2000; Kastelein et al., 2005b) and wild harbor porpoises (Johnston, 2002) responded to sound (e.g., acoustic harassment devices, acoustic deterrent devices, or other non-impulsive sound sources) are very low, approximately 120 dB re 1  $\mu$ Pa. Therefore, a SPL of 120 dB re 1  $\mu$ Pa is used in this analysis as a threshold for predicting behavioral responses in harbor porpoises.

Although there is no research on the effects of sonar on sea otters, based on their low reactivity to other acoustic and anthropogenic stressors, sea otters exposed to sonar received levels below the threshold for TTS are assumed to be unlikely to exhibit behavioral responses that would be considered “harassment” under the MMPA for military readiness activities.

The behavioral response functions only consider one aspect of an acoustic exposure, the received level. While the behavioral response functions applied in this analysis are an improvement from historical behavioral step functions (Tyack & Thomas, 2019), marine mammal behavioral response research suggests that the context of an exposure also affects a potential response (Ellison et al., 2011; also Section 3.8.3.1.1.5, Behavioral Reactions). The distance between the animal and the sound source is a strong factor in determining that animal’s potential reaction (e.g., DeRuiter et al., 2013b). For all taxa, therefore, distances beyond which significant behavioral responses to sonar and other transducers are unlikely to occur, denoted as “cutoff distances,” were defined based on existing data (Table 3.8-3). These cutoff distances include even the most distant detected responses to date (e.g., 28 km in northern bottlenose whales (Wensveen et al., 2019)). For training activities that contain multiple platforms or tactical sonar sources that exceed 215 dB re 1  $\mu$ Pa at 1 m, this cutoff distance is substantially increased (i.e., doubled) from values derived from the literature. The use of multiple platforms and intense sound sources are factors that probably increase responsiveness in marine mammals overall. There are currently few behavioral observations under these circumstances; therefore, the Navy will conservatively predict significant behavioral responses at farther ranges for these more intense activities.



**Table 3.8-3: Cutoff Distances for Moderate Source Level, Single Platform Training Events and for All Other Events with Multiple Platforms or Sonar with Source Levels at or Exceeding 215 dB re 1  $\mu$ Pa at 1 m**

<i>Criteria Group</i>	<i>Moderate SL/Single Platform Cutoff Distance</i>	<i>High SL/Multi-Platform Cutoff Distance</i>
Odontocetes	10 km	20 km
Pinnipeds and Mustelids	5 km	10 km
Mysticetes	10 km	20 km
Beaked Whales	25 km	50 km
Harbor Porpoise	20 km	40 km

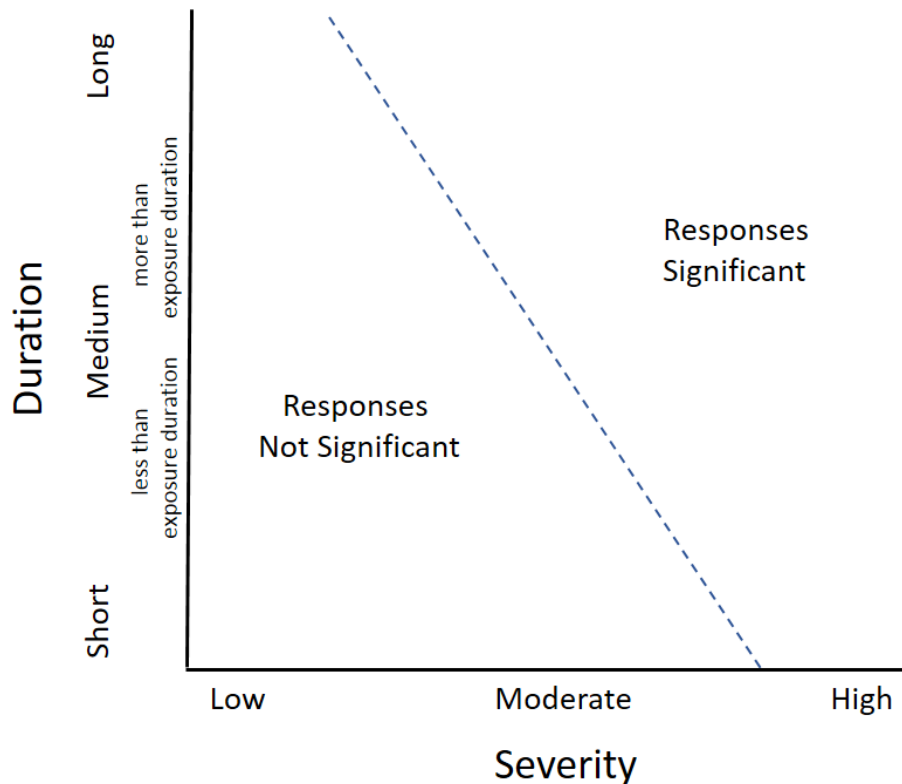
Notes: dB re 1  $\mu$ Pa at 1 m= decibels referenced to 1 micropascal at 1 meter, km= kilometer, SL= source level

### **Assessing the Severity of Behavioral Responses from Sonar Under Military Readiness**

As discussed above, the terms “significant response” or “significant behavioral response” are used in describing behavioral reactions that may lead to an abandonment or significant alteration of a natural behavior pattern. Due to the limited amount of behavioral response research to date and relatively short durations of observation, it is not possible to ascertain the true significance of the majority of the observed reactions. When deriving the behavioral criteria, it was assumed that most reactions that lasted for the duration of the sound exposure or longer were significant, even though many of the exposures lasted for 30 minutes or less. Furthermore, the experimental designs used during many of the behavioral response studies were unlike Navy activities in many important ways. These differences include tagging subject animals, following subjects for sometimes hours before the exposure, vectoring towards the subjects after animals began to avoid the sound source, and making multiple close passes on focal groups. This makes the estimated behavioral impacts from Navy activities using the criteria derived from these experiments difficult to interpret. While the state of science does not currently support definitively distinguishing between significant and insignificant behavioral reactions, as described in the technical report titled *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a), Navy’s analysis incorporates conservative assumptions to account for this uncertainty and therefore likely overestimates the potential impacts.

The estimated behavioral reactions from the Navy’s quantitative analysis are grouped into several categories based on the most powerful sonar source, the number of platforms, the duration, and geographic extent of each Navy activity attributed to the predicted impact.

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. Although the derivation of the Navy’s behavioral criteria did not count low severity responses as significant behavioral responses, in practice, some reactions estimated using the behavioral criteria are likely to be low severity (Figure 3.8-12).



**Figure 3.8-12: Relative Likelihood of a Response Being Significant Based on the Duration and Severity of Behavioral Reactions**

High severity responses are those with a higher potential for direct consequences to growth, survivability, or reproduction. Examples include prolonged separation of females and dependent offspring, panic, flight, stampede, or stranding. High severity reactions would always be considered significant; however, these types of reactions are probably rare under most conditions and may still not lead to direct consequences on survivability. For example, a separation of a killer whale mother-calf pair was observed once during a behavioral response study to an active sonar source (Miller et al., 2014), but the animals were rejoined as soon as the ship had passed. Therefore, although this was a severe response, it did not lead to a negative outcome. Five beaked whale strandings have also occurred associated with U.S. Navy active sonar use as discussed above (see Section 3.8.3.1.1.6, Stranding), but the confluence of factors that contributed to those strandings is now better understood, and the avoidance of those factors has resulted in no known marine mammal strandings associated with U.S. Navy sonar activities for over a decade. The Navy is unable to predict these high severity responses for any activities since the probability of occurrence is apparently very low, although the Navy acknowledges that severe reactions could occasionally occur. In fact, no significant behavioral responses such as panic, stranding or other severe reactions have been observed during monitoring of actual training activities.

Many of the responses estimated using the Navy's quantitative analysis are most likely to be moderate severity. Moderate severity responses would be considered significant if they were sustained for a duration long enough that it caused an animal to be outside of normal daily variations in feeding, reproduction, resting, migration/movement, or social cohesion. As mentioned previously, the behavioral response functions used within the Navy's quantitative analysis were primarily derived from

experiments using short-duration sound exposures lasting, in many cases, for less than 30 minutes. If animals exhibited moderate severity reactions for the duration of the exposure or longer, then it was conservatively assumed that the animal experienced a significant behavioral reaction. However, the experiments did not include measurements of costs to animals beyond the immediately observed reactions, and no direct correlations exist between an observed behavioral response and a cost that may result in long-term consequences. Within the Navy's quantitative analysis, many behavioral reactions are estimated from exposure to sonar that may exceed an animal's behavioral threshold for only a single ping to several minutes. While the state of science does not currently support definitively distinguishing between significant and insignificant behavioral reactions, as described in the technical report titled *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a), the Navy's analysis incorporates conservative assumptions to account for this uncertainty and therefore likely overestimates the potential impacts.

### **Accounting for Mitigation**

The Navy will implement mitigation measures to avoid or reduce potential impacts from active sonar on marine mammals, as described in Section 5.3.2.1 (Active Sonar). The benefits of mitigation are conservatively factored into the analysis for Alternative 1 of the Proposed Action. Procedural mitigation measures include a power down or shut down (i.e., power off) of applicable active sonar sources when a marine mammal is observed in a mitigation zone. The mitigation zones for active sonar activities were designed to avoid the potential for marine mammals to be exposed to levels of sound that could result in auditory injury (i.e., PTS) from active sonar to the maximum extent practicable. The mitigation zones for active sonar extend beyond the respective average ranges to auditory injury (including PTS). Therefore, the impact analysis considers the potential for procedural mitigation to reduce the risk of PTS. Two factors are considered when quantifying the effectiveness of procedural mitigation: (1) the extent to which the type of mitigation proposed for a sound-producing activity (e.g., active sonar) allows for observation of the mitigation zone prior to and during the activity; and (2) the sightability of each species that may be present in the mitigation zone, which is determined by species-specific characteristics and the viewing platform. A detailed explanation of the analysis is provided in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (U.S. Department of the Navy, 2018d).

The impact analysis does not consider the potential for mitigation to reduce TTS or behavioral effects, even though mitigation could also reduce the likelihood of these effects. In practice, mitigation also protects all unobserved (below the surface) animals in the vicinity, including other species, in addition to the observed animal. However, the analysis assumes that only animals sighted at the water surface would be protected by the applied mitigation. The analysis, therefore, does not capture the protection afforded to all marine species that may be near or within the mitigation zone.

The ability to observe the ranges to PTS was estimated for each training event. The ability of Navy Lookouts to detect marine mammals within a mitigation zone is dependent on the animal's presence at the surface and the characteristics of the animal that influence its sightability (such as group size or surface active behavior). The behaviors and characteristics of some species may make them easier to detect. Certain behaviors, such as leaping and breaching, are visible from a great distance and likely increase sighting distances and detections of those species. Environmental conditions under which the training activity could take place are also considered, such as sea surface conditions, weather (e.g., fog or rain), and day versus night.

The Navy will also implement mitigation measures for certain active sonar activities within the North Pacific Right Whale Mitigation Area from June 1 through September 30, as described in Section 5.4 (Geographic Mitigation to be Implemented). Mitigation areas are designed to help avoid or reduce impacts during biologically important life processes within particularly important habitat areas. The benefits of mitigation areas are discussed qualitatively in terms of the context of impact avoidance or reduction.

### **Marine Mammal Avoidance of Sonar and other Transducers**

Because a marine mammal is assumed to initiate avoidance behavior after an initial startle reaction when exposed to relatively high received levels of sound, a marine mammal could reduce its cumulative sound energy exposure over a sonar event with multiple pings (i.e., sound exposures). This would reduce risk of both PTS and TTS, although the quantitative analysis conservatively only considers the potential to reduce instances of PTS by accounting for marine mammals swimming away to avoid repeated high-level sound exposures. All reductions in PTS impacts from likely avoidance behaviors are instead considered TTS impacts.

#### **3.8.3.1.2.2 Impact Ranges for Sonar and Other Transducers**

The following section provides range to effects for sonar and other transducers to specific criteria determined using the Navy Acoustic Effects Model. Marine mammals within these ranges would be predicted to receive the associated effect. Range to effects is important information in not only predicting acoustic impacts, but also in verifying the accuracy of model results against real-world situations and assessing the level of impact that will likely be mitigated within applicable mitigation zones.

The ranges are the distance where the threshold is not exceeded at any depth where animals could be present (excluding negligible small convergence points in some instances). Thus, portions of the water column within the ranges shown would not exceed threshold (i.e., the range does not represent a cylinder of effect in the water column). In some instances, a significant portion of the water column within the ranges shown may not exceed threshold. These differences in propagation are captured in the actual estimation of takes within the Navy Acoustic Effects Model.

The ranges to the PTS threshold for an exposure of 30 seconds are shown in Table 3.8-4 relative to the marine mammal's functional hearing group. This period (30 seconds) was chosen based on examining the maximum amount of time a marine mammal would realistically be exposed to levels that could cause the onset of PTS based on platform (e.g., ship) speed and a nominal animal swim speed of approximately 1.5 meters per second. The ranges provided in the table include the average range to PTS, as well as the range from the minimum to the maximum distance at which PTS is possible for each hearing group. Since any hull-mounted sonar, such as the SQS-53, engaged in anti-submarine warfare training would be moving at between 10 and 15 knots and nominally pinging every 50 seconds, the vessel will have traveled a minimum distance of approximately 257 m during the time between those pings (note: 10 knots is the speed used in the Navy Acoustic Effects Model). As a result, there is little overlap of PTS footprints from successive pings, indicating that in most cases, an animal predicted to receive PTS would do so from a single exposure (i.e., ping). For all other bins (besides MF1), PTS ranges are short enough that marine mammals (with a nominal swim speed of approximately 1.5 meters per second) should be able to avoid higher sound levels capable of causing onset PTS within this 30-second period.

For a SQS-53C (i.e., bin MF1) sonar transmitting for 30 seconds at 3 kHz and a source level of 235 dB re 1  $\mu\text{Pa}^2$ s at 1 m, the average range to PTS for the most sensitive species (the high-frequency cetaceans) extends from the source to a range of 180 m. For all other functional hearing groups (low-frequency cetaceans, mid-frequency cetaceans, otariids, phocids and mustelids), 30-second average PTS zones are substantially shorter, as shown in Table 3.8-4. A scenario could occur where an animal does not leave the vicinity of a ship or travels a course parallel to the ship, however, the close distances required make PTS exposure unlikely. For a military vessel moving at a nominal 10 knots, it is unlikely a marine mammal could maintain the speed to parallel the ship and receive adequate energy over successive pings to suffer PTS.

The tables below illustrate the range to TTS for 1, 30, 60, and 120 seconds from five representative sonar systems (Table 3.8-4 through Table 3.8-7). Due to the lower acoustic thresholds for TTS versus PTS, ranges to TTS are longer. Therefore, successive pings can be expected to add together, further increasing the range to TTS onset. For some hearing groups and bins, the ranges to PTS and TTS are zero because the source level is low relative to threshold shift susceptibility at the relevant hearing frequency.

**Table 3.8-4: Range to Permanent Threshold Shift for Three Representative Sonar Systems**

<i>Hearing Group</i>	<i>Approximate PTS (30 seconds) Ranges (meters)<sup>1</sup></i>		
	<i>Sonar bin MF1</i>	<i>Sonar bin MF4</i>	<i>Sonar bin MF5</i>
High-frequency cetaceans	180 (180–180)	31 (30–35)	9 (8–10)
Low-frequency cetaceans	65 (65–65)	13 (0–15)	0 (0–0)
Mid-frequency cetaceans	16 (16–16)	3 (3–3)	0 (0–0)
Otariids and Mustelids	6 (6–6)	0 (0–0)	0 (0–0)
Phocids	45 (45–45)	11 (11–11)	0 (0–0)

<sup>1</sup>PTS ranges extend from the sonar or other transducer sound source to the indicated distance. The average range to PTS is provided as well as the range from the estimated minimum to the maximum range to PTS in parenthesis.

Notes: MF = mid-frequency, PTS = permanent threshold shift seals are separated from other phocids due to their dive behavior, which is much deeper than the other phocids analyzed

**Table 3.8-5: Ranges to Temporary Threshold Shift for Sonar Bin MF1 over a Representative Range of Environments Within the Study Area**

<i>Hearing Group</i>	<i>Approximate TTS Ranges (meters)<sup>1</sup></i>			
	<i>Sonar Bin MF1</i>			
	<i>1 second</i>	<i>30 seconds</i>	<i>60 seconds</i>	<i>120 seconds</i>
High-frequency cetaceans	3,554 (1,525–6,775)	3,554 (1,525–6,775)	5,325 (2,275–9,525)	7,066 (2,525–13,025)
Low-frequency cetaceans	920 (850–1,025)	920 (850–1,025)	1,415 (1,025–2,025)	2,394 (1,275–4,025)
Mid-frequency cetaceans	209 (200–210)	209 (200–210)	301 (300–310)	376 (370–390)
Otariids and Mustelids	65 (65–65)	65 (65–65)	100 (100–110)	132 (130–140)
Phocids	673 (650–725)	673 (650–725)	988 (900–1,025)	1,206 (1,025–1,525)

<sup>1</sup>Ranges to TTS represent the model predictions in different areas and seasons within the Study Area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.

Notes: HF = high frequency, TTS = temporary threshold shift

**Table 3.8-6: Ranges to Temporary Threshold Shift for Sonar Bin MF4 over a Representative Range of Environments Within the Study Area**

<i>Hearing Group</i>	<i>Approximate TTS Ranges (meters)<sup>1</sup></i>			
	<i>Sonar Bin MF4</i>			
	<i>1 second</i>	<i>30 seconds</i>	<i>60 seconds</i>	<i>120 seconds</i>
High-frequency cetaceans	318 (220–550)	686 (430–1,275)	867 (575–1,525)	1,225 (825–2,025)
Low-frequency cetaceans	77 (0–100)	175 (130–340)	299 (190–550)	497 (280–1,000)
Mid-frequency cetaceans	22 (22–22)	35 (35–35)	50 (50–50)	71 (70–75)
Otariids and Mustelids	8 (8–8)	15 (15–15)	19 (19–19)	25 (25–25)
Phocids	67 (65–70)	123 (110–150)	172 (150–210)	357 (240–675)

<sup>1</sup>Ranges to TTS represent the model predictions in different areas and seasons within the Study Area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.

Notes: HF = high frequency, TTS = temporary threshold shift

**Table 3.8-7: Ranges to Temporary Threshold Shift for Sonar Bin MF5 over a Representative Range of Environments Within the Study Area**

<i>Hearing Group</i>	<i>Approximate TTS Ranges (meters)<sup>1</sup></i>			
	<i>Sonar Bin MF5</i>			
	<i>1 second</i>	<i>30 seconds</i>	<i>60 seconds</i>	<i>120 seconds</i>
High-frequency cetaceans	117 (110–140)	117 (110–140)	176 (150–320)	306 (210–800)
Low-frequency cetaceans	9 (0–12)	9 (0–12)	13 (0–17)	19 (0–24)
Mid-frequency cetaceans	5 (0–9)	5 (0–9)	12 (11–13)	18 (17–18)
Otariids and Mustelids	0 (0–0)	0 (0–0)	0 (0–0)	0 (0–0)
Phocids	9 (8–10)	9 (8–10)	14 (14–15)	21 (21–22)

<sup>1</sup>Ranges to TTS represent the model predictions in different areas and seasons within the Study Area. The zone in which animals are expected to suffer TTS extend from onset-PTS to the distance indicated. The average range to TTS is provided as well as the range from the estimated minimum to the maximum range to TTS in parenthesis.

Notes: HF = high frequency, TTS = temporary threshold shift

The range to received sound levels in 6 dB steps from five representative sonar bins and the percentage of animals that may exhibit a significant behavioral response under each behavioral response function (or step function in the case of the harbor porpoise) are shown in Table 3.8-8 through Table 3.8-10, respectively. See Section 3.8.3.1.2.1 (Methods for Analyzing Impacts from Sonar and Other Transducers) for details on the derivation and use of the behavioral response functions, thresholds, and the cutoff distances.

**Table 3.8-8: Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF1 over a Representative Range of Environments Within the Study Area**

<i>Received Level (dB re 1 <math>\mu</math>Pa)</i>	<i>Mean Range (meters) with Minimum and Maximum Values in Parentheses</i>	<i>Probability of Behavioral Response for Sonar Bin MF1</i>				
		<i>Beaked whales</i>	<i>Harbor Porpoise</i>	<i>Mysticetes</i>	<i>Odontocetes</i>	<i>Pinnipeds</i>
196	105 (100–110)	100%	100%	100%	100%	100%
190	240 (240–240)	100%	100%	98%	100%	100%
184	498 (490–525)	100%	100%	88%	99%	98%
178	1,029 (950–1,275)	100%	100%	59%	97%	92%
172	3,798 (1,525–7,025)	99%	100%	30%	91%	76%
166	8,632 (2,775–14,775)	97%	100%	20%	78%	48%
160	15,000 (3,025–26,525)	93%	100%	18%	58%	27%
154	23,025 (3,275–47,775)	83%	100%	17%	40%	18%
148	47,693 (10,275–54,025)	66%	100%	16%	29%	16%
142	53,834 (12,025–72,025)	45%	100%	13%	25%	15%
136	60,035 (13,275–74,525)	28%	100%	9%	23%	15%
130	72,207 (14,025–75,025)	18%	100%	5%	20%	15%
124	73,169 (17,025–75,025)	14%	100%	2%	17%	14%
118	72,993 (25,025–75,025)	12%	0%	1%	12%	13%
112	72,940 (27,525–75,025)	11%	0%	0%	6%	9%
106	73,016 (28,525–75,025)	11%	0%	0%	3%	5%
100	73,320 (30,025–75,025)	8%	0%	0%	1%	2%

Notes: (1) Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms.

