3.9 Fishes

Supplemental Environmental Impact Statement/

Overseas Environmental Impact Statement

Mariana Islands Training and Testing

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3.9.1 Affected Environment

The purpose of this section is to supplement the analysis of impacts on fishes presented in the 2015 Mariana Islands Training and Testing (MITT) Final Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS) with new information relevant to proposed changes in training and testing activities conducted at sea and on Farallon de Medinilla. New information made available since the publication of the 2015 MITT Final EIS/OEIS is included below to better understand potential stressors and impacts on fishes resulting from training and testing activities. Comments received from the public during scoping related to fishes are addressed in Section 3.9.3 (Public Comments). Comments received from the public during the Draft Supplemental EIS (SEIS)/OEIS commenting period related to fishes are addressed in Appendix K (Public Comment Responses).

3.9.1.1 Hearing and Vocalization

A summary of fish hearing and vocalizations is described in the 2015 MITT Final EIS/OEIS. Due to the availability of new literature, including revised sound exposure criteria, the information provided below will supplement the 2015 MITT Final EIS/OEIS for fishes.

All fishes have two sensory systems that can detect sound in the water: the inner ear, which functions similarly to the inner ear in other vertebrates, and the lateral line, which consists of a series of receptors along the body of a fish (Popper, 2008). The lateral line system is sensitive to external particle motion arising from sources within a few body lengths of an animal. The lateral line detects particle motion at low frequencies from below 1 hertz (Hz) up to at least 400 Hz (Coombs & Montgomery, 1999; Hastings & Popper, 2005; Higgs & Radford, 2013; Webb et al., 2008). Generally, the inner ears of fish contain three types of dense otoliths (i.e., small calcareous bodies) that sit atop many delicate mechanoelectric hair cells within the inner ear of fishes, similar to the hair cells found in the mammalian ear. Underwater sound waves pass through the fish's body and vibrate the otoliths. This causes a relative motion between the dense otoliths and the surrounding tissues, causing a deflection of the hair cells, which is sensed by the nervous system.

Although a propagating sound wave contains pressure and particle motion components, particle motion is most significant at low frequencies (up to at least 400 Hz) and is most detectible at high sound pressures or very close to a sound source. The inner ears of fishes are directly sensitive to acoustic particle motion rather than acoustic pressure (acoustic particle motion and acoustic pressure are discussed in Appendix H, Acoustic and Explosive Concepts). Historically, studies that have investigated hearing in, and effects to, fishes have been carried out with sound pressure metrics. Although particle motion may be the more relevant exposure metric for many fish species, there is little data available that actually measures it due to a lack in standard measurement methodology and experience with particle motion detectors (Hawkins et al., 2015; Martin et al., 2016). In these instances, particle motion can be estimated from pressure measurements (Nedelec et al., 2016a).

Some fishes possess additional morphological adaptations or specializations that can enhance their sensitivity to sound pressure, such as a gas-filled swim bladder (Astrup, 1999; Popper & Fay, 2010). The swim bladder can enhance sound detection by converting acoustic pressure into localized particle motion, which may then be detected by the inner ear (Radford et al., 2012). Fishes with a swim bladder generally have greater hearing sensitivity and can detect higher frequencies than fishes without a swim bladder (Popper & Fay, 2010; Popper et al., 2014). In addition, structures such as gas-filled bubbles near the ear or swim bladder, or even connections between the swim bladder and the inner ear, also increase sensitivity and allow for high-frequency hearing capabilities and better sound pressure detection.

Although many researchers have investigated hearing and vocalizations in fish species (Ladich & Fay, 2013; Popper et al., 2014), hearing capability data only exist for just over 100 of the currently known 34,000 marine and freshwater fish species (Eschmeyer & Fong, 2017). Therefore, fish hearing groups are defined by species that possess a similar continuum of anatomical features, which result in varying degrees of hearing sensitivity (Popper & Fay, 2010). Categories and descriptions of hearing sensitivities are further defined in this document (modified from Popper et al., 2014) as the following:

- Fishes without a swim bladder—hearing capabilities are limited to particle motion detection at frequencies well below 2 kilohertz (kHz).
- Fishes with a swim bladder not involved in hearing—species lack notable anatomical specializations and primarily detect particle motion at frequencies below 2 kHz.
- Fishes with a swim bladder involved in hearing—species can detect frequencies below 2 kHz and possess anatomical specializations to enhance hearing, and are capable of sound pressure detection up to a few kHz.
- Fishes with a swim bladder and high-frequency hearing—species can detect frequencies below 2 kHz and possess anatomical specializations, and are capable of sound pressure detection at frequencies up to 10 kHz to over 100 kHz.