(2) dB re 1  $\mu$ Pa = decibels referenced to 1 micropascal, MF = mid-frequency



**Table 3.8-9: Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF4 over a Representative Range of Environments Within the Study Area**

<i>Received Level (dB re 1 <math>\mu</math>Pa)</i>	<i>Mean Range (meters) with Minimum and Maximum Values in Parentheses</i>	<i>Probability of Behavioral Response for Sonar Bin MF4</i>				
		<i>Beaked whales</i>	<i>Harbor Porpoise</i>	<i>Mysticetes</i>	<i>Odontocetes</i>	<i>Pinnipeds</i>
196	8 (0–8)	100%	100%	100%	100%	100%
190	17 (0–17)	100%	100%	98%	100%	100%
184	34 (0–35)	100%	100%	88%	99%	98%
178	69 (0–75)	100%	100%	59%	97%	92%
172	156 (120–190)	99%	100%	30%	91%	76%
166	536 (280–1,000)	97%	100%	20%	78%	48%
160	1,063 (470–1,775)	93%	100%	18%	58%	27%
154	2,063 (675–4,275)	83%	100%	17%	40%	18%
148	5,969 (1,025–9,275)	66%	100%	16%	29%	16%
142	12,319 (1,275–26,025)	45%	100%	13%	25%	15%
136	26,176 (1,775–40,025)	28%	100%	9%	23%	15%
130	42,963 (2,275–54,775)	18%	100%	5%	20%	15%
124	53,669 (2,525–65,775)	14%	100%	2%	17%	14%
118	63,387 (2,775–75,025)	12%	0%	1%	12%	13%
112	71,709 (3,025–75,025)	11%	0%	0%	6%	9%
106	73,922 (22,775–75,025)	11%	0%	0%	3%	5%
100	73,923 (25,525–75,025)	8%	0%	0%	1%	2%

Notes: (1) Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms.  
(2) dB re 1  $\mu$ Pa = decibels referenced to 1 micropascal, MF = mid-frequency

**Table 3.8-10: Ranges to a Potentially Significant Behavioral Response for Sonar Bin MF5 over a Representative Range of Environments Within the Study Area**

Received Level (dB re 1 $\mu$ Pa)	Mean Range (meters) with Minimum and Maximum Values in Parentheses	Probability of Behavioral Response for Sonar Bin MF5				
		Beaked whales	Harbor Porpoise	Mysticetes	Odontocetes	Pinnipeds
196	0 (0–0)	100%	100%	100%	100%	100%
190	1 (0–3)	100%	100%	98%	100%	100%
184	4 (0–7)	100%	100%	88%	99%	98%
178	14 (0–15)	100%	100%	59%	97%	92%
172	29 (0–30)	99%	100%	30%	91%	76%
166	59 (0–65)	97%	100%	20%	78%	48%
160	130 (0–170)	93%	100%	18%	58%	27%
154	349 (0–1,025)	83%	100%	17%	40%	18%
148	849 (410–2,275)	66%	100%	16%	29%	16%
142	1,539 (625–3,775)	45%	100%	13%	25%	15%
136	2,934 (950–8,525)	28%	100%	9%	23%	15%
130	6,115 (1,275–10,275)	18%	100%	5%	20%	15%
124	9,764 (1,525–16,025)	14%	100%	2%	17%	14%
118	13,830 (1,775–24,775)	12%	0%	1%	12%	13%
112	18,970 (2,275–30,775)	11%	0%	0%	6%	9%
106	25,790 (2,525–38,525)	11%	0%	0%	3%	5%
100	36,122 (2,775–46,775)	8%	0%	0%	1%	2%

Notes: (1) Cells are shaded if the mean range value for the specified received level exceeds the distance cut-off range for a particular hearing group. Any impacts within the cut-off range for a criteria group are included in the estimated impacts. Cut-off ranges in this table are for activities with high source levels or multiple platforms. (2) dB re 1  $\mu$ Pa = decibels referenced to 1 micropascal, MF = mid-frequency

#### 3.8.3.1.2.3 Impacts from Sonar and Other Transducers Under the No Action Alternative

Under the No Action Alternative, proposed Navy training activities would not occur within the TMAA. The impacts associated with Navy training activities would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing Navy training activities.

#### 3.8.3.1.2.4 Impacts from Sonar and Other Transducers Under Alternative 1

Sonar and other transducers proposed for use are typically transient and temporary because activities that involve sonar and other transducers take place at different locations and many platforms are generally moving throughout the TMAA. General categories and characteristics of sonar systems and the number of hours these sonars would be operated during training under Alternative 1 are described in Section 3.0.4.1 (Acoustic Sources). Activities using sonars and other transducers would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Navy Activities Descriptions). The proposed use of sonar for training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same

location as analyzed under the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Most estimated impacts are due to anti-submarine warfare sonar activities, which could vary in duration and intensity. The number of hours these sonars would be operated under Alternative 1 is described in Section 3.0.4.1 (Acoustic Sources). Although the existing baseline conditions have not changed appreciably, and no new Navy training activities are proposed in the TMAA in this SEIS/OEIS, a detailed re-analysis of Alternative 1 with respect to marine mammals is provided here to supplant previous analysis based on available new literature, adjusted sound exposure criteria, and new acoustic effects modeling.

### **Presentation of Estimated Impacts from the Quantitative Analysis**

The results of the analysis of potential impacts on marine mammals from sonar and other transducers (Section 3.8.3.1.2.1, Methods for Analyzing Impacts from Sonar and Other Transducers) are discussed below. The numbers of potential impacts estimated for individual species and stocks of marine mammals from exposure to sonar for training activities under Alternative 1 is shown in Appendix C (Estimated Marine Mammal and Sea Turtle Impacts from Exposure to Acoustic and Explosive Stressors Under Navy Training Activities) and presented below in tables for each species of marine mammal with any estimated effects. All impacts from sonar and other transducers within the TMAA are limited to training activities conducted over 21 consecutive days during April to October of any given year. There is a potential for impacts to occur anywhere within the TMAA where sound from sonar and the species overlap. It is important to note when examining the results of the quantitative analysis that the behavioral response functions used to predict the numbers of reactions in this analysis are largely derived from several studies (see Section 3.8.3.1.1.5, Behavioral Reactions). The best available science, including behavioral response studies, was used for deriving these criteria; however, many of the factors inherent in these studies that potentially increased the likelihood and severity of observed responses (e.g., close approaches by multiple vessels, tagging animals, and vectoring towards animals that have already begun avoiding the sound source) would not occur during Navy activities. Because the Navy purposely avoids approaching marine mammals, many of the behavioral responses estimated by the quantitative analysis are unlikely to occur or unlikely to rise to the severity observed during many of the behavioral response studies.

Although the statutory definition of Level B harassment for military readiness activities under the MMPA requires that the natural behavior patterns of a marine mammal be significantly altered or abandoned, the current state of science for determining those thresholds is somewhat unsettled. Therefore, in its analysis of impacts associated with acoustic sources, the Navy is adopting a conservative approach that overestimates the number of takes by Level B harassment. The responses estimated using the Navy's quantitative analysis are most likely to be moderate severity. Moderate severity responses would be considered significant if they were sustained for a duration long enough that it caused an animal to be outside of normal daily variations in feeding, reproduction, resting, migration/movement, or social cohesion. As discussed in Section 3.8.3.1.2.1 (Methods for Analyzing Impacts from Sonar and Other Transducers), the behavioral response functions used within the Navy's quantitative analysis were primarily derived from experiments using short-duration sound exposures lasting, in many cases, for less than 30 minutes. If animals exhibited moderate severity reactions for the duration of the exposure or longer, then it was conservatively assumed that the animal experienced a significant behavioral reaction. However, the experiments did not include measurements of costs to animals beyond the immediately observed reactions, and no direct correlations exist between an observed behavioral response and a cost that may result in long-term consequences. Within the Navy's quantitative analysis, many behavioral reactions are estimated from exposure to sound that may exceed an animal's

behavioral threshold for only a single exposure up to several minutes. It is likely that many of the estimated behavioral reactions within the Navy's quantitative analysis would not constitute significant behavioral reactions; however, the numbers of significant versus non-significant behavioral reactions are currently impossible to predict. Consequently, there is a high likelihood that significant numbers of marine mammals exposed to acoustic sources are not significantly altering or abandoning their natural behavior patterns. As such, the overall impact of acoustic sources from military readiness activities on marine mammal species and stocks is negligible (i.e., cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species or stocks through effects on annual rates of recruitment or survival).

### **Mysticetes**

Mysticetes may be exposed to sound from sonar and other transducers associated with training activities between April and October in the TMAA. Most low- (less than 1 kHz) and mid- (1–10 kHz) frequency sonars and other transducers produce sounds that are likely to be within the hearing range of mysticetes (Section 3.8.2.1.4, Hearing and Vocalization). Some high-frequency sonars (greater than 10 kHz) also produce sounds that should be audible to mysticetes, although only smaller species of mysticetes such as minke whales are likely to be able to hear higher frequencies, presumably up to 30 kHz. Therefore, some high-frequency sonars and other transducers with frequency ranges between 10 and 30 kHz may also be audible to some mysticetes. If a sound is within an animal's hearing range then behavioral reactions, physiological stress, masking and hearing loss are potential impacts that must be analyzed. If a marine mammal cannot hear a sound, then behavioral reactions, physiological stress, masking, or hearing loss is not likely to occur. Impact ranges for mysticetes are discussed under low-frequency cetaceans in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers).

Behavioral reactions in mysticetes resulting from exposure to sonar could occur based on the quantitative analysis. Considering best available data on observed mysticete responses to sound exposure, behavioral responses would not be expected to occur beyond 20 km from events with multiple sound source platforms or high source levels, nor beyond 10 km from moderate source level, single platform events. Any predicted behavioral reactions are much more likely to occur within a few kilometers of the sound source. As discussed above in *Assessing the Severity of Behavioral Responses from Sonar and other Transducers Under Military Readiness*, the quantitative analysis very likely overestimated the numbers of behavioral reactions due to the underlying nature of the data used to derive the behavioral response functions. Research shows that if mysticetes do respond they may react in a number of ways, depending on the characteristics of the sound source, their experience with the sound source, and whether they are migrating or on seasonal grounds (i.e., breeding or feeding). Behavioral reactions may include alerting, breaking off feeding dives and surfacing, or diving or swimming away. Overall, mysticetes have been observed to be more reactive to acoustic disturbance when a noise source is located directly on their migration route. Mysticetes disturbed while migrating could pause their migration or route around the disturbance. Animals disturbed while engaged in other activities such as feeding or reproductive behaviors may be more likely to ignore or tolerate the disturbance and continue their natural behavior patterns. Therefore, behavioral reactions from mysticetes are likely to be short term and low to moderate severity.

Some mysticetes may avoid a larger activity such as a major training exercise as it moves through an area. Vessels and aircraft associated with training activities are typically in transit during an event (they are not stationary) and activities typically do not use the same training locations day after day during multi-day activities. If an event otherwise focuses on a fixed location, mysticetes may avoid the location

of the activity for the duration of the event. If animals are displaced, they would likely return quickly after the event subsides. In the ocean, the use of sonar and other transducers is transient and is unlikely to expose the same population of animals repeatedly over a short period except around homeports and fixed instrumented ranges, which are not present in the TMAA. Overall, a few behavioral reactions per year by a single individual are unlikely to produce long-term consequences for that individual.

Behavioral research indicates that mysticetes most likely avoid sound sources at levels that would cause any hearing loss (i.e., TTS) (Section 3.8.3.1.1.5, Behavioral Reactions). Therefore, it is likely that the quantitative analysis overestimates TTS in marine mammals because it does not account for animals avoiding sound sources at closer ranges. Mysticetes that do experience PTS or TTS from sonar sounds may have reduced ability to detect biologically important sounds around the frequency band of the sonar until their hearing recovers. Recovery from hearing loss begins almost immediately after the noise exposure ceases and can take a few minutes to a few days to fully recover, depending on the magnitude of the initial threshold shift. Temporary Threshold Shift would be recoverable and PTS would leave some residual hearing loss. Most TTS, if it does actually occur, would be more likely to be minor to moderate (i.e., less than 20 dB of TTS directly after the exposure) and would recover within a matter of minutes to hours (see Section 3.8.3.1.1.2, Hearing Loss). Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. During the period that a mysticete had hearing loss, social calls from conspecifics could be more difficult to detect or interpret if they fell in the octave band of the sonar frequency. Killer whales are a primary predator of mysticetes. Some hearing loss could make killer whale calls more difficult to detect at farther ranges until hearing recovers. 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. A single or even a few minor TTS (less than 20 dB of TTS) to an individual mysticete per year are unlikely to have any long-term consequences for that individual.

Research and observations of masking in marine mammals are discussed in Section 3.8.3.1.1.4 (Masking). Most anti-submarine warfare sonars and countermeasures use mid-frequency ranges, and a few use low-frequency ranges. Most of these sonar signals are limited in the temporal, frequency, and spatial domains. The duration of most individual sounds is short, lasting up to a few seconds each. Systems typically operate with low-duty cycles for most tactical sources, but some systems may operate nearly continuously or with higher duty cycles. Nevertheless, masking may be more prevalent at closer ranges to these high-duty cycle and continuous active sonar systems. Most anti-submarine warfare activities are geographically dispersed and last for only a few hours, often with intermittent sonar use even within this period. Most anti-submarine warfare sonars also have a narrow frequency band (typically less than one-third octave). These factors reduce the likelihood of sources causing significant masking in mysticetes. High-frequency (greater than 10 kHz) sonars fall outside of the best hearing and vocalization ranges of mysticetes (see Section 3.8.2.1.4, Hearing and Vocalization). Furthermore, high frequencies (above 10 kHz) attenuate more rapidly in the water due to absorption than do lower frequency signals, thus producing only a small zone of potential masking. High-frequency sonars are typically used for mine hunting, navigation, and object detection (avoidance). Masking in mysticetes due to exposure to high-frequency sonar is unlikely. Potential costs to mysticetes from masking are similar to those discussed above for mild to moderate levels of TTS, with the primary difference being that the effects of masking are only present when the sound source (i.e., sonar) is actively pinging and the effect is over the moment the sound has ceased. By contrast, hearing loss lasts beyond the exposure for a period. Nevertheless, mysticetes that do experience some masking for a short period from low- or

mid-frequency sonar may have their ability to communicate with conspecifics reduced, especially at further ranges. However, larger mysticetes (e.g., blue whale, fin whale, sei whale) communicate at frequencies below those of mid-frequency sonar and even most low-frequency sonars. Mysticetes that communicate at higher frequencies (e.g., minke whale) may be affected by some short-term and intermittent masking. Sounds from mid-frequency sonar could mask killer whale vocalizations, making them more difficult to detect, especially at further ranges. It is unknown whether masking would affect a mysticete's ability to feed since it is unclear how or if mysticetes use sound for finding prey or feeding. A single or even a few short periods of masking, if it were to occur, to an individual mysticete per year are unlikely to have any long-term consequences for that individual.

**North Pacific Right Whales (Endangered Species Act-Listed)**

North Pacific right whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although North Pacific right whales are considered rare in the TMAA due to their low abundance, their occurrence in the TMAA is year round, and they are most likely to be present June through September. The quantitative analysis estimates TTS under Alternative 1 (Table 3.8-11). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the Eastern North Pacific Stock (Table 3.8-11).

As described for mysticetes above, even if an individual right whale experiences TTS a couple times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. In addition to implementing procedural mitigation for active sonar, from June through September (i.e., the months when North Pacific right whales are most likely to be present in the TMAA), the Navy will not use surface ship hull-mounted MF1 mid-frequency active sonar within the North Pacific Right Whale Mitigation Area. This mitigation area encompasses the portion of the biologically important habitat identified by Ferguson et al. (2015) for North Pacific right whale feeding that overlaps the TMAA. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of North Pacific right whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed North Pacific right whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-11: Estimated Impacts on Individual North Pacific Right Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Eastern North Pacific	0	2	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Humpback Whales (some DPSs are Endangered Species Act-Listed)**

Humpback whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although the timing of humpback whale migrations may change year to year, they are most likely to be present in the TMAA June through September. Impacts have been modeled for the Hawaii DPS (Central North Pacific stock) population of humpback whales, which are not ESA-listed, and for the Mexico DPS (California, Oregon, and Washington stock) and Western North Pacific DPS (Western North Pacific stock) populations of humpback whales, which are ESA-listed.

The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-12). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to multiple stocks (Table 3.8-12). In addition to procedural mitigation, the Navy will implement mitigation within mitigation areas, which will further help avoid or reduce potential impacts from active sonar on humpback whales. The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft operating within the TMAA to the possible presence of increased concentrations of humpback whales from June 1 to September 30. To maintain safety of navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of large whales that may be vulnerable to vessel strikes or potential impacts from training activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training activities and to aid in the implementation of procedural mitigation during activities using active sonar. The Navy will not use surface ship hull-mounted MF1 mid-frequency active sonar from June 1 to September 30 within the North Pacific Right Whale Mitigation Area, which overlaps a portion of the proposed humpback whale critical habitat.

As described for mysticetes above, minor to moderate TTS or behavioral reactions to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

Sound from sonars and other transducers during training activities would overlap proposed critical habitat for the ESA-listed Mexico DPS, and Western North Pacific DPS of humpback whales in the TMAA (whales belonging to the Central America DPS should not be present in the GOA or the TMAA; see National Marine Fisheries Service (2016d, 2019b, 2019c)). As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), one essential feature was identified for humpback whale critical habitat, and that essential feature is defined as prey species, primarily euphausiids and small pelagic schooling fishes, of sufficient quality, abundance, and accessibility within humpback whale feeding areas

to support feeding and population growth. This essential feature would not be adversely affected by sonar use proposed in this action, as follows.

In the TMAA, the humpback whale's diet is consistently dominated by euphausiids and small pelagic fishes, such as northern anchovy, Pacific herring, Pacific sardine, and capelin (Fleming et al., 2016; Gabriele et al., 2017; Keen et al., 2017; Santora et al., 2010; Straley et al., 2017; Szabo, 2015; Witteveen & Wynne, 2017). As described in Section 3.6 (Fishes), non-impulsive sound sources, such as sonar and other transducers, have not been known to cause direct injury or mortality to fish under conditions that would be found in the wild (Halvorsen et al., 2012; Kane et al., 2010; Popper et al., 2007) and would only be expected to result in behavioral reactions or potential masking in marine invertebrates. Most sources proposed for use during training activities overlapping the critical habitat in the TMAA would not fall within the frequency range of marine invertebrate or fish hearing, thereby presenting no plausible route of effect on either species. The few sources used within invertebrate and fish hearing ranges would be limited, temporary, and transient, as described in Appendix A (Navy Activities Descriptions) and examined in Section 3.6.3.1.2 (Impacts from Sonar and Other Transducers) of Section 3.6 (Fishes). Additionally, the use of active sonar would not chronically elevate background noise or cause a reduction in foraging space in critical habitat for humpback whales. Brief periods of masking due to spatially and temporally isolated exposures are accounted for in the quantitative assessment of the potential for direct behavioral disturbance as a level-based response, as explained in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017d).

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of humpback whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed humpback whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA. The use of sonar and other transducers during training activities would have no effect on proposed critical habitat for humpback whales.*

**Table 3.8-12: Estimated Impacts on Individual Humpback Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
California, Oregon, & Washington	1	8	0
Central North Pacific	4	66	0
Western North Pacific	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.



**Blue Whales (Endangered Species Act-Listed)**

Blue whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although blue whales' occurrence in the TMAA is year round, they are most likely to be present June through December. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-13). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to multiple stocks (Table 3.8-13).

As described for mysticetes above, minor to moderate behavioral reactions and TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of blue whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed blue whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-13: Estimated Impacts on Individual Blue Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Central North Pacific	0	3	0
Eastern North Pacific	3	32	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Fin Whales (Endangered Species Act-Listed)**

Fin whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although fin whales' occurrence in the TMAA is year round, they are most likely to be present June through August. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-14). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the Northeast Pacific stock (Table 3.8-14).

As described for mysticetes above, minor to moderate TTS or behavioral reactions to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of fin whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed fin whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-14: Estimated Impacts on Individual Fin Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
Stock	Behavioral	TTS	PTS
Northeast Pacific	104	1,125	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

#### **Sei Whales (Endangered Species Act-Listed)**

Sei whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although sei whales' occurrence in the TMAA is year round, they are considered rare, even during the summer time period. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-15). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the Eastern North Pacific stock (Table 3.8-15).

As described for mysticetes above, minor to moderate TTS or behavioral reactions to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of sei whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed sei whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-15: Estimated Impacts on Individual Sei Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
Stock	Behavioral	TTS	PTS
Eastern North Pacific	2	34	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### Minke Whales

Minke whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Even though very few minke whales have been seen during surveys in the area, their occurrence in the TMAA is considered year round. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-16). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the Alaska stock (Table 3.8-16).

As described for mysticetes above, minor to moderate TTS or behavioral reactions to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of minke whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-16: Estimated Impacts on Individual Minke Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska	4	44	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### Gray Whales (one DPS is Endangered Species Act-Listed)

Gray whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although Western North Pacific gray whales are rare, both stocks of gray whales are migratory and their occurrence in the TMAA would be seasonal with their highest likelihood of occurring being between June and August. Impacts have been modeled for the Eastern North Pacific stock of gray whales, which are not ESA-listed, and for the Western North Pacific stock of gray whales, which are ESA-listed.

The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). In addition to procedural mitigation, the Navy will implement mitigation within mitigation areas, which will further help avoid the already low potential for impacts from active sonar on gray whales. The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft operating within the TMAA to the possible presence of increased concentrations of gray whales from April 1 to August 31. To maintain safety of navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of large whales that may be vulnerable to vessel strikes or potential impacts from training activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training activities and to aid in the implementation of procedural mitigation during activities using active sonar. This mitigation

area overlaps habitat within the northernmost corner and southwestern edge of the TMAA that has been identified by Ferguson et al. (2015) as biologically important gray whale migration habitat. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of gray whales.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed gray whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

### **Odontocetes**

Odontocetes may be exposed to sound from sonar and other transducers associated with training activities throughout the year. Low- (less than 1 kHz), mid- (1–10 kHz), high-frequency (10–100 kHz), and very high-frequency (100–200 kHz) sonars produce sounds that are likely to be within the audible range of odontocetes (see Section 3.8.2.1.4, Hearing and Vocalization). If a sound is within an animal's hearing range, then behavioral reactions, physiological stress, masking and hearing loss are potential impacts that must be analyzed. If a marine mammal cannot hear a sound, then behavioral reactions, physiological stress, masking, or hearing loss could not occur. Impact ranges for odontocetes are discussed under mid-frequency cetaceans in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers).

Behavioral reactions in odontocetes (except beaked whales and harbor porpoise) resulting from exposure to sonar could take place at distances of up to 20 km. Beaked whales and harbor porpoise have demonstrated a high level of sensitivity to human-made noise and activity; therefore, the quantitative analysis assumes that some harbor porpoises and some beaked whales could experience significant behavioral reactions at a distance of up to 50 km from the sound source. Behavioral reactions, however, are much more likely within a few kilometers of the sound source for most species of odontocetes such as delphinids and sperm whales. Even for harbor porpoise and beaked whales, as discussed above in *Assessing the Severity of Behavioral Responses from Sonar Under Military Readiness*, the quantitative analysis has very likely overestimated the numbers of behavioral reactions due to the underlying nature of the data used to derive the behavioral response functions.

Research shows that if odontocetes do respond they may react in a number of ways, depending on the characteristics of the sound source and their experience with the sound source. Behavioral reactions may include alerting; breaking off feeding dives and surfacing; or diving or swimming away. Animals disturbed while engaged in other activities such as feeding or reproductive behaviors may be more likely to ignore or tolerate the disturbance and continue their natural behavior patterns. Therefore, most behavioral reactions from odontocetes are likely to be short term and low to moderate severity.

Large odontocetes such as killer whales and pilot whales have been the subject of behavioral response studies (see Section 3.8.3.1.1.5, Behavioral Reactions). Based on these studies, a number of reactions could occur such as a short-term cessation of natural behavior such as feeding, avoidance of the sound source, or even attraction towards the sound source as seen in pilot whales. Due to the factors involved in Navy training exercises versus the conditions under which pilot whales and killer whales were exposed during behavioral response studies, large odontocetes are unlikely to have more than short-term and moderate severity reactions to sounds from sonar or other human disturbance, and typically only at ranges within a few kilometers. Most estimated impacts are due to anti-submarine warfare sonar

activities. Major training exercises involve multiple sonar systems and can last for a period of days, making significant response more likely. A single or few short-lived TTS or behavioral reactions per year are unlikely to have any significant costs or long-term consequences for individuals.

Small odontocetes have been the subject of behavioral response studies and observations in the field (see Section 3.8.3.1.1.5, Behavioral Reactions). Based on these studies, small odontocetes (dolphins) appear to be less sensitive to sound and human disturbance than other cetacean species. If reactions did occur, they could consist of a short-term behavior response such as cessation of feeding, avoidance of the sound source, or even attraction towards the sound source. Small odontocetes are unlikely to have more than short-term and moderate severity reactions to sounds from sonar or other human disturbance, and typically only at ranges within a few kilometers. Most estimated impacts are due to anti-submarine warfare sonar activities, which could vary in duration and intensity. Major training exercises involve multiple sonar systems and can last for a period of days, making significant response more likely. A single or few short-lived TTS or behavioral reactions per year are unlikely to have any significant costs or long-term consequences for individuals.

Some odontocetes may avoid larger activities such as a major training exercise as it moves through an area. Vessels and aircraft associated with training activities are typically in transit during an event (they are not stationary) and activities typically do not use the same training locations day-after-day during multi-day activities. If an event otherwise focuses on a fixed location, sensitive species of odontocetes, such as beaked whales, may avoid the location of the activity for the duration of the event.

Section 3.8.3.1.1.5 (Behavioral Reactions) discusses these species' observed reactions to sonar and other transducers. If animals are displaced, they would likely return after the sonar activity subsides within an area, as seen in Blainville's beaked whales in the Bahamas (Tyack et al., 2011) and Hawaii (Henderson et al., 2015; Henderson et al., 2016; Manzano-Roth et al., 2016). This would allow the animal to recover from any energy expenditure or missed resources, reducing the likelihood of long-term consequences for the individual. Outside of Navy instrumented ranges and homeports, the use of sonar and other transducers is transient and is unlikely to expose the same population of animals repeatedly over a short period. However, a few behavioral reactions per year from a single individual are unlikely to produce long-term consequences for that individual.

Behavioral research indicates that most odontocetes avoid sound sources at levels that would cause any temporary hearing loss (i.e., TTS) (see Section 3.8.3.1.1.5, Behavioral Reactions). TTS and even PTS is more likely for high-frequency cetaceans, such as Dall's porpoises and harbor porpoises, because hearing loss thresholds for these animals are lower than for all other marine mammals. These species, especially harbor porpoises, have demonstrated a high level of sensitivity to human-made sound and activities and may avoid at further distances. This increased distance could avoid or minimize hearing loss for these species as well, especially as compared to the estimates from the quantitative analysis. Therefore, it is likely that the quantitative analysis overestimates TTS and PTS in marine mammals because it does not account for animals avoiding sound sources at closer ranges. Recovery from hearing loss begins almost immediately after the noise exposure ceases and can take a few minutes to a few days to fully recover, depending on the magnitude of the initial threshold shift. TTS would be recoverable and PTS would leave some residual hearing loss. Most TTS, if it does actually occur, would be more likely to be minor to moderate (i.e., less than 20 dB of TTS directly after the exposure) and would recover within a matter of minutes to hours. Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. During the period that an odontocete had hearing loss, social calls from

conspecifics could be more difficult to detect or interpret. Killer whales are a primary predator of odontocetes. Some hearing loss could make killer whale calls more difficult to detect at further ranges until hearing recovers. Odontocetes use echolocation clicks to find and capture prey. These echolocation clicks and vocalizations are at frequencies above a few tens of kHz for delphinids, beaked whales, and sperm whales, and above 100 kHz for porpoises. Echolocation associated with feeding and navigation in odontocetes is unlikely to be affected by threshold shift at lower frequencies and should not have any significant effect on an odontocete's ability to locate prey or navigate, even in the short term. Therefore, a single or even a few minor TTS (less than 20 dB of TTS) to an individual odontocete per year are unlikely to have any long-term consequences for that individual. Minor PTS (a few dB or less) in an individual could have no to minor long-term consequences for individuals.

Research and observations of masking in marine mammals are discussed in Section 3.8.3.1.1.4 (Masking). Many anti-submarine warfare sonars and countermeasures use low- and mid-frequency sonar. Most low- and mid-frequency sonar signals (i.e., sounds) are limited in their temporal, frequency, and spatial domains. The duration of most individual sounds is short, lasting up to a few seconds each. Some systems operate with higher duty cycles or nearly continuously, but typically use lower power. Nevertheless, masking may be more prevalent at closer ranges to these high-duty cycle and continuous active sonar systems. Most anti-submarine warfare activities are geographically dispersed and last for only a few hours, often with intermittent sonar use even within this period. Most anti-submarine warfare sonars also have a narrow frequency band (typically much less than one-third octave). These factors reduce the likelihood of sources causing significant masking in odontocetes due to exposure to sonar used during anti-submarine warfare activities. Odontocetes may experience some limited masking at closer ranges from high-frequency sonars and other transducers; however, the frequency band of the sonar is narrow, limiting the likelihood of masking. High-frequency sonars are typically used for mine hunting, navigation, and object detection (avoidance). Potential costs to odontocetes from masking are similar to those discussed above for mild to moderate levels of TTS, with the primary difference being that the effects of masking are only present when the sound source (i.e., sonar) is actively pinging and the effect is over the moment the sound has ceased.

Nevertheless, odontocetes that do experience some masking from sonar or other transducers may have their ability to communicate with conspecifics reduced, especially at further ranges. Sounds from mid-frequency sonar could mask killer whale vocalizations, making them more difficult to detect, especially at further ranges. As discussed above for TTS, odontocetes use echolocation to find prey and navigate. The echolocation clicks of odontocetes are above the frequencies of most sonar systems. Therefore, echolocation associated with feeding and navigation in odontocetes is unlikely to be masked by sounds from sonars or other transducers. A single or even a few short periods of masking, if it were to occur, to an individual odontocete per year are unlikely to have any long-term consequences for that individual.

#### **Sperm Whales (Endangered Species Act-Listed)**

Sperm whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although sperm whales' occurrence in the TMAA is year round, they are most likely to be present June through September. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-17). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the North Pacific stock (Table 3.8-17).

As described for odontocetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that

individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of sperm whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed sperm whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-17: Estimated Impacts on Individual Sperm Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
North Pacific	107	5	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Killer Whales**

Killer whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although killer whales' occurrence in the TMAA is year round, the one offshore population and the two transient types are more likely to be present in the majority of the TMAA given the deep and far offshore waters of the Navy training area. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-18). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to multiple stocks (Table 3.8-18).

As described for odontocetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of killer whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-18: Estimated Impacts on Individual Killer Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska Resident	0	0	0
AT1 Transient	0	0	0
Eastern Pacific Offshore	64	17	0
Gulf of Alaska, Aleutian Island, & Bering Sea Transient	119	24	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Pacific White-Sided Dolphins**

Pacific white-sided dolphins may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the Pacific white-sided dolphins' occurrence in the TMAA would be likely year round. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-19). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the North Pacific stock (Table 3.8-19).

As described for odontocetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of Pacific white-sided dolphins incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-19: Estimated Impacts on Individual Pacific White-Sided Dolphin Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
North Pacific	1,102	472	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.



### **Harbor Porpoises**

Harbor porpoises may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the harbor porpoises' occurrence in the TMAA would be likely year round in the nearshore locations to the shelf break. The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers).

TTS and PTS thresholds for high-frequency cetaceans, including Harbor porpoise, are lower than for all other marine mammals, which leads to a higher number of estimated impacts relative to the number of animals exposed to the sound as compared to other hearing groups (e.g., mid-frequency cetaceans). Harbor porpoises are particularly sensitive to human-made noise and disturbance and will avoid sound levels between 120 and 140 dB re 1  $\mu$ Pa at distances up to 30 km for more intense activities (as discussed below). This means that the quantitative analysis greatly overestimates hearing loss in harbor porpoises because most animals would avoid sound levels that could cause TTS or PTS.

Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of harbor porpoises.*

### **Dall's Porpoises**

Dall's porpoises may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the Dall's porpoises' occurrence in the TMAA would be likely year round. The quantitative analysis estimates behavioral reactions, TTS, and PTS under Alternative 1 (Table 3.8-20). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the Alaska stock (Table 3.8-20).

TTS and PTS thresholds for high-frequency cetaceans, including Dall's porpoises, are lower than for all other marine mammals, which leads to a higher number of estimated impacts relative to the number of animals exposed to the sound as compared to other hearing groups (e.g., mid-frequency cetaceans). The information available on harbor porpoise behavioral reactions to human disturbance (a closely related species) suggests that these species may be more sensitive and avoid human activity, and sound sources, to a longer range than most other odontocetes. This would make Dall's porpoises less susceptible to hearing loss; therefore, it is likely that the quantitative analysis over-predicted hearing loss impacts (i.e., TTS and PTS) in Dall's porpoises.

As described for odontocetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. PTS could reduce an animal's ability to detect biologically important sounds; however, as discussed above, hearing loss beyond a minor TTS is unlikely and a small threshold shift due to exposure to sonar is unlikely to affect the hearing range that Dall's porpoise rely upon if it did occur. Nevertheless, PTS could have minor long-term consequences for individuals if it were to occur. This minor consequence for an individual is unlikely to have any long-term consequences for the species or stock. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of Dall's porpoises incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-20: Estimated Impacts on Individual Dall's Porpoise Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska	310	8,710	19

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Beaked Whales**

Beaked whales may be exposed to sounds from sonar and other transducers associated with training activities April through October. Beaked whales within the GOA TMAA include Baird's beaked whale, Cuvier's beaked whale, and Stejneger's beaked whale. Although beaked whales' occurrence in the TMAA would be likely year round, Cuvier's beaked whales are most likely to be present April through June. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-21 through Table 3.8-23). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts to Baird's beaked whales, Cuvier's beaked whale, and Stejneger's beaked whales apply to the Alaska stocks (Table 3.8-21, Table 3.8-22, and Table 3.8-23).

As discussed above for odontocetes overall, the quantitative analysis overestimates hearing loss in marine mammals because behavioral response research has shown that most marine mammals are likely to avoid sound levels that could cause more than minor to moderate TTS (6–20 dB). Specifically for beaked whales, behavioral response research discussed below and in Section 3.8.3.1.1.5 (Behavioral Reactions) has demonstrated that beaked whales are sensitive to sound from sonars and usually avoid sound sources by 10 or more kilometers. These are well beyond the ranges to TTS for mid-frequency cetaceans such as beaked whales. Therefore, any TTS predicted by the quantitative analysis is unlikely to occur in beaked whales.

Research and observations (Section 3.8.3.1.1.5, Behavioral Reactions) show that if beaked whales are exposed to sonar or other transducers they may startle, break off feeding dives, and avoid the area of the sound source at levels ranging between 95 and 157 dB re 1  $\mu$ Pa (McCarthy et al., 2011). Furthermore, in research done at the Navy's fixed tracking range in the Bahamas and Hawaii, animals leave the immediate area of the anti-submarine warfare training exercise but return within a few days after the event ends (Henderson et al., 2015; Henderson et al., 2016; Manzano-Roth et al., 2016; Tyack et al., 2011). Populations of beaked whales and other odontocetes on Navy fixed ranges that have been operating for decades appear to be stable, and analysis is ongoing. Significant behavioral reactions seem likely in most cases if beaked whales are exposed to anti-submarine sonar within a few tens of kilometers, especially for prolonged periods (a few hours or more), since this is one of the most sensitive marine mammal groups to human-made sound of any species or group studied to date.

Based on the best available science, the Navy believes beaked whales that exhibit a significant behavioral reaction due to sonar and other transducers during training activities would generally not

have long-term consequences for individuals or populations. However, because of a lack of scientific consensus regarding the causal link between sonar and stranding events, NMFS has stated in a letter to the Navy dated October 2006 that it “cannot conclude with certainty the degree to which mitigation measures would eliminate or reduce the potential for serious injury or mortality.” The Navy does not anticipate that marine mammal strandings or mortality will result from the operation of sonar during Navy exercises within the TMAA. Additionally, through the MMPA process (which allows for adaptive management), NMFS and the Navy will determine the appropriate way to proceed in the event that a causal relationship were to be found between Navy activities and a future stranding.

As described for odontocetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of Baird’s, Cuvier’s, and Stejneger’s beaked whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-21: Estimated Impacts on Individual Baird’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska	106	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Table 3.8-22: Estimated Impacts on Individual Cuvier’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska	429	3	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Table 3.8-23: Estimated Impacts on Individual Stejneger’s Beaked Whale Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Alaska	467	15	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Pinnipeds and Mustelids**

Pinnipeds include phocid seals (true seals) and otariids (sea lions and fur seals), and mustelids include sea otters.

Pinnipeds may be exposed to sound from sonar and other transducers associated with training activities throughout the year. Low- (less than 1 kHz), mid- (1–10 kHz), and high-frequency (10–100 kHz) sonars produce sounds that are likely to be within the audible range of pinnipeds (see Section 3.8.2.1.4, Hearing and Vocalization). Comparatively, hearing sensitivities are significantly reduced in mustelids and exposure to these sounds may have lower overall severity. If a sound is within an animal’s hearing range then behavioral reactions, physiological stress, masking and hearing loss are potential impacts that must be analyzed. If a marine mammal cannot hear a sound, then behavioral reactions, physiological stress, masking, or hearing loss could not occur. Impact ranges for pinnipeds and mustelids are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers).

There is no research on the effects of sonar on sea otters. As described in Section 3.8.3.1.1.5 (Behavioral Reactions), mustelids have similar or reduced hearing capabilities compared to pinnipeds (specifically otariids). Thus, it is reasonable to assume that mustelids use their hearing similarly to that of otariids, and the types of impacts from exposure to sonar and other transducers may also be similar to those described below for pinnipeds, including behavioral reactions, physiological stress, masking, and hearing loss; however, because mustelids spend the majority of their time with their heads above or at the water’s surface and live near shore, they are less likely to be exposed to or impacted by sonars and other transducers used in training activities.

A few behavioral reactions by pinnipeds resulting from exposure to sonar could take place at distances of up to 10 km. Behavioral reactions, however, are much more likely within a kilometer or less of the sound source (see Section 3.8.3.1.1.5, Behavioral Reactions). As discussed above in *Assessing the Severity of Behavioral Responses from Sonar Under Military Readiness*, the quantitative analysis very likely overestimated the numbers of behavioral reactions due to the underlying nature of the data used to derive the behavioral response functions. Research shows that pinnipeds in the water are generally tolerant of human-made sound and activity, while mustelids have reduced underwater hearing abilities (see Section 3.8.3.1.1.5, Behavioral Reactions). If pinnipeds or mustelids are exposed to sonar or other transducers, they may react in various ways, depending on their experience with the sound source and what activity they are engaged in at the time of the acoustic exposure. Pinnipeds or mustelids may not react at all until the sound source is approaching within a few hundred meters and then may alert, ignore the stimulus, change their behaviors, or avoid the immediate area by swimming away or diving. Significant behavioral reactions would not be expected in most cases, and long-term consequences for

individual pinnipeds or mustelids from a single or several impacts per year are unlikely. Behavioral research indicates that most pinnipeds probably avoid sound sources at levels that could cause higher levels of TTS (greater than 20 dB of TTS) and PTS. Recovery from TTS begins almost immediately after the noise exposure ceases and can take a few minutes to a few days to fully recover, depending on the magnitude of the initial threshold shift. Most TTS, if it does actually occur, would be more likely to be minor to moderate (i.e., less than 20 dB of TTS directly after the exposure) and would recover within a matter of minutes to hours. Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. During the short period that a pinniped had TTS, social calls from conspecifics could be more difficult to detect or interpret. Killer whales are a primary predator of pinnipeds. Some TTS could make killer whale calls more difficult to detect at further ranges until hearing recovers. Pinnipeds probably use sound and vibrations to find and capture prey underwater. Therefore, it could be more difficult for pinnipeds with TTS to locate food for a short period before their hearing recovers. Because TTS would likely be minor to moderate (less than 20 dB of TTS), costs would be short term and could be recovered. A single or even a few mild to moderate TTS per year are unlikely to have any long-term consequences for that individual.

Research and observations of masking in marine mammals are discussed in Section 3.8.3.1.1.4 (Masking). Many low- (less than 1 kHz), mid- (1–10 kHz), and high-frequency (10–100 kHz) sonars produce sounds that are likely to be within the hearing range of pinnipeds and potentially mustelids. Most anti-submarine warfare sonar use low- and mid-frequency sonar signals (i.e., sounds) which are limited in the temporal, frequency, and spatial domains. The duration of most individual sounds is short, lasting up to a few seconds each. Some systems operate with higher duty cycles or nearly continuously, but typically use lower power and have a narrow frequency band (typically less than one-third octave). These factors reduce the likelihood of sources causing significant masking in pinnipeds due to exposure to sonar used during anti-submarine warfare activities. Pinnipeds and mustelids may experience some limited masking at closer ranges from high-frequency sonars and other transducers; however, the frequency band of the sonar is narrow, limiting the likelihood of masking. Sonars that employ high frequencies are typically used for mine hunting, navigation, and object detection (avoidance). Potential costs to pinnipeds and mustelids from masking are similar to those discussed above for mild to moderate levels of TTS, with the primary difference being that the effects of masking are only present when the sound source (i.e., sonar) is actively transmitting and the effect is over the moment the sound has ceased. Nevertheless, pinnipeds that do experience some masking for a short period from sonar or other transducers may have their ability to communicate with conspecifics reduced, especially at further ranges. Sounds from mid-frequency sonar could mask killer whale vocalizations making them more difficult to detect, especially at further ranges. Pinnipeds probably use sound and vibrations to find and capture prey underwater. Therefore, it could be more difficult for pinnipeds to locate food if masking is occurring. A single or even a few short periods of masking, if it were to occur, to an individual pinniped or mustelid per year are unlikely to have any long-term consequences for that individual.

**Steller Sea Lions (one DPS is Endangered Species Act-Listed)**

Steller sea lions may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the Steller sea lions' occurrence in the TMAA would be likely year round. Impacts have been modeled for the Eastern U.S. stock of Steller sea lions, which are not ESA-listed, and for the Western U.S. stock of Steller sea lions, which are ESA-listed.

The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of Steller sea lions.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed Steller sea lions. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

### **California Sea Lions**

California sea lions may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the California sea lions' occurrence in the TMAA would be seasonal and are most likely to be present April through May. The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of California sea lions.*

### **Northern Fur Seals**

Northern fur seals may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although northern fur seals are most likely to be present in the TMAA December through July, males may potentially be present year round. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-24). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to multiple stocks (Table 3.8-24).

As described above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of northern fur seals incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-24: Estimated Impacts on Individual Northern Fur Seal Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
Eastern Pacific	2,836	25	0
California	58	1	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Northern Elephant Seals**

Northern elephant seals may be exposed to sounds from sonar and other transducers associated with training activities April through October. The Navy has determined the northern elephant seals' occurrence in the TMAA would be seasonal and are most likely to be present March through October. The quantitative analysis estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-25). Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Estimated impacts apply to the California stock (Table 3.8-25).

As described above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. This minor consequence for an individual is unlikely to have any long-term consequences for the species or stock. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would result in the unintentional taking of northern elephant seals incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-25: Estimated Impacts on Individual Northern Elephant Seal Stocks Within the Study Area per Year from Sonar and Other Transducers Used During Training Under Alternative 1**

Estimated Impacts by Effect			
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>
California	898	1,634	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Harbor Seals**

Harbor seals may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although harbor seals' occurrence in the TMAA is year round, they are rarely found more than 20 km from shore and are therefore more likely to be present in the inshore water locations of the GOA, versus being found beyond the slope or further offshore within the TMAA. The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are

discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of harbor seals.*

#### **Ribbon Seals**

Ribbon seals may be exposed to sounds from sonar and other transducers associated with training activities April through October. Although ribbon seals are considered rare in the TMAA, their occurrence is year round, and they are most likely to be present in the TMAA July through September. The quantitative analysis estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of harbor seals.*

#### **Northern Sea Otters (one DPS is Endangered Species Act-Listed)**

Northern sea otters are unlikely to be exposed to sounds from sonar and other transducers associated with training activities April through October. Although northern sea otters occur in the nearshore margins of the Gulf of Alaska year round, they would rarely be present in the TMAA since the normal range and habitat of sea otters is well inland of the TMAA boundaries. Sea otters seldom range more than 2 km from shore, and in this region they are mainly concentrated within 400 m from shore because they are benthic foragers. (Bodkin, 2015) notes that sea otters can be found many kilometers from shore in locations where there are shoals far from land, but there are no known offshore populations near the TMAA. Individuals from the Southwest Alaska stock (ESA-listed) are not expected to be present in the TMAA. It is possible that vagrant individuals from the Southcentral Alaska stock or the Southeast Alaska stock of sea otters (neither are ESA-listed) could potentially occur in the nearshore margins of the TMAA. Some individuals, particularly juvenile males, may travel farther offshore (Calambokidis et al., 1987; Laidre et al., 2009; Muto et al., 2017; Riedman & Estes, 1990).

Ghoul and Reichmuth (2014b) have shown that sea otters are not especially well adapted for hearing underwater, which suggests that the function of this sense has been less important in their survival and evolution than in comparison to pinnipeds. Due to their low sensitivity to underwater sounds, their preferred habitat, behavioral pattern of spending a majority of their time above water, and the short range to effects for phocids as described in Section 3.8.3.1.2.2 (Impact Ranges for Sonar and Other Transducers), impacts to northern sea otters from Navy training activities involving sonar and other transducers are highly unlikely to occur. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.



*Pursuant to the MMPA, the use of sonar and other transducers during training activities as described under Alternative 1 would not result in the incidental taking of Northern sea otters.*

*Pursuant to the ESA, the use of sonar and other transducers during training activities as described under Alternative 1 may affect ESA-listed Northern sea otters. The Navy will consult with USFWS as required by Section 7(a)(2) of the ESA.*

### **3.8.3.1.3 Impacts from Vessel Noise**

#### **3.8.3.1.3.1 Impacts from Vessel Noise Under the No Action Alternative**

Under the No Action Alternative, proposed Navy training activities would not occur within the TMAA. The impacts associated with Navy training activities would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing Navy training activities.

#### **3.8.3.1.3.2 Impacts from Vessel Noise Under Alternative 1**

Training activities within the TMAA involve maneuvers by various types of surface ships, boats, and submarines (collectively referred to as vessels). Marine mammals may be exposed to noise from vessel movement throughout the TMAA. A detailed description of the acoustic characteristics and typical sound levels of vessel noise are in Section 3.0.4.1.2 (Vessel Noise). Proposed training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Section 3.8.3.1.1 (Background) summarizes and synthesizes available information on behavioral reactions, masking, and physiological stress due to noise exposure, including vessel noise (Sections 3.8.3.1.1.2, Hearing Loss; 3.8.3.1.1.3, Physiological Stress; 3.8.3.1.1.4, Masking; and 3.8.3.1.1.5, Behavioral Reactions).

Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Because the existing baseline conditions have not changed appreciably, and no new Navy training activities are being proposed for use in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to marine mammals is not warranted. The Navy will implement mitigation measures for vessel movement to avoid the potential for marine mammal vessel strikes, as discussed in Section 5.3.4.1 (Vessel Movement). The mitigation for vessel movement (i.e., maneuvering to maintain a specified distance from a marine mammal) will also help the Navy avoid or reduce potential impacts from vessel noise on marine mammals.

Sound from naval vessels would overlap proposed critical habitat for the ESA-listed Western North Pacific and Mexico DPSs of humpback whales. As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), one essential feature was identified for humpback whale critical habitat: prey species, primarily euphausiids and small pelagic schooling fishes, of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth. Although vessel noise may elicit a brief response from individual prey species in close proximity to a vessel, noise from naval vessels presents no plausible route of impact to prey species of sufficient quantity, abundance, and accessibility.

*Pursuant to the MMPA, sound produced by vessel movement during training activities as described under Alternative 1 would not result in the incidental taking of marine mammals.*

*Pursuant to the ESA, sound produced by vessel movement during training activities as described under Alternative 1 may affect ESA-listed marine mammals. The Navy will consult with NMFS as required by section 7(a)(2) of the ESA. Vessel noise during training activities would have no effect on the proposed critical habitat for humpback whales.*

*Pursuant to the ESA, vessel noise during training activities as described under Alternative 1 may affect Northern sea otters. The Navy will consult with USFWS as required by Section 7(a)(2) of the ESA.*

#### **3.8.3.1.4 Impacts from Aircraft Noise**

##### **3.8.3.1.4.1 Impacts from Aircraft Noise Under the No Action Alternative**

Under the No Action Alternative, proposed Navy training activities would not occur within the TMAA. The impacts associated with Navy training activities would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing Navy training activities.

##### **3.8.3.1.4.2 Impacts from Aircraft Noise Under Alternative 1**

Many ongoing and proposed training activities within the TMAA involve maneuvers by various types of fixed, rotary-wing, and tilt-rotor aircraft (collectively referred to as aircraft). Most aircraft noise would be concentrated around airbases and fixed ranges within the range complex, especially in the waters immediately surrounding aircraft carriers at sea during takeoff and landing. Other aircraft overflights include commercial air traffic in addition to U.S. Navy aircraft. Aircraft produce extensive airborne noise from either turbofan or turbojet engines. An infrequent type of aircraft noise is the sonic boom, produced when the aircraft exceeds the speed of sound. Rotary-wing aircraft produce low-frequency sound and vibration (Pepper et al., 2003). Section 3.8.3.1.1 (Background) summarizes and synthesizes available information on behavioral reactions, masking, and physiological stress due to noise exposure, including aircraft noise (Sections 3.8.3.1.1.2, Hearing Loss; 3.8.3.1.1.3, Physiological Stress; 3.8.3.1.1.4, Masking; and 3.8.3.1.1.5, Behavioral Reactions).

Marine mammals may be exposed to aircraft-generated noise throughout the TMAA. A detailed description of aircraft noise as a stressor is in Section 3.0.4.1.3 (Aircraft Noise) of this document and the 2011 GOA Final EIS/OEIS. Proposed training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Because the existing baseline conditions have not changed appreciably, and no new Navy training activities are being proposed for use in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to marine mammals is not warranted.

Sound from naval aircraft would overlap proposed critical habitat for the ESA-listed Western North Pacific and Mexico DPSs of humpback whales. As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), one essential feature was identified for humpback whale critical habitat: prey species, primarily euphausiids and small pelagic schooling fishes, of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth. Although aircraft noise may elicit a brief response from individual prey species in close proximity to a

low-flying aircraft, noise from aircraft presents no plausible route of impact to prey species of sufficient quantity, abundance, and accessibility.

*Pursuant to the MMPA, aircraft noise during training activities as described under Alternative 1 would not result in the incidental taking of marine mammals.*

*Pursuant to the ESA, aircraft noise during training activities as described under Alternative 1 may affect ESA-listed marine mammals. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA. Aircraft noise during training activities would have no effect on the proposed critical habitat for humpback whales.*

*Pursuant to the ESA, aircraft noise during training activities as described under Alternative 1 may affect Northern sea otters. The Navy will consult with USFWS as required by Section 7(a)(2) of the ESA.*

### **3.8.3.1.5 Impacts from Weapon Noise**

#### **3.8.3.1.5.1 Impacts from Weapon Noise Under the No Action Alternative**

Under the No Action Alternative, proposed Navy training activities would not occur within the TMAA. The impacts associated with Navy training activities would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing Navy training activities.

#### **3.8.3.1.5.2 Impacts from Weapon Noise Under Alternative 1**

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, which are described in Section 3.0.4.1.4 (Weapon Noise). In general, these are impulsive sounds generated in close vicinity to or at the water surface, with the exception of items that are launched underwater. The firing of a weapon may have several components of associated noise. Firing of guns could include sound generated in air by firing a gun (muzzle blast) and a crack sound due to a low amplitude shock wave generated by a supersonic projectile flying through the air. Most in-air sound would be reflected at the air-water interface.

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. Vibration from the blast propagating through a ship's hull, the sound generated by the impact of an object with the water surface, and the sound generated by launching an object underwater are other sources of impulsive sound in the water. 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.

Section 3.8.3.1.1 (Background) summarizes and synthesizes available information on behavioral reactions, masking, and physiological stress due to impulsive noise exposure (Sections 3.8.3.1.1.2, Hearing Loss; 3.8.3.1.1.3, Physiological Stress; 3.8.3.1.1.4, Masking; and 3.8.3.1.1.5, Behavioral Reactions).

Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Because the existing baseline conditions have not changed appreciably, and no new Navy training activities are proposed in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to marine mammals is not warranted. The Navy will implement mitigation measures to avoid or reduce potential impacts from weapon noise during large-caliber gunnery activities, as discussed in Section 5.3.2.2 (Weapon Firing Noise).

Weapon noise would overlap proposed critical habitat for the ESA-listed Western North Pacific and Mexico DPSs of humpback whales in the Offshore Area, although implementation of the North Pacific Right Whale and Portlock Bank Mitigation Areas would limit any potential overlap of weapon noise with the proposed critical habitat in the Offshore Area during training, as described in Chapter 5 (Mitigation). As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), one essential feature was identified for humpback whale critical habitat: prey species, primarily euphausiids and small pelagic schooling fishes, of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth. Weapon noise would not remove humpback prey items or reduce the quality of prey in terms of nutritional content.

*Pursuant to the MMPA, weapon noise during training activities as described under Alternative 1 would not result in the incidental taking of marine mammals.*

*Pursuant to the ESA, weapon noise during training activities as described under Alternative 1 may affect ESA-listed marine mammals. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA. Weapon noise during training activities would have no effect on the proposed critical habitat for humpback whales.*

*Pursuant to the ESA, weapon noise during training activities as described under Alternative 1 may affect Northern sea otters. The Navy will consult with USFWS as required by Section 7(a)(2) of the ESA.*

### **3.8.3.2 Explosive Stressors**

Assessing whether an explosive detonation may disturb or injure a marine mammal involves understanding the characteristics of the explosive sources, the marine mammals that may be present near the sources, the physiological effects of a close explosive exposure, and the effects of impulsive sound on marine mammal hearing and behavior. Many other factors besides just the received level or pressure wave of an explosion such as the animal's physical condition and size, prior experience with the explosive sound, and proximity to the explosion may influence physiological effects and behavioral reactions.

The ways in which an explosive exposure could result in immediate effects or lead to long-term consequences for an animal are explained in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). The following Background section discusses what is currently known about explosive effects to marine mammals.

Due to new acoustic impact criteria, marine mammal densities, and revisions to the Navy Acoustic Effects Model, the analysis provided in Section 3.8.3.2.2 (Impacts from Explosives) of this SEIS/OEIS supplants the 2016 GOA Final SEIS/OEIS for marine mammals and changes estimated impacts for some species since the 2016 GOA Final SEIS/OEIS.

#### **3.8.3.2.1 Background**

##### **3.8.3.2.1.1 Injury**

Injury refers to the direct effects on the tissues or organs of an animal due to exposure to pressure waves. Injury in marine mammals can be caused directly by exposure to explosions. Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on injury and the framework used to analyze this potential impact.

### **Injury due to Explosives**

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 (Greaves et al., 1943; Office of the Surgeon General, 1991; Richmond et al., 1973). The near instantaneous high magnitude pressure change near an explosion can injure an animal where tissue material properties significantly differ from the surrounding environment, such as around air-filled cavities such as in the lungs or gastrointestinal tract. Large pressure changes at tissue-air interfaces in the lungs and gastrointestinal tract may cause tissue rupture, resulting in a range of injuries depending on degree of exposure. The lungs are typically the first site to show any damage, while the solid organs (e.g., liver, spleen, and kidney) are more resistant to blast injury (Clark & Ward, 1943). Recoverable injuries would include slight lung injury, such as capillary interstitial bleeding, and contusions to the gastrointestinal tract. More severe injuries, such as tissue lacerations, major hemorrhage, organ rupture, or air in the chest cavity (pneumothorax), would significantly reduce fitness and likely cause death in the wild. Rupture of the lung may also introduce air into the vascular system, producing air emboli that can cause a stroke or heart attack by restricting oxygen delivery to critical organs.

If an animal is exposed to an explosive blast underwater, the likelihood of injury depends on the charge size, the geometry of the exposure (distance to the charge, depth of the animal and the charge), and the size of the animal. In general, an animal would be less susceptible to injury near the water surface because the pressure wave reflected from the water surface would interfere with the direct path pressure wave, reducing positive pressure exposure. Susceptibility would increase with depth, until normal lung collapse (due to increasing hydrostatic pressure) and increasing ambient pressures again reduce susceptibility. See Appendix B (Acoustic and Explosives Concepts) for an overview of explosive propagation and an explanation of explosive effects on gas cavities.

The only known occurrence of mortality or injury to a marine mammal due to a Navy training event involving explosives occurred in March 2011 in nearshore waters off San Diego, California, at the Silver Strand Training Complex. This area had been used for underwater demolitions training for at least three decades without prior known incident. On this occasion, however, a group of approximately 100–150 long-beaked common dolphins entered the mitigation zone surrounding an area where a time-delayed firing device had been initiated on an explosive with a net explosive weight of 8.76 pounds (lb.) (3.97 kilograms [kg]) placed at a depth of 48 ft. (14.6 m). Approximately one minute after detonation, three animals were observed dead at the surface. The Navy recovered those animals and transferred them to the local stranding network for necropsy. A fourth animal was discovered stranded and dead 42 NM to the north of the detonation three days later. It is unknown exactly how close those four animals were to the detonation. Upon necropsy, all four animals were found to have sustained typical mammalian primary blast injuries (Danil & St Leger, 2011). In the TMAA, there is no known occurrence of mortality or injury to marine mammals due to Navy training events involving explosives.

Relatively little is known about auditory system trauma in marine mammals resulting from explosive exposure, although it is assumed that auditory structures would be vulnerable to blast injuries. Auditory trauma was found in two humpback whales that died following the detonation of a 5,000 kg explosive used off Newfoundland during demolition of an offshore oil rig platform (Ketten et al., 1993), but the proximity of the whales to the detonation was unknown. Eardrum rupture was examined in submerged terrestrial mammals exposed to underwater explosions (Richmond et al., 1973; Yelverton et al., 1973); however, results may not be applicable to the anatomical adaptations for underwater hearing in marine

mammals. In this discussion, primary blast injury to auditory tissues is considered gross structural tissue damage distinct from threshold shift or other auditory effects (see Section 3.8.3.2.1.2, Hearing Loss).

Controlled tests with a variety of lab animals (mice, rats, dogs, pigs, sheep, and other species) are the best data sources on actual injury to mammals due to underwater exposure to explosions. In the early 1970s, the Lovelace Foundation for Medical Education and Research conducted a series of tests in an artificial pond at Kirtland Air Force Base, New Mexico, to determine the effects of underwater explosions on mammals, with the goal of determining safe ranges for human divers. The resulting data were summarized in two reports (Richmond et al., 1973; Yelverton et al., 1973). Specific physiological observations for each test animal are documented in Richmond et al. (1973). Gas-containing internal organs, such as lungs and intestines, were the principle damage sites in submerged terrestrial mammals; this is consistent with earlier studies of mammal exposures to underwater explosions in which lungs were consistently the first areas to show damage, with less consistent damage observed in the gastrointestinal tract (Clark & Ward, 1943; Greaves et al., 1943). Results from all of these tests suggest two explosive metrics are predictive of explosive injury: peak pressure and impulse.

### **Impulse as a Predictor of Explosive Injury**

In the Lovelace studies, acoustic impulse was found to be the metric most related to degree of injury, and size of an animal's gas-containing cavities was thought to play a role in blast injury susceptibility. The lungs of most marine mammals are similar in proportion to overall body size as those of terrestrial mammals, so the magnitude of lung damage in the tests may approximate the magnitude of injury to marine mammals when scaled for body size. Within the marine mammals, mysticetes and deeper divers (e.g., Kogiidae, Physteridae, Ziphiidae) tend to have lung to body size ratios that are smaller and more similar to terrestrial animal ratios than the shallow diving odontocetes (e.g., Phocoenidae, Delphinidae) and pinnipeds (Fahlman et al., 2014a; Piscitelli et al., 2010). The use of test data with smaller lung-to-body ratios to set injury thresholds may result in a more conservative estimate of potential for damaging effects (i.e., lower thresholds) for animals with larger lung-to-body ratios.

For these shallow exposures of small terrestrial mammals (masses ranging from 3.4 to 50 kg) to underwater detonations, Richmond et al. (1973) reported that no blast injuries were observed when exposures were less than 6 pounds per square inch per millisecond (psi-ms) (40 pascal seconds [Pa-s]), no instances of slight lung hemorrhage occurred below 20 psi-ms (140 Pa-s), and instances of no lung damage were observed in some exposures at higher levels up to 40 psi-ms (280 Pa-s). An impulse of 34 psi-ms (230 Pa-s) resulted in about 50 percent incidence of slight lung hemorrhage. About half of the animals had gastrointestinal tract contusions (with slight ulceration, i.e., some perforation of the mucosal layer) at exposures of 25–27 psi-ms (170–190 Pa-s). Lung injuries were found to be slightly more prevalent than gastrointestinal tract injuries for the same exposure.

The Lovelace subject animals were exposed near the water surface; therefore, depth effects were not discernible in this data set. In addition, this data set included only small terrestrial animals, whereas marine mammals may be several orders of magnitude larger and have respiratory structures adapted for the high pressures experienced at depth. The anatomical differences between the terrestrial animals used in the Lovelace tests and marine mammals are summarized in Fetherston (2019). Goertner (1982) examined how lung cavity size would affect susceptibility to blast injury by considering both marine mammal size and depth in a bubble oscillation model of the lung; however, the Goertner (1982) model did not consider how tissues surrounding the respiratory air spaces would reflect shock wave energy or constrain oscillation (Fetherston et al., 2019). Animal depth relates to injury susceptibility in two ways: injury is related to the relative increase in explosive pressure over hydrostatic pressure, and lung

collapse with depth reduces the potential for air cavity oscillatory damage. The period over which an impulse must be delivered to cause damage is assumed to be related to the natural oscillation period of an animal's lung, which depends on lung size.

Because gas-containing organs are more vulnerable to primary blast injury, adaptations for diving that allow for collapse of lung tissues with depth may make animals less vulnerable to lung injury with depth. Adaptations for diving include a flexible thoracic cavity, distensible veins that can fill space as air compresses, elastic lung tissue, and resilient tracheas with interlocking cartilaginous rings that provide strength and flexibility (Ridgway, 1972). Denk et al. (2020) found intra-species differences in the compliance of tracheobronchial structures of post-mortem cetaceans and pinnipeds under diving hydrostatic pressures, which would affect depth of alveolar collapse. Older literature suggested complete lung collapse depths at approximately 70 m for dolphins (Ridgway & Howard, 1979) and 20–50 m for phocid seals (Falke et al., 1985; Kooyman et al., 1972). Follow-on work by Kooyman and Sinnott (1982), in which pulmonary shunting was studied in harbor seals and sea lions, suggested that complete lung collapse for these species would be about 170 m and about 180 m, respectively. More recently, evidence in sea lions suggests that complete collapse might not occur until depths as great as 225 m; although the depth of collapse and depth of the dive are related, sea lions can affect the depth of lung collapse by varying the amount of air inhaled on a dive (McDonald & Ponganis, 2012). This is an important consideration for all divers who can modulate lung volume and gas exchange prior to diving via the degree of inhalation and during diving via exhalation (Fahlman et al., 2009); indeed, there are noted differences in pre-dive respiratory behavior, with some marine mammals exhibiting pre-dive exhalation to reduce the lung volume (e.g., phocid seals (Kooyman et al., 1973)).

#### **Peak Pressure as a Predictor of Explosive Injury**

High instantaneous peak pressures can cause damaging tissue distortion. Goertner (1982) suggested a peak overpressure gastrointestinal tract injury criterion because the size of gas bubbles in the gastrointestinal tract are variable, and their oscillation period could be short relative to primary blast wave exposure duration. The potential for gastrointestinal tract injury, therefore, may not be adequately modeled by the single oscillation bubble methodology used to estimate lung injury due to impulse. Like impulse, however, high instantaneous pressures may damage many parts of the body, but damage to the gastrointestinal tract is used as an indicator of any peak pressure-induced injury due to its vulnerability.

Older military reports documenting exposure of human divers to blast exposure generally describe peak pressure exposures around 100 pounds per square inch (psi) (237 dB re 1  $\mu$ Pa peak) to feel like slight pressure or stinging sensation on skin, with no enduring effects (Christian & Gaspin, 1974). Around 200 psi, the shock wave felt like a blow to the head and chest. Data from the Lovelace Foundation experiments show instances of gastrointestinal tract contusions after exposures up to 1,147 psi peak pressure, while exposures of up to 588 psi peak pressure resulted in many instances of no observed gastrointestinal tract effects. The lowest exposure for which slight contusions to the gastrointestinal tract were reported was 237 dB re 1  $\mu$ Pa peak. As a vulnerable gas-containing organ, the gastrointestinal tract is vulnerable to both high peak pressure and high impulse, which may vary to differing extents due to blast exposure conditions (i.e., animal depth, distance from the charge). This likely explains the range of effects seen at similar peak pressure exposure levels and shows the utility of considering both peak pressure and impulse when analyzing the potential for injury due to explosives.

#### **3.8.3.2.1.2 Hearing Loss**

Exposure to intense sound may result in noise-induced hearing loss that persists after cessation of the noise exposure. Hearing loss may be temporary or permanent, depending on factors such as the exposure frequency, received SPL, temporal pattern, and duration. The frequencies affected by hearing loss may vary depending on the exposure frequency, with frequencies at and above the exposure frequency most strongly affected. The amount of hearing loss may range from slight to profound, depending on the ability of the individual to hear at the affected frequencies. Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on hearing loss and the framework used to analyze this potential impact.

Hearing loss has only been studied in a few species of marine mammals, although hearing studies with terrestrial mammals are also informative. There are no direct measurements of hearing loss in marine mammals due to exposure to explosive sources. The sound resulting from an explosive detonation is considered an impulsive sound and shares important qualities (i.e., short duration and fast rise time) with other impulsive sounds such as those produced by air guns. General research findings regarding TTS and PTS in marine mammals as well as findings specific to exposure to other impulsive sound sources are discussed in Section 3.8.3.1.1.2 (Hearing Loss) and Section 3.8.3.1.1.1 (Injury) under Acoustic Stressors above.

#### **3.8.3.2.1.3 Physiological Stress**

Marine mammals naturally experience stress within their environment and as part of their life histories. The stress response is a suite of physiological changes that are meant to help an organism mitigate the impact of a stressor. However, if the magnitude and duration of the stress response is too great or too long, then it can have negative consequences to the organism (e.g., decreased immune function, decreased reproduction). Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on physiological stress and the framework used to analyze this potential impact.

There are no direct measurements of physiological stress in marine mammals due to exposure to explosive sources. General research findings regarding physiological stress in marine mammals due to exposure to sound and other stressors are discussed in detail in Section 3.8.3.1.1.3 (Physiological Stress) under Acoustic Stressors above. Because there are many unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, it is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

#### **3.8.3.2.1.4 Masking**

Masking occurs when one sound, distinguished as the “noise,” interferes with the detection, discrimination, or recognition of another sound. The quantitative definition of masking is the amount in decibels an auditory detection, discrimination, or recognition threshold is raised in the presence of a masker (Erbe et al., 2016). As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), 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 (with the potential exceptions of reverberations from impulsive noise). Masking can lead to vocal changes, such as the Lombard effect (increasing amplitude) or other noise-induced vocal modifications, such as changing frequency (Hotchkin & Parks, 2013); and behavioral changes (e.g., cessation of



foraging, leaving an area) to both signalers and receivers, in an attempt to compensate for noise levels (Erbe et al., 2016).

There are no direct observations of masking in marine mammals due to exposure to explosive sources. General research findings regarding masking in marine mammals due to exposure to sound and other stressors are discussed in detail in Section 3.8.3.1.1.4 (Masking) under Acoustic Stressors above. Potential masking from explosive sounds is likely to be similar to masking studied for other impulsive sounds such as air guns.

#### **3.8.3.2.1.5 Behavioral Reactions**

As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), any stimuli in the environment can cause a behavioral response in marine mammals, including noise from explosions. There are few direct observations of behavioral reactions from marine mammals due to exposure to explosive sounds. Lammers et al. (2017) recorded dolphin detections near naval mine neutralization exercises and found that although the immediate response (within 30 seconds of the explosion) was an increase in whistles relative to the 30 seconds before the explosion, there was a reduction in daytime acoustic activity during the day of and the day after the exercise within 6 km. However, the nighttime activity did not seem to be different than that prior to the exercise, and two days after there appeared to be an increase in daytime acoustic activity, indicating a rapid return to the area by the dolphins (Lammers et al., 2017). Vallejo et al. (2017) report on boat-based line-transect surveys which were run over 10 years in an area where an offshore wind farm was built; these surveys included the periods of preconstruction, construction, and postconstruction. Harbor porpoise were observed throughout the area during all three phases, but were not detected within the footprint of the windfarm during the construction phase, and were overall less frequent throughout the study area. However, they returned after the construction was completed at a slightly higher level than in the preconstruction phase. Furthermore, there was no large-scale displacement of harbor porpoises during construction, and in fact their avoidance behavior only occurred out to about 18 km, in contrast to the approximately 25 km avoidance distance found in other windfarm construction and pile driving monitoring efforts.

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. However, 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 (Hastie et al., 2019; Martin et al., 2020). Behavioral reactions from explosive sounds are likely to be similar to reactions studied for other impulsive sounds, such as those produced by air guns and impact pile driving. Data on behavioral responses to impulsive sound sources are limited across all marine mammal groups, with only a few studies available for mysticetes and odontocetes. Most data have come from seismic surveys that occur over long durations (e.g., on the order of days to weeks), and typically utilize large multi-air gun arrays that fire repeatedly. While seismic data provide the best available science for assessing behavioral responses to impulsive sounds by marine mammals, it is likely that these responses represent a worst-case scenario compared to responses to explosives used in Navy activities, which would typically consist of single impulses or a cluster of impulses, rather than long-duration, repeated impulses.

See Section 3.8.3.1.1.5 (Behavioral Reactions) under Section 3.8.3.1 (Acoustic Stressors) for a summary of information on marine mammal reactions to impulsive sounds.

#### **3.8.3.2.1.6 Stranding**

When a marine mammal (alive or dead) swims or floats onto shore and becomes beached or incapable of returning to sea, the event is termed a “stranding” (Geraci et al., 1999; Geraci & Lounsbury, 2005; Perrin & Geraci, 2002). Specifically, under U.S. law, a stranding is an event in the wild where: “(A) a marine mammal is dead and is (i) on a beach or shore of the United States; or (ii) in waters under the jurisdiction of the United States (including any navigable waters); or (B) a marine mammal is alive and is (i) on a beach or shore of the United States and is unable to return to the water; (ii) on a beach or shore of the United States and, although able to return to the water, is in need of medical attention; or (iii) in the waters under the jurisdiction of the United States (including any navigable waters), but is unable to return to its natural habitat under its own power or without assistance” (16 U.S.C. section 1421h).

Impulsive sources (e.g., explosions) also have the potential to contribute to strandings, but such occurrences are even less common than those that have been related to certain sonar activities. During a Navy training event on March 4, 2011, at the Silver Strand Training Complex in San Diego, California, three long-beaked common dolphins were killed by an underwater detonation. Further details are provided above. Discussions of mitigation measures associated with these and other training events are presented in Chapter 5 (Mitigation).

#### **3.8.3.2.1.7 Long-Term Consequences**

Long-term consequences to a population are determined by examining changes in the population growth rate. For additional information on the determination of long-term consequences, see Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). Physical effects from explosive sources that could lead to a reduction in the population growth rate include mortality or injury, which could remove animals from the reproductive pool, and permanent hearing impairment or chronic masking, which could impact navigation, foraging, predator avoidance, or communication. The long-term consequences due to individual behavioral reactions, masking and short-term instances of physiological stress are especially difficult to predict because individual experience over time can create complex contingencies, especially for long-lived animals like marine mammals. For example, a lost reproductive opportunity could be a measurable cost to the individual; however, short-term costs may be recouped during the life of an otherwise healthy individual. These factors are taken into consideration when assessing risk of long-term consequences.

#### **3.8.3.2.2 Impacts from Explosives**

Marine mammals could be exposed to energy, sound, and fragments from explosions in the water and near the water surface associated with the proposed activities. Energy from an explosion is capable of causing mortality, injury, hearing loss, a behavioral response, masking, or physiological stress, depending on the level and duration of exposure.

The death of an animal would eliminate future reproductive potential, which is considered in the analysis of potential long-term consequences to the population. Exposures that result in non-auditory injuries or PTS may limit an animal’s ability to find food, communicate with other animals, or interpret the surrounding environment. Impairment of these abilities can decrease an individual’s chance of survival or impact its ability to successfully reproduce. TTS can also impair an animal’s abilities, but the individual is likely to recover quickly with little significant effect.

Explosions in the ocean or near the water surface can introduce loud, impulsive, broadband sounds into the marine environment. These sounds, which are within the audible range of most marine mammals, could cause behavioral reactions, masking, and elevated physiological stress. Behavioral responses can

include shorter surfacings, shorter dives, fewer blows (breaths) per surfacing, longer intervals between blows, ceasing or increasing vocalizations, shortening or lengthening vocalizations, and changing frequency or intensity of vocalizations (National Research Council 2005). Sounds from explosives could also mask biologically important sounds; however, the duration of individual sounds is very short, reducing the likelihood of substantial auditory masking.

#### **3.8.3.2.2.1 Methods for Analyzing Impacts from Explosives**

The Navy performed a quantitative analysis to estimate the number of times that marine mammals could be impacted by explosions used during Navy training activities. The Navy's quantitative analysis to determine impacts on marine mammals uses the Navy Acoustic Effects Model to produce initial estimates of the number of instances that animals may experience these effects; these estimates are further refined by considering animal avoidance of sound-producing activities and implementation of procedural mitigation measures. The steps of this quantitative analysis are described in Section 3.0.1.2 (Navy's Quantitative Analysis to Determine Impacts to Sea Turtles and Marine Mammals), which takes into account:

- criteria and thresholds used to predict impacts from explosives (see below);
- the density (U.S. Department of the Navy, 2020b) and spatial distribution (Watwood et al., 2018) of marine mammals; and
- the influence of environmental parameters (e.g., temperature, depth, salinity) on sound propagation and explosive energy when estimating the received sound level and pressure on the animals.

A detailed explanation of this analysis is provided in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (U.S. Department of the Navy, 2018d).

### **Criteria and Thresholds used to Estimate Impacts on Marine Mammals from Explosives**

#### **Mortality and Injury from Explosives**

As discussed above in Section 3.8.3.2.1.1 (Injury), two metrics have been identified as predictive of injury: impulse and peak pressure. Peak pressure contributes to the "crack" or "stinging" sensation of a blast wave, compared to the "thump" associated with received impulse. Older military reports documenting exposure of human divers to blast exposure generally describe peak pressure exposures around 100 psi (237 dB re 1  $\mu$ Pa SPL peak) to feel like slight pressure or stinging sensation on skin, with no enduring effects (Christian & Gaspin, 1974).

Because data on explosive injury do not indicate a set threshold for injury, rather a range of risk for explosive exposures, two sets of criteria are provided for use in non-auditory injury assessment. The exposure thresholds are used to estimate the number of animals that may be affected during Navy training activities (Table 3.8-26). The thresholds for the farthest range to effect are based on the received level at which 1 percent risk of onset is predicted and are useful for assessing potential effects to marine mammals and the level of potential impacts covered by the mitigation zones. Increasing animal mass and increasing animal depth both increase the impulse thresholds (i.e., decrease susceptibility), whereas smaller mass and decreased animal depth reduce the impulse thresholds (i.e., increase susceptibility). For impact assessment, marine mammal populations are assumed to be 70 percent adult and 30 percent calf/pup. Sub-adult masses are used to determine onset of effect, in order to estimate the farthest range at which an effect may first be observable. The derivation of these

injury criteria and the species mass estimates are provided in the technical report *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)* (U.S. Department of the Navy, 2017a).

**Table 3.8-26: Criteria to Quantitatively Assess Non-Auditory Injury Due to Explosions in Water**

<i>Impact Category</i>	<i>Impact Threshold</i>	<i>Threshold for Farthest Range to Effect<sup>2</sup></i>
Mortality <sup>1</sup>	$144M^{1/3} \left(1 + \frac{D}{10.1}\right)^{1/6} \text{ Pa-s}$	$103 \left(1 + \frac{D}{10.1}\right)^{1/6} \text{ Pa-s}$
Injury <sup>1</sup>	$65.8M^{1/3} \left(1 + \frac{D}{10.1}\right)^{1/6} \text{ Pa-s}$	$47.5M^{1/3} \left(1 + \frac{D}{10.1}\right)^{1/6} \text{ Pa-s}$
	243 dB re 1 µPa SPL peak	237 dB re 1 µPa SPL peak

<sup>1</sup> Impulse delivered over 20 percent of the estimated lung resonance period. See U.S. Department of the Navy (2017a).

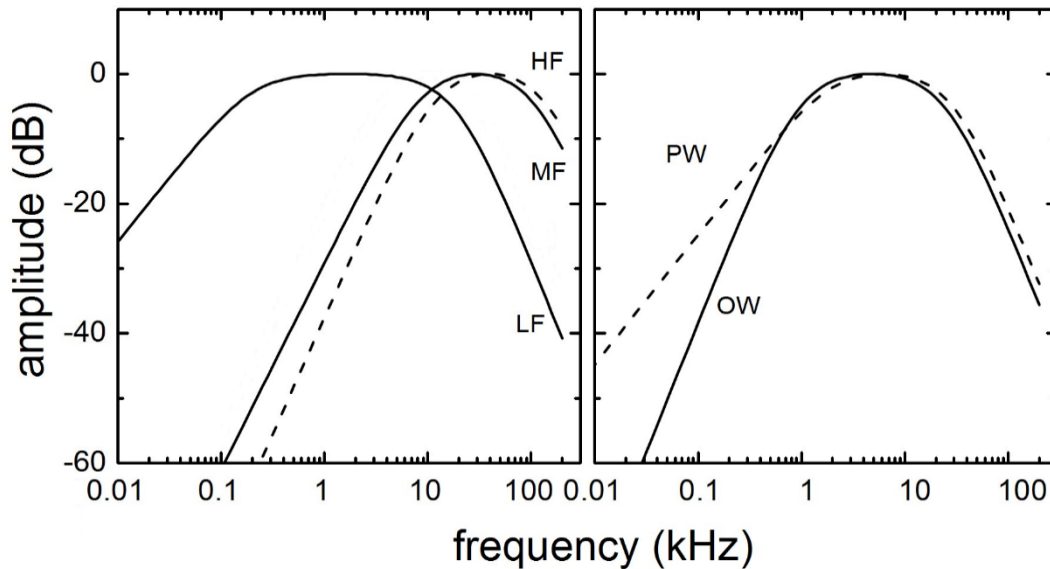
<sup>2</sup> Threshold for one percent risk used to assess mitigation effectiveness.

Notes: D = animal depth (m), dB re 1 µPa = decibels referenced to 1 micropascal, M = animal mass (kg), Pa-s = Pascal-second, SPL = sound pressure level.

When explosive ordnance (e.g., bomb or missile) detonates, fragments of the weapon are thrown at high-velocity from the detonation point, which can injure or kill marine mammals if they are struck. Risk of fragment injury reduces exponentially with distance as the fragment density is reduced. Fragments underwater tend to be larger than fragments produced by in-air explosions (Swisdak & Montanaro, 1992). Underwater, the friction of the water would quickly slow these fragments to a point where they no longer pose a threat. On the other hand, the blast wave from an explosive detonation moves efficiently through the seawater. Because the ranges to mortality and injury due to exposure to the blast wave are likely to far exceed the zone where fragments could injure or kill an animal, the above thresholds are assumed to encompass risk due to fragmentation.

### **Auditory Weighting Functions**

Animals are not equally sensitive to noise at all frequencies. To capture the frequency-dependent nature of the effects of noise, auditory weighting functions are used (Figure 3.8-13). Auditory weighting functions are mathematical functions based on a generic band-pass filter and incorporate species-specific hearing abilities to calculate a weighted received sound level in units SPL or SEL. Due to the band pass nature of auditory weighting functions, they resemble an inverted “U” shape with amplitude plotted as a function of frequency. The flatter portion of the plotted function, where the amplitude is closest to zero, is the emphasized frequency range (i.e., the pass-band), while the frequencies below and above this range (where amplitude declines) are de-emphasized.



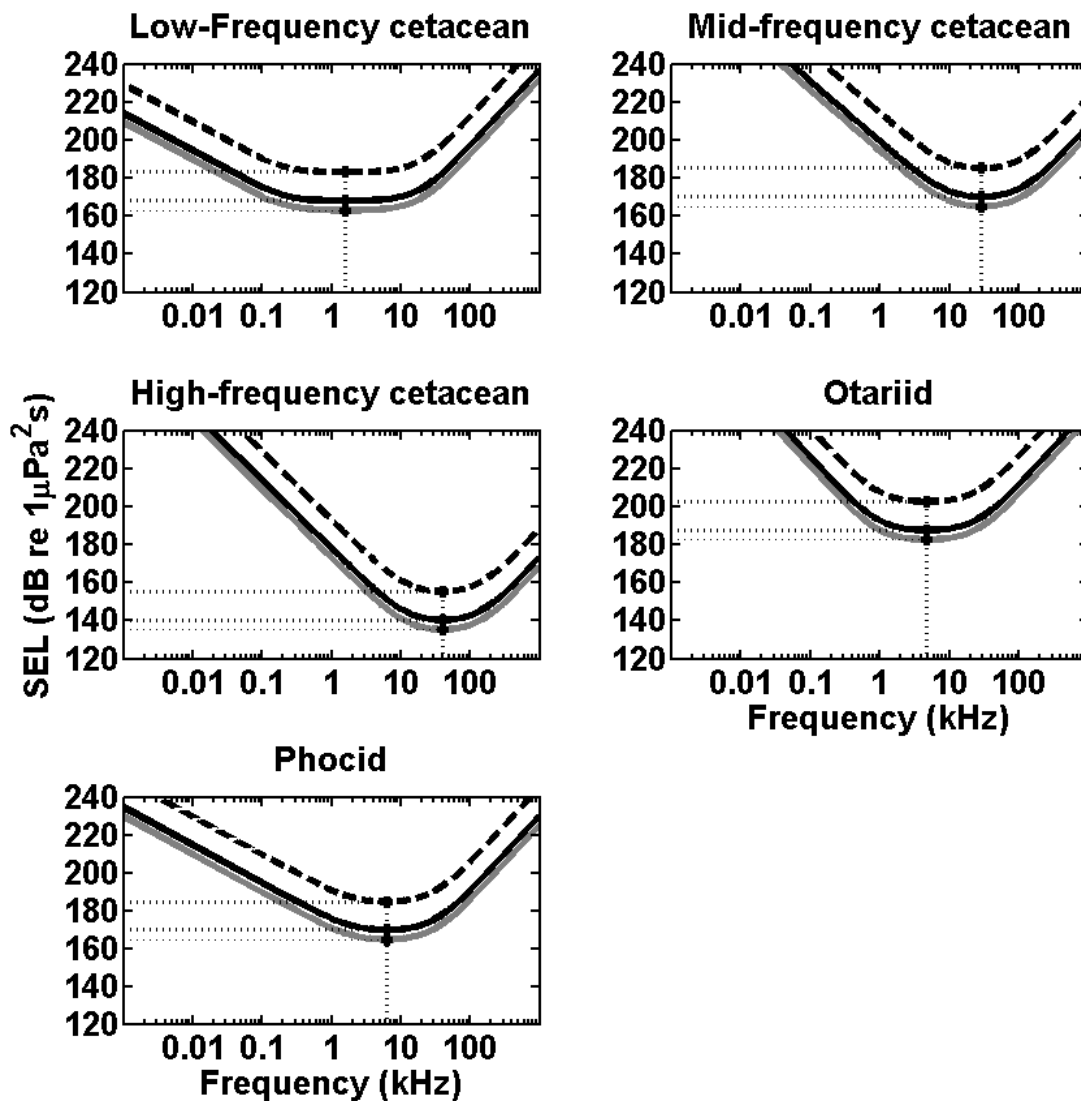
Source: See U.S. Department of the Navy (2017a) for parameters used to generate the functions and more information on weighting function derivation.

Notes: MF = mid-frequency cetacean, HF = high-frequency cetacean, LF = low-frequency cetacean, PW = phocid (in-water), and OW = otariid and other non-phocid marine carnivores (in-water)

**Figure 3.8-13: Navy Phase III Weighting Functions for All Species Groups**

### Hearing Loss from Explosives

Criteria used to define threshold shifts from explosions are derived from the two known studies designed to induce TTS in marine mammals from impulsive sources. Finneran et al. (2002) reported behaviorally measured TTS of 6 and 7 dB in a beluga exposed to single impulses from a seismic water gun and Lucke et al. (2009) reported AEP-measured TTS of 7 to 20 dB in a harbor porpoise exposed to single impulses from a seismic air gun. Since marine mammal PTS data from impulsive noise exposures do not exist, onset-PTS levels for all groups were estimated by adding 15 dB to the threshold for non-impulsive sources. This relationship was derived by Southall et al. (2007) from impulsive noise TTS growth rates in chinchillas. These frequency dependent thresholds are depicted by the exposure functions for each group's range of best hearing (Figure 3.8-14). Weighted sound exposure thresholds for underwater explosive sounds used in the analysis are shown in Table 3.8-27).



Notes: The dark dashed curve is the exposure function for PTS onset, the solid black curve is the exposure function for TTS onset, and the light grey curve is the exposure function for behavioral response. Small dashed lines indicate the SEL threshold for behavioral response, TTS, and PTS onset at each group's most sensitive frequency (i.e., the weighted SEL threshold).

**Figure 3.8-14: Navy Phase III Behavioral, TTS, and PTS Exposure Functions for Explosives**

**Table 3.8-27: Navy Phase III Weighted Sound Exposure Thresholds for Underwater Explosive Sounds**

Hearing Group	Explosive Sound Source				
	Behavior (SEL) weighted (dB)	TTS (SEL) weighted (dB)	TTS (Peak SPL) unweighted (dB)	PTS (SEL) weighted (dB)	PTS (Peak SPL) unweighted (dB)
Low-frequency Cetacean (LF)	163	168	213	183	219
Mid-frequency Cetacean (MF)	165	170	224	185	230
High-frequency Cetacean (HF)	135	140	196	155	202
Otariids <sup>1</sup> in water (OW)	183	188	226	203	232
Phocid seal in water (PW)	165	170	212	185	218

<sup>1</sup> Threshold shift for mustelids (sea otters) is assessed using the otariid sound exposure thresholds. Any behavioral reactions by sea otters are assumed to occur within the TTS threshold.

Notes: dB = decibels, PTS = permanent threshold shift, SEL = sound exposure level, SPL = sound pressure level, and TTS = temporary threshold shift.

### Behavioral Responses from Explosives

Marine mammals may be exposed to isolated impulses in their natural environment (e.g., lightning). For single explosions at received sound levels below hearing loss thresholds, the most likely behavioral response is a brief alerting or orienting response; therefore, the analysis assumes that any modeled instance of temporally or spatially separated detonations occurring in a single 24-hour period could result in harassment under the MMPA for military readiness activities within the range to TTS. Some multiple explosive exercises, such as certain naval gunnery exercises, may be treated as a single event because a few explosions occur closely spaced within a very short time (a few seconds). Since no further sounds follow the initial brief impulses, significant behavioral reactions would not be expected to occur. This reasoning was applied to previous shock trials (63 FR 230; 66 FR 87; 73 FR 143) and is extended to the criteria used in this analysis.

If more than one explosive event occurs within any given 24-hour period within a training activity, criteria are applied to predict the number of animals that may have a behavioral reaction at a behavioral threshold 5 dB less than the TTS onset threshold (in SEL). This value is derived from observed onsets of behavioral response by test subjects (bottlenose dolphins) during non-impulsive TTS testing (Schlundt et al., 2000).

Although there is no research on the effects of explosives on sea otter behavior, based on their low reactivity to other acoustic and anthropogenic stressors, sea otters exposed to received levels below the threshold for TTS are assumed to be unlikely to exhibit behavioral responses that would be considered “harassment” under the MMPA for military readiness activities, if behavioral reactions to distant sounds occur at all.

### Accounting for Mitigation

The Navy will implement mitigation measures to avoid or reduce potential impacts from explosives on marine mammals, as described in Section 5.3.3 (Explosive Stressors). Procedural mitigation measures

include delaying or ceasing applicable detonations when a marine mammal is observed in a mitigation zone. The mitigation zones for explosives extend beyond the respective average ranges to mortality. Navy impact analyses typically consider the potential for procedural mitigation to reduce the risk of mortality due to exposure to explosives; however, the Navy Acoustic Effects Model estimated zero mortality takes for all marine mammal species in the TMAA. Therefore, mitigation for explosives is discussed qualitatively but was not factored into the quantitative analysis for marine mammals under Alternative 1. A detailed explanation of the quantitative analysis process is provided in the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing* (U.S. Department of the Navy, 2018d).

The Navy will also implement mitigation measures to prohibit the use of explosives at or near the surface from June 1 through September 30 in the North Pacific Right Whale Mitigation Area, and from April 1 through October 31 in the Portlock Bank Mitigation Area. Mitigation areas are designed to help avoid or reduce impacts during biologically important life processes within particularly important habitat areas. The benefits of mitigation areas are discussed qualitatively in terms of the context of impact avoidance or reduction.

#### 3.8.3.2.2.2 Impact Ranges for Explosives

The following section provides the range (distance) over which specific physiological or behavioral effects are expected to occur based on the explosive criteria and the explosive propagation calculations from the Navy Acoustic Effects Model (Section 3.8.3.2.2.1, Methods for Analyzing Impacts from Explosives). The range to effects is shown for a range of explosive bins, from E5 (greater than 5–10 lb. net explosive weight) to E12 (greater than 650 lb. to 1,000 lb. net explosive weight). Ranges are determined by modeling the distance that noise from an explosion will need to propagate to reach exposure level thresholds specific to a hearing group that will cause behavioral response, TTS, PTS, and non-auditory injury. Range to effects is important information in not only predicting impacts from explosives, but also in verifying the accuracy of model results against real-world situations and assessing the level of impact that will likely be mitigated within applicable mitigation zones.

The ranges are the distance where the threshold is not exceeded at any depth where animals could be present (excluding negligible small convergence points in some instances). Thus, portions of the water column within the ranges shown would not exceed threshold (i.e., the range does not represent a cylinder of effect in the water column). In some instances, a significant portion of the water column within the ranges shown may not exceed threshold. These differences in propagation are captured in the actual estimation of takes within the Navy Acoustic Effects Model.

Table 3.8-28 shows the minimum, average, and maximum ranges due to varying propagation conditions to non-auditory injury as a function of animal mass and explosive bin. Ranges to gastrointestinal tract injury typically exceed ranges to slight lung injury; therefore, the maximum range to effect is not mass-dependent. Animals within these water volumes would be expected to receive minor injuries at the outer ranges, increasing to more substantial injuries, and finally mortality as an animal approaches the detonation point. Ranges to mortality, based on animal mass, are shown in Table 3.8-29.

Table 3.8-30 through Table 3.8-41 show the minimum, average, and maximum ranges to onset of auditory and behavioral effects based on the thresholds described in Section 3.8.3.2.2.1 (Methods for Analyzing Impacts from Explosives) are provided for a representative source depth and cluster size (the number of rounds fired [or buoys dropped] within a very short duration) for each bin. For events with multiple explosions, sound from successive explosions can be expected to accumulate and increase the



range to the onset of an impact based on SEL thresholds. Modeled ranges to TTS and PTS based on peak pressure for a single explosion generally exceed the modeled ranges based on SEL even when accumulated for multiple explosions. Peak pressure-based ranges are estimated using the best available science; however, data on peak pressure at far distances from explosions are very limited. For additional information on how ranges to impacts from explosions were estimated, see the technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase III Training and Testing Ranges* (U.S. Department of the Navy, 2018d).

**Table 3.8-28: Ranges to Non-Auditory Injury (in meters) for All Marine Mammal Hearing Groups**

<i>Bin<sup>1</sup></i>	<i>Range to Non-Auditory Injury (meters)<sup>2</sup></i>
E5	40 (40–40)
E9	121 (90–130)
E10	152 (100–160)
E12	190 (110–200)

<sup>1</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

<sup>2</sup>Average distance is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

Notes: All ranges to non-auditory injury within this table are driven by gastrointestinal tract injury thresholds regardless of animal mass.

**Table 3.8-29: Ranges to Mortality (in meters) for All Marine Mammal Hearing Groups as a Function of Animal Mass**

<i>Bin<sup>1</sup></i>	<i>Animal Mass Intervals (kg)<sup>2</sup></i>					
	<i>10</i>	<i>250</i>	<i>1,000</i>	<i>5,000</i>	<i>25,000</i>	<i>72,000</i>
E5	13 (12–14)	7 (4–11)	3 (3–4)	2 (1–3)	1 (1–1)	1 (0–1)
E9	35 (30–40)	20 (13–30)	10 (9–13)	7 (6–9)	4 (3–4)	3 (2–3)
E10	43 (40–50)	25 (16–40)	13 (11–16)	9 (7–11)	5 (4–5)	4 (3–4)
E12	55 (50–60)	30 (20–50)	17 (14–20)	11 (9–14)	6 (5–7)	5 (4–6)

<sup>1</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

<sup>2</sup>Average distance to mortality (meters) is depicted above the minimum and maximum distances, which are in parentheses for each animal mass interval.

**Table 3.8-30: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for High-Frequency Cetaceans**

<i>Range to Effects for Explosives: High-frequency cetaceans<sup>1</sup></i>					
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	910 (850–975)	1,761 (1,275–2,275)	2,449 (1,775–3,275)
		7	1,275 (1,025–1,525)	3,095 (2,025–4,525)	4,664 (2,275–7,775)
E9	0.1	1	1,348 (1,025–1,775)	3,615 (2,025–5,775)	5,365 (2,525–8,525)
E10	0.1	1	1,546 (1,025–2,025)	4,352 (2,275–7,275)	5,949 (2,525–9,275)
E12	0.1	1	1,713 (1,275–2,025)	5,115 (2,275–7,775)	6,831 (2,775–10,275)

<sup>1</sup>Average distance (meters) to PTS, TTS, and behavioral thresholds are depicted above the minimum and maximum distances, which are in parentheses. Values depict the range produced by SEL hearing threshold criteria levels.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

**Table 3.8-31: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for High-Frequency Cetaceans**

<i>Range to Effects for Explosives: High-frequency cetaceans<sup>1</sup></i>				
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	1,161 (1,000–1,525)	1,789 (1,025–2,275)
		7	1,161 (1,000–1,525)	1,789 (1,025–2,275)
E9	0.1	1	2,331 (1,525–2,775)	5,053 (2,025–9,275)
E10	0.1	1	2,994 (1,775–4,525)	7,227 (2,025–14,775)
E12	0.1	1	4,327 (2,025–7,275)	10,060 (2,025–22,275)

<sup>1</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-32: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Low-Frequency Cetaceans**

<i>Range to Effects for Explosives: Low-frequency cetaceans<sup>1</sup></i>					
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	171 (100–190)	633 (230–825)	934 (310–1,525)
		7	382 (170–450)	1,552 (380–5,775)	3,712 (600–13,025)
E9	0.1	1	453 (180–550)	3,119 (550–9,025)	6,462 (1,275–19,275)
E10	0.1	1	554 (210–700)	4,213 (600–13,025)	9,472 (1,775–27,275)
E12	0.1	1	643 (230–825)	6,402 (1,275–19,775)	13,562 (2,025–34,775)

<sup>1</sup>Average distance (meters) to PTS, TTS, and behavioral thresholds are depicted above the minimum and maximum distances, which are in parentheses. Values depict the range produced by SEL hearing threshold criteria levels.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

**Table 3.8-33: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Low-Frequency Cetaceans**

<i>Range to Effects for Explosives: Low-frequency cetaceans<sup>1</sup></i>				
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	419 (170–500)	690 (210–875)
		7	419 (170–500)	690 (210–875)
E9	0.1	1	855 (270–1,275)	1,269 (400–1,775)
E10	0.1	1	953 (300–1,525)	1,500 (450–2,525)
E12	0.1	1	1,135 (360–1,525)	1,928 (525–4,775)

<sup>1</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-34: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Mid-Frequency Cetaceans**

<i>Range to Effects for Explosives: Mid-frequency cetaceans<sup>1</sup></i>					
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	79 (75–80)	363 (360–370)	581 (550–600)
		7	185 (180–190)	777 (650–825)	1,157 (800–1,275)
E9	0.1	1	215 (210–220)	890 (700–950)	1,190 (825–1,525)
E10	0.1	1	275 (270–280)	974 (750–1,025)	1,455 (875–1,775)
E12	0.1	1	340 (340–340)	1,164 (825–1,275)	1,746 (925–2,025)

<sup>1</sup>Average distance (meters) to PTS, TTS, and behavioral thresholds are depicted above the minimum and maximum distances, which are in parentheses. Values depict the range produced by SEL hearing threshold criteria levels.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

**Table 3.8-35: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Mid-Frequency Cetaceans**

<i>Range to Effects for Explosives: Mid-frequency cetaceans<sup>1</sup></i>				
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	158 (150–160)	295 (290–300)
		7	158 (150–160)	295 (290–300)
E9	0.1	1	463 (430–470)	771 (575–850)
E10	0.1	1	558 (490–575)	919 (625–1,025)
E12	0.1	1	679 (550–725)	1,110 (675–1,275)

<sup>1</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-36: SEL Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Otariids and Mustelids**

<i>Range to Effects for Explosives: Otariids<sup>1</sup></i>					
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	25 (24–25)	110 (110–110)	185 (180–190)
		7	58 (55–60)	265 (260–270)	443 (430–450)
E9	0.1	1	68 (65–70)	320 (310–330)	512 (490–525)
E10	0.1	1	88 (85–90)	400 (390–410)	619 (575–675)
E12	0.1	1	105 (100–110)	490 (470–500)	733 (650–825)

<sup>1</sup>Average distance (meters) to PTS, TTS, and behavioral thresholds are depicted above the minimum and maximum distances, which are in parentheses. Values depict the range produced by SEL hearing threshold criteria levels.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

**Table 3.8-37: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Otariids and Mustelids**

<i>Range to Effects for Explosives: Otariids<sup>1</sup></i>				
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	128 (120–130)	243 (240–250)
		7	128 (120–130)	243 (240–250)
E9	0.1	1	383 (380–390)	656 (600–700)
E10	0.1	1	478 (470–480)	775 (675–850)
E12	0.1	1	583 (550–600)	896 (750–1,025)

<sup>1</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>2</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-38: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Phocids<sup>1</sup>**

<i>Range to Effects for Explosives: Phocids<sup>1</sup></i>					
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	150 (150–150)	681 (675–700)	1,009 (975–1,025)
		7	360 (350–370)	1,306 (1,025–1,525)	1,779 (1,275–2,275)
E9	0.1	1	425 (420–430)	1,369 (1,025–1,525)	2,084 (1,525–2,775)
E10	0.1	1	525 (525–525)	1,716 (1,275–2,275)	2,723 (1,525–4,025)
E12	0.1	1	653 (650–675)	1,935 (1,275–2,775)	3,379 (1,775–5,775)

<sup>1</sup>Excluding elephant seals

<sup>2</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>3</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-39: Peak Pressure Based Ranges to Onset PTS and Onset TTS (in meters) for Phocids<sup>1</sup>**

<i>Range to Effects for Explosives: Phocids<sup>1</sup></i>				
<i>Bin<sup>2</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	537 (525–550)	931 (875–975)
		7	537 (525–550)	931 (875–975)
E9	0.1	1	1,150 (1,025–1,275)	1,845 (1,275–2,525)
E10	0.1	1	1,400 (1,025–1,775)	2,067 (1,275–2,525)
E12	0.1	1	1,713 (1,275–2,025)	2,306 (1,525–2,775)

<sup>1</sup>Excluding elephant seals

<sup>2</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>3</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

**Table 3.8-40: SEL-Based Ranges to Onset PTS, Onset TTS, and Behavioral Reaction (in meters) for Phocids (Elephant Seals)<sup>1</sup>**

<i>Range to Effects for Explosives: Phocids (Elephant Seals)<sup>2</sup></i>					
<i>Bin<sup>3</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>	<i>Behavioral</i>
E5	0.1	1	150 (150–150)	688 (675–700)	1,025 (1,025–1,025)
		7	360 (350–370)	1,525 (1,525–1,525)	2,345 (2,275–2,525)
E9	0.1	1	425 (420–430)	1,775 (1,775–1,775)	2,858 (2,775–3,275)
E10	0.1	1	525 (525–525)	2,150 (2,025–2,525)	3,421 (3,025–4,025)
E12	0.1	1	656 (650–675)	2,609 (2,525–3,025)	4,178 (3,525–5,775)

<sup>1</sup>Elephant seals are separated from other phocids due to their dive behavior, which far exceeds the dive depths of the other phocids analyzed.

<sup>2</sup>Average distance (meters) to PTS, TTS, and behavioral thresholds are depicted above the minimum and maximum distances which are in parentheses. Values depict the range produced by SEL hearing threshold criteria levels.

<sup>3</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, SEL = sound exposure level, TTS = temporary threshold shift

**Table 3.8-41: Peak Pressure-Based Ranges to Onset PTS and Onset TTS (in meters) for Phocids (Elephant Seals)<sup>1</sup>**

<i>Range to Effects for Explosives: Phocids (Elephant Seals)<sup>2</sup></i>				
<i>Bin<sup>3</sup></i>	<i>Source Depth (m)</i>	<i>Cluster Size</i>	<i>PTS</i>	<i>TTS</i>
E5	0.1	1	537 (525–550)	963 (950–975)
		7	537 (525–550)	963 (950–975)
E9	0.1	1	1,275 (1,275–1,275)	2,525 (2,525–2,525)
E10	0.1	1	1,775 (1,775–1,775)	3,046 (3,025–3,275)
E12	0.1	1	2,025 (2,025–2,025)	3,539 (3,525–3,775)

<sup>1</sup>Elephant seals are separated from other phocids due to their dive behavior, which far exceeds the dive depths of the other phocids analyzed

<sup>2</sup>Average distance (meters) is shown with the minimum and maximum distances due to varying propagation environments in parentheses.

<sup>3</sup>Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)

Notes: PTS = permanent threshold shift, TTS = temporary threshold shift

#### **3.8.3.2.2.3 Impacts from Explosives Under the No Action Alternative**

Under the No Action Alternative, proposed Navy training activities would not occur within the TMAA. The impacts associated with Navy training activities would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing Navy training activities.

#### **3.8.3.2.2.4 Impacts from Explosives Under Alternative 1**

Training activities under Alternative 1 would involve detonations in-air at altitudes above 10 m and higher and detonations at or near the surface occurring at or below 10 m in altitude. As noted previously, those detonations occurring at or near the surface were modeled as if they occurred underwater and were analyzed for their potential underwater acoustic effects on marine mammals. The use of explosives would occur throughout the TMAA and are typically dispersed in space and time. The number and type (i.e., source bin) of explosives that would be used during training under Alternative 1 are described in Section 3.0.4.2 (Explosive Stressors). Activities using explosives would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Navy Activities Descriptions). The proposed use of explosives for training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Although the existing baseline conditions have not changed appreciably, and no new Navy training activities are being proposed for use in the TMAA in this SEIS/OEIS, a detailed re-analysis of Alternative 1 with respect to marine mammals is provided here to supplant previous analysis based on available new literature, adjusted sound exposure criteria, and new acoustic effects modeling.

#### **Presentation of Estimated Impacts from the Quantitative Analysis**

The results of the analysis of potential impacts on marine mammals from explosives (see above Section 3.8.3.2.2.1, Methods for Analyzing Impacts from Explosives) are discussed below. The numbers of potential impacts estimated for individual species of marine mammals from exposure to explosive energy and sound for training activities under Alternative 1 are shown in Appendix C (Estimated Marine Mammal and Sea Turtle Impacts from Exposure to Acoustic and Explosive Stressors Under Navy Training Activities).

Since training activities involving explosions for this SEIS/OEIS only occur in the TMAA, the predictions of numbers of marine mammals that may be affected are assessed solely within the TMAA. The number of explosive sources in this SEIS/OEIS compared with the totals analyzed in the 2016 GOA Final SEIS/OEIS are described in Table 2.6-1 (Current and Proposed Training Activities). Estimated numbers of potential impacts from the quantitative analysis for each species are presented below and estimated impacts for all species can be found in Appendix C (Estimated Marine Mammal and Sea Turtle Impacts from Exposure to Acoustic and Explosive Stressors Under Navy Training Activities).

#### **Mysticetes**

Mysticetes may be exposed to sound and energy from explosions associated with training activities throughout the year. Explosions produce sounds that are within the hearing range of mysticetes (see Section 3.8.2.1.4, Hearing and Vocalization). Potential impacts from explosive energy and sound include non-auditory injury, behavioral reactions, physiological stress, masking, and hearing loss. The quantitative analysis estimates behavioral reactions, TTS, and PTS in mysticetes. Impact ranges for mysticetes exposed to explosive sound and energy are discussed under low-frequency cetaceans in Section 3.8.3.2.2.2 (Impact Ranges for Explosives).



Mysticetes that do experience threshold shift from explosive sounds may have reduced ability to detect biologically important sounds (e.g., social vocalizations) until their hearing recovers. Recovery from threshold shift begins almost immediately after the noise exposure ceases and can take a few minutes to a few days, depending on the severity of the initial shift, to recover. TTS would recover fully and PTS would leave some residual hearing loss. Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. Noise from explosions is broadband with most energy below a few hundred Hertz; therefore, any hearing loss from exposure to explosive sounds is likely to be broadband with effects predominantly at lower frequencies. During the short period that a mysticete had TTS, or permanently for PTS, social calls from conspecifics could be more difficult to detect or interpret, the ability to detect predators may be reduced, and the ability to detect and avoid sounds from approaching vessels or other stressors might be reduced. It is unclear how or if mysticetes use sound for finding prey or feeding; therefore, it is unknown whether a TTS would affect a mysticete's ability to locate prey or rate of feeding.

Research and observations of auditory masking in marine mammals due to impulsive sounds are discussed in Section 3.8.3.2.1.4 (Masking). Explosions introduce low-frequency, broadband sounds into the environment, which could mask hearing thresholds in mysticetes that are nearby, although sounds from explosions last for only a few seconds at most. Masking due to time-isolated detonations would not be significant. Activities that have multiple detonations such as some naval gunfire exercises could create some masking for mysticetes in the area over the short duration of the event. Potential costs to mysticetes from masking are similar to those discussed above for TTS, with the primary difference being that the effects of masking are only present when the sound from the explosion is present within the water and the effect is over the moment the sound has ceased.

Research and observations (see Behavioral Responses from Explosives) show that if mysticetes are exposed to impulsive sounds such as those from explosives, they may react in a variety of ways, which may include alerting, startling, breaking off feeding dives and surfacing, diving or swimming away, changing vocalization, or showing no response at all. Overall, mysticetes have been observed to be more reactive to acoustic disturbance when a noise source is located directly on their migration route. Mysticetes disturbed while migrating could pause their migration or route around the disturbance. Animals disturbed while engaged in other activities such as feeding or reproductive behaviors may be more likely to ignore or tolerate the disturbance and continue their natural behavior patterns. Because noise from most activities using explosives is short term and intermittent, and because detonations usually occur within a small area, behavioral reactions from mysticetes are likely to be short-term and low to moderate severity.

Physiological stress could be caused by injury or hearing loss and could accompany any behavioral reaction as well. Research and observations of physiological stress in marine mammals are discussed in Section 3.8.3.2.1.3 (Physiological Stress). Due to the short-term and intermittent use of explosives, physiological stress is also likely to be short term and intermittent. Long-term consequences from physiological stress due to the sound of explosives would not be expected.

#### **North Pacific Right Whales (Endangered Species Act-Listed)**

North Pacific right whales may be exposed to sound or energy from explosions associated with training activities April through October. Although North Pacific right whales are considered rare in the TMAA due to their low abundance, their occurrence in the TMAA is year round and are most likely to be

present June through September. The quantitative analysis estimates a behavioral reaction under Alternative 1 (Table 3.8-42). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Eastern North Pacific Stock (Table 3.8-42).

Even if an individual right whale experiences a behavioral reaction a few times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. In addition to implementing procedural mitigation for explosives, from June through September (i.e., the months when North Pacific right whales are most likely to be present in the TMAA) the Navy will not use explosives at or near the surface within the North Pacific Right Whale Mitigation Area. This mitigation area encompasses the portion of the biologically important habitat identified by Ferguson et al. (2015) for North Pacific right whale feeding that overlaps the TMAA. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of North Pacific right whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed North Pacific right whales. The Navy will consult with NMFS as required by section 7(a)(2) of the ESA.*

**Table 3.8-42: Estimated Impacts on Individual North Pacific Right Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
Stock	Behavioral	TTS	PTS	Injury
Eastern North Pacific	1	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

#### **Humpback Whales (some DPSs are Endangered Species Act-Listed)**

Humpback whales may be exposed to sound or energy from explosions associated with training activities April through October. Although the timing of humpback whale migrations may change year to year, they are most likely to be present in the TMAA June through September. Impacts have been modeled for the Hawaii (Central North Pacific stock) population of humpback whales, which are not ESA-listed, and for the Mexico (California, Oregon, and Washington stock), Central America (California, Oregon, and Washington stock), and Western North Pacific DPS (Western North Pacific stock) populations of humpback whales, which are ESA-listed.

The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reactions and TTS under Alternative 1 (Table 3.8-43). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to multiple stocks (Table 3.8-43). In addition to procedural mitigation, the Navy will implement mitigation within mitigation areas, which will further help avoid or reduce potential impacts from explosives on humpback whales. The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft operating within the TMAA to the possible presence of increased concentrations of humpback whales

from June 1 to September 30. To maintain safety of navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of large whales that may be vulnerable to vessel strikes or potential impacts from training activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training activities and to aid in the implementation of procedural mitigation during activities using explosives. The Navy will not use explosives at or near the surface from June 1 to September 30 within the North Pacific Right Whale Mitigation Area or from April 1 to October 31 in the Portlock Bank Mitigation Area, which overlap a portion of the proposed humpback whale critical habitat. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), proposed critical habitat for the ESA-listed Western North Pacific and Mexico DPS of humpback whales (NMFS designated Region/Units 5, 7, and 8) overlaps the northwestern portion of the TMAA over the continental shelf.

As described in Section 3.8.2.3 (Humpback Whale [*Megaptera novaeangliae*]), one essential feature was identified for humpback whale critical habitat: prey species, primarily euphausiids and small pelagic schooling fishes, of sufficient quality, abundance, and accessibility within humpback whale feeding areas to support feeding and population growth. This essential feature has the potential to be affected by explosives used under the Proposed Action. The best available science and description of methods used to assess explosive impacts to fishes (i.e., prey species) are provided in Section 3.6.3.2 (Explosive Stressors). The thresholds applied to estimate potential mortality impacts on fishes are based on a conservative application of available data. As shown in Table 3.6-8 in Section 3.6.3.2.2.1 (Methods for Analyzing Impacts from Explosives), the average range to fish mortality due to an explosive in bin E12 (> 650–1,000 lb. net explosive weight [NEW]), the largest explosive proposed in the TMAA, is 800 m. The ranges for smaller explosive bins are correspondingly shorter. Fish that occur within the estimated ranges to mortality could be killed, and those that are killed within the proposed critical habitat would no longer be available as prey items. Other potential impacts from exposure to explosions include injury, TTS, physiological stress, and behavioral reactions. The ranges to these lower level impacts would be considerably larger than the range to mortality. However, these impacts would not be anticipated to remove individual fish (i.e., prey species) from the population, 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 have been shown to be relatively resilient to explosive exposures, and it is anticipated that other invertebrates (including euphausiids) would respond similarly to explosive exposures. Although individuals of widespread marine invertebrate species could be killed during an explosion, the number of such invertebrates affected would be small relative to overall population sizes, and activities would be unlikely to impact survival, growth, recruitment, or reproduction of populations or subpopulations. Impacts of a limited number of explosions on widespread invertebrate populations, and therefore humpback prey items, would likely be undetectable.

Overall, if prey items are killed within the critical habitat, it is likely that only a low number of individuals and therefore a small portion of prey species populations may be killed. Although some prey items could be killed within the described mortality ranges during an explosive activity, other prey items would likely be available to humpback whales in the immediate area surrounding the activity or would return to the area after the activity is complete. Exposure to explosions would be highly dependent on the limited number of explosive activities that overlap proposed critical habitat and the actual presence of prey

species at the time explosive activities occur. The portion of proposed critical habitat that overlaps the TMAA is limited, and only a small number of explosive training activities could potentially occur within the proposed critical habitat. 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. Population-level impacts on fishes and invertebrates in the TMAA from explosive training activities are not anticipated and would not impact humpback whales through a reduction in prey availability.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of humpback whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed humpback whales and proposed critical habitat. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-43: Estimated Impacts on Individual Humpback Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
Stock	Behavioral	TTS	PTS	Injury
California, Oregon, & Washington	1	0	0	0
Central North Pacific	7	2	0	0
Western North Pacific	0	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

#### **Blue Whales (Endangered Species Act-Listed)**

Blue whales may be exposed to sound or energy from explosions associated with training activities April through October. Although blue whales' occurrence in the TMAA is year round, they are most likely to be present June through December. The quantitative analysis, using the maximum number of explosives per year, estimates behavioral reaction under Alternative 1 (Table 3.8-44). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Eastern North Pacific stock (Table 3.8-44).

Even if an individual blue whale experiences behavioral reactions a few times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of blue whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed blue whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-44: Estimated Impacts on Individual Blue Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Central North Pacific	0	0	0	0
Eastern North Pacific	1	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

#### **Fin Whales (Endangered Species Act-Listed)**

Fin whales may be exposed to sound or energy from explosions associated with training activities April through October. Although fin whales' occurrence in the TMAA is year round, they are most likely to be present June through August. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reaction, TTS and PTS under Alternative 1 (Table 3.8-45). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Northeast Pacific stock (Table 3.8-45).

As described for mysticetes above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. PTS could reduce an animal's ability to detect biologically important sounds; however, as discussed above, hearing loss beyond a minor TTS is unlikely, and a small threshold shift due to exposure to sonar is unlikely to affect the hearing range that fin whales rely upon if it did occur. Nevertheless, PTS could have minor long-term consequences for individuals if it were to occur. This minor consequence for an individual is unlikely to have any long-term consequences for the species or stock. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of fin whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed fin whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-45: Estimated Impacts on Individual Fin Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Northeast Pacific	11	2	2	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Sei Whales (Endangered Species Act-Listed)**

Sei whales may be exposed to sound or energy from explosions associated with training activities April through October. Although sei whales' occurrence in the TMAA is year round, they are considered rare, even during the summer time period. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reaction under Alternative 1 (Table 3.8-46). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Eastern North Pacific stock (Table 3.8-46).

Even if an individual sei whale experiences behavioral reactions a few times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of sei whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed sei whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

**Table 3.8-46: Estimated Impacts on Individual Sei Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Eastern North Pacific	1	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Minke Whales**

Minke whales may be exposed to sound or energy from explosions associated with training activities April through October. Even though very few minke whales have been seen during surveys in the area, their occurrence in the TMAA is considered year round. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reactions under Alternative 1 (Table 3.8-47). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Alaska stock (Table 3.8-47).

Even if an individual minke whale experiences behavioral reactions a few times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of minke whales incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-47: Estimated Impacts on Individual Minke Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Alaska	2	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

**Gray Whales (one DPS is Endangered Species Act-Listed)**

Gray whales may be exposed to sound or energy from explosions associated with training activities April through October. Although Western North Pacific gray whales are rare, both stocks of gray whales are migratory and their occurrence in the TMAA would be seasonal with their highest likelihood of occurring being between June and August. Impacts have been modeled for the Eastern North Pacific stock of gray whales, which are not ESA-listed, and for the Western North Pacific stock of gray whales, which are ESA-listed.

The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives).

In addition to procedural mitigation, the Navy will implement mitigation within mitigation areas, which will further help avoid the already low potential for impacts from explosives on gray whales. The Navy will issue annual seasonal awareness notification messages to alert ships and aircraft operating within the TMAA to the possible presence of increased concentrations of gray whales from April 1 to August 31. To maintain safety of navigation and to avoid interactions with large whales, the Navy will instruct vessels to remain vigilant to the presence of large whales that may be vulnerable to vessel strikes or potential impacts from training activities. Platforms will use the information from the awareness notification messages to assist their visual observation of applicable mitigation zones during training activities and to aid in the implementation of procedural mitigation during activities using active sonar. This mitigation area overlaps habitat within the northernmost corner and southwestern edge of the TMAA that has been identified by Ferguson et al. (2015) as biologically important gray whale migration habitat. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of gray whales.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed gray whales.*

### **Odontocetes**

Odontocetes may be exposed to sound and energy from explosives associated with training activities from April to October. Explosions produce sounds that are within the hearing range of odontocetes (see Section 3.8.2.1.4, Hearing and Vocalization). Potential impacts from explosive energy and sound include non-auditory injury, behavioral reactions, physiological stress, masking, and hearing loss. Impact ranges for odontocetes exposed to explosive sound and energy are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives) under mid-frequency cetaceans for most species, and under high-frequency cetaceans for Dall's porpoises and harbor porpoises.

Non-auditory injuries to odontocetes, if they did occur, could include anything from mild injuries that are recoverable and are unlikely to have long-term consequences, to more serious injuries, including mortality. It is possible for marine mammals to be injured or killed by an explosion in isolated instances. Individuals that sustain injury from explosives could have long-term consequences. Considering that dolphin species for which these impacts are predicted have populations with tens to hundreds of thousands of animals, removing several animals from the population would be unlikely to have measurable long-term consequences for the species or stocks. As discussed in Section 5.3.3 (Explosive Stressors), the Navy will implement procedural mitigation measures to delay or cease detonations when a marine mammal is sighted in a mitigation zone to avoid or reduce potential explosive impacts.

Odontocetes that do experience a hearing threshold shift from explosive sounds may have reduced ability to detect biologically important sounds (e.g., social vocalizations) until their hearing recovers. Recovery from a hearing threshold shift begins almost immediately after the noise exposure ceases. A threshold shift can take a few minutes to a few days, depending on the severity of the initial shift, to recover. TTS would recover fully and PTS would leave some residual hearing loss. Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. Noise from explosions is broadband with most energy below a few hundred Hertz; therefore, any hearing loss from exposure to explosive sounds is likely to be broadband with effects predominantly at lower frequencies. During the period that an odontocete had hearing loss, social calls from conspecifics and sounds from predators such as killer whale vocalizations could be more difficult to detect or interpret, although many of these sounds may be above the frequencies of the threshold shift. Odontocetes use echolocation clicks to find and capture prey. These echolocation clicks and vocalizations are at frequencies above a few kHz, which are less likely to be affected by threshold shift at lower frequencies, and should not affect odontocete's ability to locate prey or rate of feeding.

Research and observations of masking in marine mammals due to impulsive sounds are discussed in Section 3.8.3.2.1.4 (Masking). Explosions introduce low-frequency, broadband sounds into the environment, which could mask hearing thresholds in odontocetes that are nearby, although sounds from explosions last for only a few seconds at most. Also, odontocetes typically communicate, vocalize, and echolocate at higher frequencies that would be less affected by masking noise at lower frequencies such as those produced by an explosion. Masking due to time-isolated detonations would not be significant. Activities that have multiple detonations such as some naval gunfire exercises could create



some masking for odontocetes in the area over the short duration of the event. Potential costs to odontocetes from masking are similar to those discussed above for TTS, with the primary difference being that the effects of masking are only present when the sound from the explosion is present within the water and the effect is over the moment the sound has ceased.

Research and observations (see Section 3.8.3.2.1.5, Behavioral Reactions) show that odontocetes do not typically show strong behavioral reactions to impulsive sounds such as explosions. Reactions, if they did occur, would likely be limited to short ranges, within a few kilometers of multiple explosions. Reactions could include alerting, startling, breaking off feeding dives and surfacing, diving or swimming away, change in vocalization, or showing no response at all. Animals disturbed while engaged in other activities such as feeding or reproductive behaviors may be more likely to ignore or tolerate the disturbance and continue their natural behavior patterns. Because noise from most activities using explosives is short term and intermittent, and because detonations usually occur within a small area, behavioral reactions from odontocetes are likely to be short term and low to moderate severity.

Physiological stress could be caused by injury or hearing loss and could accompany any behavioral reaction as well. Research and observations of physiological stress in marine mammals are discussed in Section 3.8.3.2.1.3 (Physiological Stress). Due to the short-term and intermittent use of explosives, physiological stress is also likely to be short term and intermittent. Long-term consequences from physiological stress due to the sound of explosives would not be expected.

#### **Sperm Whales (Endangered Species Act-Listed)**

Sperm whales may be exposed to sound or energy from explosions associated with training activities April through October. Although sperm whales' occurrence in the TMAA is year round, they are most likely to be present June through September. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of sperm whales.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed sperm whales. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.*

#### **Killer Whales**

Killer whales may be exposed to sound or energy from explosions associated with training activities April through October. Although killer whales' occurrence in the TMAA is year round, the one offshore population and the two transient types are more likely to be present in the majority of the TMAA given the deep and far offshore waters of the Navy training area. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of killer whales.*

### **Pacific White-Sided Dolphins**

Pacific white-sided dolphins may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the Pacific white-sided dolphins' occurrence in the TMAA would be likely year round. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of Pacific white-sided dolphins.*

### **Harbor Porpoises**

Harbor porpoises may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the harbor porpoises' occurrence in the TMAA would be likely year round in the nearshore locations to the shelf break. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of harbor porpoises.*

### **Dall's Porpoises**

Dall's porpoises may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the Dall's porpoises' occurrence in the TMAA would be likely year round. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reaction, TTS, and PTS (Table 3.8-48). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Alaska stock (Table 3.8-48).

TTS and PTS thresholds for high-frequency cetaceans, including Dall's porpoises, are lower than for all other marine mammals, which leads to a higher number of estimated impacts relative to the number of animals exposed to the sound as compared to other hearing groups (e.g., mid-frequency cetaceans). The information available on harbor porpoise behavioral reactions to human disturbance (a closely related species) suggests that these species may be more sensitive and avoid human activity, and sound sources, to a longer range than most other odontocetes. This would make Dall's porpoises less susceptible to hearing loss; therefore, it is likely that the quantitative analysis over-predicted hearing loss impacts (i.e., TTS and PTS) in Dall's porpoises.

As described for odontocetes above, minor to moderate TTS or behavioral reactions to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. PTS in an individual could have no to minor long-term consequences for individuals, although a single minor long-term consequence for an individual is unlikely to lead to long-term consequences for a population. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of Dall's porpoises incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-48: Estimated Impacts on Individual Dall's Porpoise Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Alaska	38	229	45	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Beaked Whales**

Beaked whales may be exposed to sound or energy from explosions associated with training activities April through October. Beaked whales within the GOA TMAA include Baird's beaked whale, Cuvier's beaked whale, and Stejneger's beaked whale. Although beaked whales' occurrence in the TMAA would be likely year round, Cuvier's beaked whales are most likely to be present April through June. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reaction for Cuvier's beaked whale and no impacts on Baird's or Stejneger's beaked whales under Alternative 1 (Table 3.8-49). Impact ranges for these species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the Alaska stock of Cuvier's beaked whales (Table 3.8-49).

Research and observations (see *Behavioral Responses from Explosives*) show that beaked whales are sensitive to human disturbance including noise from sonars, although no research on specific reactions to impulsive sounds or noise from explosions is available. Odontocetes overall have shown little responsiveness to impulsive sounds, although it is likely that beaked whales are more reactive than most other odontocetes. Reactions could include alerting, startling, breaking off feeding dives and surfacing, diving or swimming away, change in vocalization, or showing no response at all. Beaked whales on Navy ranges have been observed leaving the area for a few days during sonar training exercises. It is reasonable to expect that animals may leave an area of more intense explosive activity for a few days; however, most explosive use during Navy activities is short duration, consisting of only a single or few closely timed explosions (i.e., detonated within a few minutes) with a limited footprint due to a single detonation point. Because noise from most activities using explosives is short term and intermittent and because detonations usually occur within a small area, behavioral reactions from beaked whales are likely to be short term and moderate severity.

Even if an individual Cuvier's beaked whale experiences behavioral reactions a few times over the course of a year, impacts are unlikely to have any significant costs or long-term consequences for that individual. Considering these factors and the mitigation measures that would be conducted as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of Baird's beaked whales and Stejneger's beaked whales.*

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the incidental taking of Cuvier's beaked whales. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-49: Estimated Impacts on Individual Cuvier's Beaked Whale Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
Alaska	1	0	0	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Pinnipeds and Mustelids**

Pinnipeds include phocid seals (true seals) and otariids (sea lions and fur seals), and mustelids include sea otters.

As described in Section 3.8.3.2.1.5 (Behavioral Reactions), mustelids have similar or reduced hearing capabilities compared to pinnipeds (specifically otariids). Thus, it is reasonable to assume that mustelids use their hearing similarly to that of otariids, and the types of impacts from exposure explosions may also be similar to those described below for pinnipeds, including behavioral reactions, physiological stress, masking, and hearing loss.

If a pinnipeds or mustelid were to experience TTS from explosive sounds, it may have reduced ability to detect biologically important sounds until their hearing recovers. Recovery from TTS begins almost immediately after the noise exposure ceases and can take a few minutes to a few days, depending on the severity of the initial shift, to fully recover. Threshold shifts do not necessarily affect all hearing frequencies equally, and typically manifest themselves at the exposure frequency or within an octave above the exposure frequency. Noise from explosions is broadband with most energy below a few hundred Hertz; therefore, any hearing loss from exposure to explosive sounds is likely to be broadband with effects predominantly at lower frequencies. During the short period that a pinniped had TTS, social calls from conspecifics could be more difficult to detect or interpret; however, most pinniped vocalizations may be above the frequency of TTS induced by an explosion. Killer whales are one of the pinniped primary predators. Killer whale vocalizations are typically above a few kHz, well above the region of hearing that is likely to be affected by exposure to explosive energy. Therefore, TTS in pinnipeds due to sound from explosions is unlikely to reduce detection of killer whale calls. Pinnipeds may use sound underwater to find prey and feed; therefore, a TTS could have a minor and temporary effect on a phocid seal's ability to locate prey.

Research and observations of auditory masking in marine mammals due to impulsive sounds are discussed in Section 3.8.3.2.1.4 (Masking). Explosions introduce low-frequency, broadband sounds into the environment, which could mask hearing thresholds in pinnipeds that are nearby, although sounds from explosions last for only a few seconds at most. Masking due to time-isolated detonations would not be significant. Activities that have multiple detonations such as some naval gunfire exercises could

create some masking for pinnipeds in the area over the short duration of the event. Potential costs to pinnipeds and mustelids from masking are similar to those discussed above for TTS, with the primary difference being that the effects of masking are only present when the sound from the explosion is present within the water and the effect is over the moment the sound has ceased.

Research and observations (see Behavioral Responses from Explosives) show that pinnipeds may be the least sensitive taxonomic group to most noise sources. They are likely to only respond to loud impulsive sound sources at close ranges by startling, jumping into the water when hauled out, or even cease foraging, but only for brief periods before returning to their previous behavior. Pinnipeds may even experience TTS before exhibiting a behavioral response (Southall et al., 2007). Because noise from most activities using explosives is short term and intermittent, and because detonations usually occur within a small area, behavioral reactions from phocid seals are likely to be short term and low severity.

Physiological stress could be caused by injury or hearing loss and could accompany any behavioral reaction as well. Research and observations of physiological stress in marine mammals are discussed in Section 3.8.3.2.1.3 (Physiological Stress). Due to the short-term and intermittent use of explosives, physiological stress is also likely to be short term and intermittent. Long-term consequences from physiological stress due to the sound of explosives would not be expected.

**Steller Sea Lions (one DPS is Endangered Species Act-Listed)**

Steller sea lions may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the Steller sea lions' occurrence in the TMAA would be likely year round. Impacts have been modeled for the Eastern U.S. stock of Steller sea lions, which are not ESA-listed, and for the Western U.S. stock of Steller sea lions, which are ESA-listed. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of Steller sea lions.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed Steller sea lions.*

**California Sea Lions**

California sea lions may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the California sea lions' occurrence in the TMAA would be seasonal and are most likely to be present April through May. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of California sea lions.*

### **Northern Fur Seals**

Northern fur seals may be exposed to sound or energy from explosions associated with training activities April through October. Although northern fur seals are most likely to be present in the TMAA December through July, males may potentially be present year round. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for these species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of northern fur seals.*

### **Northern Elephant Seals**

Northern elephant seals may be exposed to sound or energy from explosions associated with training activities April through October. The Navy has determined the northern elephant seals' occurrence in the TMAA would be seasonal and are most likely to be present March through October. The quantitative analysis, using the maximum number of explosions per year, estimates behavioral reaction, TTS, and PTS (Table 3.8-50). Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Estimated impacts apply to the California stock (Table 3.8-50).

As described above, minor to moderate behavioral reactions or TTS to an individual over the course of a year are unlikely to have significant costs or long-term consequences for that individual. PTS in an individual could have no to minor long-term consequences for individuals although a single minor long-term consequence for an individual is unlikely to lead to long-term consequences for a population. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would result in the unintentional taking of northern elephant seals incidental to those activities. The Navy will request authorization from NMFS as required by Section 101(a)(5)(A) of the MMPA.*

**Table 3.8-50: Estimated Impacts on Individual Northern Elephant Seal Stocks Within the Study Area per Year from Explosions Used During Training Under Alternative 1**

Estimated Impacts by Effect				
<i>Stock</i>	<i>Behavioral</i>	<i>TTS</i>	<i>PTS</i>	<i>Injury</i>
California	6	9	8	0

Note: Estimated impacts are based on the maximum number of activities in a given year under Alternative 1.

### **Harbor Seals**

Harbor seals may be exposed to sound or energy from explosions associated with training activities April through October. Although harbor seals' occurrence in the TMAA is year round, they are rarely found more than 20 km from shore and are therefore more likely to be present in the inshore water locations. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges

for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stocks would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of harbor seals.*

#### **Ribbon Seals**

Ribbon seals may be exposed to sound or energy from explosions associated with training activities April through October. Although ribbon seals are considered rare in the TMAA, their occurrence is year round and are most likely to be present in the TMAA July through September. The quantitative analysis, using the maximum number of explosions per year, estimates no impacts under Alternative 1. Impact ranges for this species are discussed in Section 3.8.3.2.2.2 (Impact Ranges for Explosives). Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of ribbon seals.*

#### **Northern Sea Otters (one DPS is Endangered Species Act-Listed)**

Northern sea otters are unlikely to be exposed to sound or energy from explosions associated with training activities April through October. Although northern sea otters occur in the Gulf of Alaska year round, they would rarely be present in the TMAA since the normal range and habitat of sea otters is well inland of the TMAA boundaries. Sea otters seldom range more than 2 km from shore, and in this region are mainly concentrated within 400 m from shore because they are benthic foragers. (Bodkin, 2015) notes that sea otters can be found many kilometers from shore in locations where there are shoals far from land, but there are no known offshore populations near the TMAA. Individuals from the Southwest Alaska stock (ESA-listed) are not expected to be present in the TMAA. It is possible that individual sea otters from the Southcentral Alaska stock or the Southeast Alaska stock (neither are ESA-listed) could potentially occur in the nearshore margins of the TMAA. Juvenile males in particular may travel farther offshore (Calambokidis et al., 1987; Laidre et al., 2009; Muto et al., 2017; Riedman & Estes, 1990).

Detonations would generally occur further offshore than the nearshore areas that sea otters inhabit. Impacts are highly unlikely due to limited use of explosives nearshore and the unlikely occurrence of sea otters overlapping with explosions during training activities. In addition, Ghaul and Reichmuth (2014b) have shown that sea otters are not especially well adapted for hearing underwater, which suggests that the function of this sense has been less important in their survival and evolution than in comparison to pinnipeds. Due to their low sensitivity to underwater sounds, their preferred habitat, and the lack of normal geographical overlap between sea otter habitat and training activities, impacts to northern sea otters from Navy training activities involving explosives are highly unlikely to occur. Considering these factors and the mitigation measures that would be implemented as described in Chapter 5 (Mitigation), long-term consequences for the species or stock would not be expected.

*Pursuant to the MMPA, the use of explosives during training activities as described under Alternative 1 would not result in the incidental taking of Northern sea otters.*

*Pursuant to the ESA, the use of explosives during training activities as described under Alternative 1 may affect ESA-listed Northern sea otters. The Navy will consult with USFWS as required by Section 7(a)(2) of the ESA.*

### 3.8.3.3 Secondary Stressors

Navy training activities could pose indirect impacts on marine mammals via habitat or prey as a result of explosives by-products, metals, chemicals, and transmission of disease and parasites. Analysis of the potential impacts on sediment and water quality are discussed in Section 3.3 (Water Resources) of the 2016 GOA Final SEIS/OEIS. The relatively low solubility of most explosives and their degradation products, metals, and chemicals means that concentrations of these contaminants in the marine environment, including those associated with either high-order or low-order detonations, are relatively low and readily diluted. For example, in the Study Area the concentration of unexploded ordnance, explosion byproducts, metals, and other chemicals would never exceed that of a World War II dump site. A series of studies of a World War II dump site off Hawaii have demonstrated only minimal concentrations of degradation products were detected in the adjacent sediments and that there was no detectable uptake in sampled organisms living on or in proximity to the site (Briggs et al., 2016; Carniel et al., 2019; Edwards et al., 2016; Hawaii Undersea Military Munitions Assessment, 2010; Kelley et al., 2016; Koide et al., 2016). It has also been documented that the degradation products of Royal Demolition Explosive are not toxic to marine organisms at realistic exposure levels (Lotufo, 2017; Rosen & Lotufo, 2010). Any remnant undetonated components from explosives such as trinitrotoluene (TNT), royal demolition explosive, and high melting explosive experience rapid biological and photochemical degradation in marine systems (Carniel et al., 2019; Cruz-Urbe et al., 2007; Juhasz & Naidu, 2007; Pavlostathis & Jackson, 2002; Singh et al., 2009; Walker et al., 2006). As another example, the Canadian Forces Maritime Experimental and Test Ranges near Nanoose, British Columbia, began operating in 1965 conducting test events for both U.S. and Canadian forces, which included many of the same test events that are conducted in the TMAA. Environmental analyses of the impacts from years of testing at Nanoose were documented in 1996 and 2005 (Environmental Science Advisory Committee, 2005). These analyses concluded the Navy test activities "...had limited and perhaps negligible effects on the natural environment" (Environmental Science Advisory Committee, 2005). Therefore, based these and other similar applicable findings from multiple Navy ranges and based on the analysis in Section 3.3 (Water Resources) of the 2016 GOA Final SEIS/OEIS, indirect impacts on marine mammals from the training and testing activities in the TMAA would be negligible and would have no long-term effect on habitat.

Secondary stressors from training and testing activities were analyzed for potential indirect impacts on marine mammal prey availability. Underwater explosions could impact other species in the food web, including prey species that marine mammals feed upon. The impacts of explosions would differ depending upon the type of prey species in the area of the detonation. A reduction in availability of prey may cause animals to forage for longer periods, travel to alternate locations, or abandon foraging efforts (National Oceanic and Atmospheric Administration, 2015c). However, there are other factors such as commercial fisheries or competition between species that have much greater and widespread effect than Navy activities. For example, in the 2016 GOA Final SEIS/OEIS, the Navy analyzed effects to humpback whale prey, including zooplankton and fish species not listed under the ESA such as herring and capelin. As previously determined by Navy (U.S. Department of the Navy, 2016a) and by NMFS (National Marine Fisheries Service (2017b); 82 FR 19530), the environmental baseline for the affected



environment includes previous training with sonar that has not affected humpback whale prey. Individual prey have the potential to be affected by the use of underwater explosives; however, this has not resulted in changes to overall prey availability.

The proposed Critical Habitat for humpback whales (see Figure 3.8-2) occurs on the continental shelf and does not overlap the slope or deeper waters of the TMAA where the Navy primarily operates for reasons described in the 2016 GOA Final SEIS/OEIS (see U.S. Department of the Navy (2016a), Section 3.8.3.3.2, Model Predicted Effects from Use of Sonar and Other Active Acoustic Sources; and Section 5.3.3.1.11, Avoiding Marine Species Habitats and Biologically Important Areas). Generally, the Navy does not conduct training with explosives on the shelf. Thus, there is minimal to no overlap of those activities with humpback whale prey species. In addition to explosive use occurring offshore and outside of the areas proposed as Critical Habitat, the Navy's use of the TMAA is short term—a maximum of a few weeks per year and typically occurring every other year. Under the current authorizations and consultations for Navy training activities in the TMAA, NMFS analyzed the potential effects on humpback feeding in designated Biologically Important Areas (see Ferguson et al. (2015); 82 FR 19530) and impacts on ESA-listed fish species (e.g., salmon, steelhead) pursuant to ESA consultations (National Marine Fisheries Service, 2017b). All these factors combined are the basis under which the Navy has determined that there are no national security concerns related to the proposed Critical Habitat overlap with the TMAA, given the one essential feature that should not be affected by the continuation of Navy training in the TMAA.

Based on authorizations pursuant to the MMPA reached by NMFS for all other Navy areas analyzed in the Pacific and Atlantic and because testing and training activities are similar in nature to those analyzed in the Pacific and Atlantic regions, indirect effects (secondary stressors) would be discountable, negligible, or insignificant. Pursuant to the MMPA, indirect effects (secondary stressors) are not expected to result in mortality, Level A harassment, or Level B harassment of any marine mammal. Pursuant to the ESA, indirect effects may affect but are not likely to adversely affect certain ESA-listed marine mammals and would have no effect on marine mammal critical habitats.

#### **3.8.4 Summary of Stressor Assessment (Combined Impacts of All Stressors) on Marine Mammals**

As listed in Section 3.0.4 (Stressors-Based Analysis), this section evaluates the potential for combined impacts of all identified stressors resulting from the Proposed Action. The analysis and conclusions for the potential impacts from each of the individual stressors are discussed in Sections 3.8.3.1 (Acoustic Stressors) through 3.8.3.3 (Secondary Stressors) and, for ESA-listed species, summarized in this section.

Understanding the combined effects of stressors on marine organisms in general and marine mammal populations in particular is extremely difficult to predict (National Academies of Sciences Engineering and Medicine, 2017). Recognizing the difficulties with measuring trends in marine mammal populations, the focus has been on indicators for adverse impacts, including health and other population metrics (National Academies of Sciences Engineering and Medicine, 2017). This recommended use of population indicators is the approach the Navy presented in the 2016 GOA Final EIS/OEIS Section 3.8.5 (Summary of Observations During Previous Navy Activities) and formed part of the 2017 analyses by NMFS in their MMPA authorization (National Marine Fisheries Service, 2017c), and the Biological Opinion for the 2016 GOA Final SEIS/OEIS (National Marine Fisheries Service, 2017a).

Stressors associated with the proposed activities do not typically occur in isolation, but rather occur in some combination. For example, an event involving gunfire may include elements of acoustic, physical disturbance and strike, ingestion, and secondary stressors that are all coincident in space and time. An

analysis of the combined impacts of all stressors considers the potential consequences of additive stressors and synergistic stressors, as described below. This analysis makes the reasonable assumption, which is supported by the Navy Acoustic Effects Model for acoustic stressors, that the majority of exposures to stressors are non-lethal and non-injurious, and instead focuses on consequences potentially impacting marine mammal fitness (e.g., physiology, behavior, reproductive potential).

There are generally two ways that a marine mammal could be exposed to multiple additive stressors. The first would be if a marine mammal were exposed to multiple sources of stress from a single event or activity within a single event (e.g., a Gunnery Exercise event may include the use of a sound source, explosives, and a vessel). The potential for a combination of these impacts from a single activity would depend on the range to effects of each of the stressors and the response or lack of response to that stressor. Most of the activities proposed under Alternative 1 generally involve the use of moving platforms (e.g., ships and aircraft) that may produce one or more stressors; therefore, it is likely that if a marine mammal were within the potential impact range of those activities, it may be impacted by multiple stressors simultaneously. Individual stressors that would otherwise have minimal to no impact, may combine to have a measurable response. However, due to the wide dispersion of stressors, speed of the platforms, general dynamic movement of many military readiness activities, and behavioral avoidance exhibited by many marine mammal species, it is very unlikely that a marine mammal would remain in the potential impact range of multiple sources or sequential events. Exposure to multiple stressors from multiple platforms is not likely to occur in the TMAA where the proposed activities are conducted in the open ocean and participating units are separated by large distances. In such cases, a behavioral reaction resulting in avoidance of the immediate vicinity of the activity would reduce the likelihood of exposure to additional stressors.

Secondly, a marine mammal could be exposed to multiple events over the duration of the Norther Edge Exercise; however, those events are generally separated in space and time in such a way that it would be unlikely that any individual marine mammal would be exposed to stressors from multiple activities within a short timeframe.

Multiple stressors may also have synergistic effects. For example, marine mammals that experience temporary hearing loss from acoustic stressors could in theory be more susceptible to physical strike and disturbance stressors via a decreased ability to detect and avoid threats. These cumulative, synergistic, and antagonistic interactions between multiple stressors both natural and anthropogenic, have just begun to be investigated and the exact mechanisms each stressor contributes to individual fitness is poorly understood (Balmer et al., 2019; Murray et al., 2020; National Academies of Sciences Engineering and Medicine, 2017; National Marine Fisheries Service, 2018a). Based on current best available science, the effects of multiple synergistic stressors over time cannot be realistically or precisely modeled for marine mammals. The Navy's quantitative and qualitative analyses are consistently conservative and likely over-predict impacts on marine mammals.

Research and monitoring efforts have included before, during, and after-event observations and surveys, data collection through conducting long-term studies in areas of Navy activity, occurrence surveys over large geographic areas, biopsy of animals occurring in areas of Navy activity, and tagging studies where animals are exposed to Navy stressors. These efforts are intended to contribute to the overall understanding of what impacts may be occurring overall to animals in these areas. To date, the findings from the research and monitoring efforts and the regulatory conclusions from previous analyses by NMFS for the TMAA (National Marine Fisheries Service, 2017b, 2017c), have been that the majority of impacts from Navy activities are not expected to have deleterious impacts on the fitness of any

individuals or long-term consequences to populations of marine mammals, and are not likely to jeopardize listed species or destroy or adversely modify critical habitat.

#### **3.8.4.1 Summary of Monitoring and Observations During Navy Activities Since 2016**

This section summarizes the information provided in detail in the 2016 GOA Final SEIS/OEIS (Section 3.8.5, Summary of Monitoring and Observations During Navy Activities). The results of previous monitoring and research since 2006 taking place in and around Navy ranges and occurring before, during, and after navy training and testing events, has been included as part of the Navy analyses as well as the prior analyses by NMFS in their MMPA authorization (National Marine Fisheries Service, 2017c), and the Biological Opinion for the 2016 GOA Final SEIS/OEIS (National Marine Fisheries Service, 2017a). As noted previously in the introductory paragraphs in Section 3.8.3 (Environmental Consequences), these reporting, monitoring, and research efforts from locations across the Pacific and in the Atlantic have added to the baseline data for understanding potential impacts to marine mammals in general. Given that this record involves many of the same Navy training activities being considered for the TMAA, and includes all the marine mammal taxonomic families present in the TMAA, many of the same species, and some of the same populations as they seasonally migrate from other range complexes, that compendium of Navy reporting is directly applicable to the TMAA. In addition, subsequent research and monitoring has continued to broaden the sample of observations regarding the general health of marine mammal populations in locations where the Navy has been conducting training and testing activities for decades. All available and applicable scientific findings have been considered in the analysis of marine mammal impacts presented in this SEIS/OEIS.

It has long been recognized that even when multiple years of marine mammal survey data are available for analysis, the ability for researchers to assess the magnitude and direction of trends in the abundance of individual marine mammal populations is often limited (Forney, 2000; Forney et al., 1991; Gerrodette, 1987; Moore & Barlow, 2017; Moore & Barlow, 2014; Taylor et al., 2007). For example, even for waters off the U.S. West Coast that have relatively good survey coverage over decades, it cannot be conclusively determined if the sperm whale population in this region is increasing, decreasing, or has remained static (Moore & Barlow, 2017). Additional types of information must therefore be considered when assessing the likely impacts of Navy activities on marine mammal populations.

Since 2006, the Navy, non-Navy marine mammal scientists, and research groups and institutions have conducted scientific monitoring and research in and around ocean areas in the Atlantic and Pacific where the Navy has been and proposes to continue testing and training. There have been three rounds of analysis for Navy training and testing activities at sea in various Navy range complexes in the Pacific by the Navy and NMFS (see for example 83 FR 66846, December 27, 2018). Data collected from Navy monitoring, scientific research findings, and annual reports have been provided to NMFS, and this public<sup>5</sup> record is an important contribution to the analysis of potential impacts on marine mammals. For example, these data provide information relevant to species distribution, habitat use, and evaluation of potential responses to Navy activities.

Monitoring is performed using a variety of methods, including visual surveys from surface vessels and aircraft, as well as passive acoustics before, during, and after Navy activities have been conducted. The

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<sup>5</sup> Navy monitoring reports are available at the Navy website ([www.navy.mil/speciesmonitoring.us/](http://www.navy.mil/speciesmonitoring.us/)) and also at the NMFS website ([www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities](http://www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities)).

Navy also has continued to contribute to funding of basic research, including behavioral response studies specifically designed to determine the effects to marine mammals from the Navy's main mid-frequency surface ship anti-submarine warfare sonar and other acoustic sources of potential impact.

The Navy and NMFS have coordinated the allocation of research across the Navy, with the intention of putting funding resources and efforts where they will be most effective. As a result, the majority of the Navy's monitoring and research effort has been focused on those other locations where Navy activities are more numerous and generally more intense. Research funded by the Navy in the TMAA since 2015 has included passive acoustic data collected using an unmanned glider, maintenance and data collection from five passive acoustic monitoring sites, and the collection and analysis of data from those sites located within the TMAA, and a line transect survey to improve our knowledge with regards to cetacean abundance and distribution in the TMAA (Klinck et al., 2016; Rice et al., 2018b; Rice et al., 2020; Rone et al., 2017; Wiggins et al., 2017; Wiggins & Hildebrand, 2018).

The training activities Navy is proposing for the TMAA are similar if not nearly identical to activities that have been occurring in Alaska waters for decades. Training in the TMAA is, by comparison to other Navy areas, less frequent and is in general smaller in scope than in other locations, so as a result the majority of the Navy's research effort has been focused elsewhere. For this reason, the vast majority of scientific field work, research, and monitoring efforts in the Pacific have been expended in the SOCAL Range Complex and Hawaii, where Navy training and testing activities have been more concentrated. In addition and since 2006, the Navy has been submitting exercise reports and monitoring reports to NMFS for the Navy's range complexes in the Pacific and the Atlantic. These publicly available exercise reports, monitoring reports, and the associated research findings have been integrated into adaptive management decisions regarding the focus for subsequent research and monitoring as determined in collaborations between Navy, NMFS, Marine Mammal Commission, and other marine resource subject matter experts. For example, see the 2019 U.S. Navy Annual Marine Species Monitoring Report for the Pacific that was made available to the public in April 2020 (U.S. Department of the Navy, 2020a).

To summarize what was presented in detail in Section 3.8.2 (Affected Environment), the evidence from reporting, monitoring, and research over more than a decade indicates that while the Proposed Action will result in the incidental harassment of marine mammals and may include auditory injury to some individuals, these impacts are expected to be inconsequential at the level of their marine mammal populations. It therefore remains that based on the best available science, including data developed in exercise and monitoring reports submitted to NMFS for over a decade, that long-term consequences for marine mammal populations are unlikely to result from Navy activities in the TMAA.

### **3.8.5 Endangered Species Act Determinations**

As part of the analysis in this SEIS/OEIS, the Navy has considered the prior analyses from the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS as reviewed and amended by this SEIS/OEIS, the findings from the 2017 NMFS Biological Opinion (National Marine Fisheries Service, 2017a), and the USFWS determinations made in consultation with Navy (U.S. Fish and Wildlife Service, 2011b) where they remain valid. The Navy is again consulting under section 7 of the ESA with NMFS and USFWS for the ESA-listed marine mammals that may be effected by the Proposed Action.(National Marine Fisheries Service, 2017a; U.S. Fish and Wildlife Service, 2011b) As noted in this SEIS/OEIS previously, there are no new Navy training activities in the TMAA that have not been previously considered in the TMAA or elsewhere where Navy trains. There have been no new ESA-listed marine mammal species and no newly designated critical habitat in the TMAA. Critical habitat has been proposed for ESA-listed humpback

whales along the Pacific coast of the U. S. (84 FR 54354; 9 October 2019), which partially overlaps the inshore portion of the TMAA, and the analysis of potential impacts to that habitat and the identified essential feature of that habitat have been considered using information available regarding that habitat (National Marine Fisheries Service, 2019b, 2019c) as detailed in prior sections of this SEIS/OEIS (see Section 3.8.3, Environmental Consequences).

Pursuant to the ESA, the Navy has determined that the continuation of the Navy's activities in the TMAA may affect the North Pacific right whale, blue whale, fin whale, Western North Pacific gray whale, Mexico DPS humpback whale, Western North Pacific DPS humpback whale, sei whale, sperm whale, Western DPS Steller sea lion and Northern sea otter. The Navy will have consulted with NMFS and USFWS as required by section 7(a)(2) of the ESA for these listed species. The Navy has also determined that Navy activities in the TMAA may affect the proposed humpback whale critical habitat.

### **3.8.6 Marine Mammal Protection Act Determinations**

The Navy is seeking a Letter of Authorization in accordance with the MMPA from NMFS for the use of certain stressors (the use of sonar and other transducers and explosives), as described under the Preferred Alternative (Alternative 1). The use of sonar and other transducers may result in Level A and Level B harassment of certain marine mammals. The use of explosives may result in Level A harassment and Level B harassment of certain marine mammals. The acoustic modeling Refer to Section 3.8.3.1.2 (Impacts from Sonar and Other Transducers) for details on the estimated impacts from sonar and other transducers, and Section 3.8.3.2.2 (Impacts from Explosives) for impacts from explosives. The estimated acoustic effects on marine mammals were modeled consistent with recent Navy analyses (U.S. Department of the Navy, 2017a, 2018c) and with recent regulations promulgated by NMFS (83 FR 66846, December 27, 2018). The modeling results indicate that non-auditory injury (i.e., lung or digestive tract injuries) or mortality should not be expected to result from the proposed testing and training activities under any of the alternatives. The only injury effects expected are PTSs (i.e., permanent damage to cells in the ear associated with hearing), resulting in Level A harassment as defined under the MMPA.

Based on the previous analyses for the same actions in TMAA as presented in the 2011 GOA Final EIS/OEIS and the 2016 GOA SEIS/OEIS, consistent with the current MMPA authorization for Navy training and testing in the TMAA (National Marine Fisheries Service, 2017c), and consistent with recent determinations for the same activities in other locations where Navy trains and tests,<sup>6</sup> the Navy has determined that weapon noise, vessel noise, aircraft noise, the use of in-water electromagnetic devices, in-air electromagnetic devices, high-energy lasers, vessel strike, in-water devices, seafloor devices, wires and cables, decelerators/parachutes, biodegradable polymers, and military expended materials are not expected to result in mortality or Level A or Level B harassment of any marine mammals.

#### **3.8.6.1 Summary of Science in the Temporary Maritime Activities Area by the Navy Related to Potential Effects on Marine Mammals Since 2006**

It has long been recognized that even when multiple years of marine mammal survey data are available for analysis, the ability for researchers to assess the magnitude and direction of trends in the abundance of individual marine mammal populations is often limited (Forney, 2000; Forney et al., 1991; Gerrodette, 1987; Moore & Barlow, 2017; Moore & Barlow, 2014; Taylor et al., 2007). For example, even for waters

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<sup>6</sup> Conclusions in this regard refer to the findings reached by the Navy and NMFS for many of the same actions in Southern California and Hawaii (FR 83[247]: 66846-67031; December 27, 2018).

off the U.S. West Coast that have relatively good survey coverage over decades, it cannot be conclusively determined if the sperm whale population in this region is increasing, decreasing, or has remained static (Moore & Barlow, 2017). Additional types of information must therefore be considered when assessing the likely impacts of Navy activities on marine mammal populations.

Since 2006, the Navy, non-Navy marine mammal scientists, and research groups and institutions have conducted scientific monitoring and research in and around ocean areas in the Atlantic and Pacific where the Navy has been and proposes to continue testing and training. There have been three rounds of analysis for Navy training and testing activities at sea in various Navy range complexes in the Pacific by the Navy and NMFS (see for example 83 FR 66846, December 27, 2018). Data collected from Navy monitoring, scientific research findings, and annual reports have been provided to NMFS, and this public<sup>7</sup> record is an important contribution to the analysis of potential impacts on marine mammals. For example, these data provide information relevant to species distribution, habitat use, and evaluation of potential responses to Navy activities.

Monitoring is performed using a variety of methods, including visual surveys from surface vessels and aircraft, as well as passive acoustics before, during, and after Navy activities have been conducted. The Navy also has continued to contribute to funding of basic research, including behavioral response studies specifically designed to determine the effects to marine mammals from the Navy's main mid-frequency surface ship anti-submarine warfare sonar and other acoustic sources of potential impact.

The Navy and NMFS have coordinated the allocation of research across the Navy, with the intention of putting funding resources and efforts where they will be most effective. As a result, the majority of the Navy's monitoring and research effort has been focused on those other locations where Navy activities are more numerous and generally more intense. Research funded by the Navy in the TMAA since 2015 has included passive acoustic data collected using an unmanned glider, maintenance of five passive acoustic monitoring sites, and the collection and analysis of data from those sites located within the TMAA (Klinck et al., 2016; Rice et al., 2018b; Wiggins et al., 2017; Wiggins & Hildebrand, 2018).

The training activities Navy is proposing for the TMAA are similar if not nearly identical to activities that have been occurring in Alaska waters for decades. Training in the TMAA are, by comparison to other Navy areas, less frequent and are in general small in scope, so as a result the majority of the Navy's research effort has been focused elsewhere. For this reason, the vast majority of scientific field work, research, and monitoring efforts have been expended in the SOCAL Range Complex and Hawaii, where Navy training and testing activities have been more concentrated. Since 2006, the Navy has been submitting exercise reports and monitoring reports to NMFS for the Navy's range complexes in the Pacific and the Atlantic. These publicly available exercise reports, monitoring reports, and the associated research findings have been integrated into adaptive management decisions regarding the focus for subsequent research and monitoring as determined in collaborations between Navy, NMFS, Marine Mammal Commission, and other marine resource subject matter experts using an adaptive

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<sup>7</sup> Navy monitoring reports are available at the Navy website ([www.navy.mil/speciesmonitoring.us/](http://www.navy.mil/speciesmonitoring.us/)) and also at the NMFS website ([www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities](http://www.fisheries.noaa.gov/national/marine-mammal-protection/incidental-take-authorizations-military-readiness-activities)).

management approach. For example, see the 2017 U.S. Navy Annual Marine Species Monitoring Report for the Pacific that was made available to the public in April 2018 (U.S. Department of the Navy, 2018a).

As noted previously in Section 3.8.2 (Affected Environment), these reporting, monitoring, and research efforts from locations across the Pacific and in the Atlantic have added to the baseline data for marine mammals inhabiting the TMAA. In addition, subsequent research and monitoring has continued to broaden the sample of observations regarding the general health of marine mammal populations in locations where the Navy has been conducting training and testing activities for decades. All available and applicable scientific findings have been considered in the analysis of marine mammal impacts presented in this SEIS/OEIS. Given that this record involves many of the same Navy training activities being considered for the TMAA, and includes all the marine mammal taxonomic families present in the TMAA, many of the same species, and some of the same populations as they seasonally migrate from other range complexes, that compendium of Navy reporting is directly applicable to the TMAA.

To summarize what was presented in detail in Section 3.8.2 (Affected Environment), the evidence from reporting, monitoring, and research over more than a decade indicates that while the Proposed Action will result in the incidental harassment of marine mammals and may include auditory injury to some individuals, these impacts are expected to be inconsequential at the level of their marine mammal populations. It therefore remains that based on the best available science, including data developed in exercise and monitoring reports submitted to NMFS for over a decade, that long-term consequences for marine mammal populations are unlikely to result from Navy activities in the TMAA.

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