Data suggest that most species of marine fish either lack a swim bladder (e.g., sharks and flatfishes) or have a swim bladder not involved in hearing and can only detect sounds below 1 kHz. Some marine fishes (Clupeiformes) with a swim bladder involved in hearing are able to detect sounds to about 4 kHz (Colleye et al., 2016; Mann et al., 2001; Mann et al., 1997). One subfamily of clupeids (i.e., Alosinae) can detect high- and very high-frequency sounds (i.e., frequencies from 10 to 100 kHz, and frequencies above 100 kHz, respectively), although auditory thresholds at these higher frequencies are elevated and the range of best hearing is still in the low-frequency range (below 1 kHz) similar to other fishes. Mann et al. (1997; 1998) theorize that this subfamily may have evolved the ability to hear relatively high sound levels at these higher frequencies in order to detect echolocations of nearby foraging dolphins. For fishes that have not had their hearing tested, such as deep sea fishes, the suspected hearing capabilities are based on the structure of the ear, the relationship between the ear and the swim bladder, and other potential adaptations such as the presence of highly developed areas of the brain related to inner ear and lateral line functions (Buran et al., 2005; Deng et al., 2011, 2013). It is believed that most fishes have their best hearing sensitivity from 100 to 400 Hz (Popper, 2003).

Endangered Species Act (ESA)-listed species with the potential to occur within the MITT Study Area hammerhead shark (*Sphyrna lewini*), the oceanic whitetip shark (*Carcharhinus longimanus*), and the giant manta ray (*Manta birostris*). As discussed above, most marine fishes investigated to date lack hearing capabilities greater than 1,000 Hz. Rays and sharks are cartilaginous fishes (i.e., elasmobranchs) lacking a swim bladder. Available data suggest these species can detect sounds from 20 to 1,000 Hz, with best sensitivity at lower ranges (Casper et al., 2003; Casper & Mann, 2006; Casper & Mann, 2009; Myrberg, 2001).

Some fishes are known to produce sound. Bony fishes can produce sounds in a number of ways and use them for a number of behavioral functions (Ladich, 2008, 2014). Over 30 families of fishes are known to use vocalizations in aggressive interactions, and over 20 families are known to use vocalizations in mating (Ladich, 2008). Sounds generated by fishes as a means of communication are generally below

500 Hz (Slabbekoorn et al., 2010). The air in the swim bladder is vibrated by the sound producing structures (often muscles that are integral to the swim bladder wall) and radiates sound into the water (Zelick et al., 1999). Sprague and Luczkovich (2004) calculated that silver perch, of the family sciaenidae, can produce drumming sounds ranging from 128 to 135 decibels (dB) referenced to 1 micropascal (dB re 1 µPa). Female midshipman fish apparently detect and locate the "hums" (approximately 90–400 Hz) of vocalizing males during the breeding season (McIver et al., 2014; Sisneros & Bass, 2003). Sciaenids produce a variety of sounds, including calls produced by males on breeding grounds (Ramcharitar et al., 2001), and a "drumming" call produced during chorusing that suggests a seasonal pattern to reproductive-related function (McCauley & Cato, 2000). Other sounds produced by chorusing reef fishes include "popping," "banging," and "trumpet" sounds; altogether, these choruses produce sound levels 35 dB above background levels, at peak frequencies between 250 and 1,200 Hz, and source levels between 144 and 157 dB re 1 µPa (McCauley & Cato, 2000).

Additional research using visual surveys (such as baited underwater video) and passive acoustic monitoring continue to reveal new sounds produced by fishes, both in the marine and freshwater environments, and allow for specific behaviors to be paired with those sounds (Radford et al., 2018; Rountree et al., 2018; Rowell et al., 2018).

3.9.1.2 General Threats

A summary of the major threats to fish species within the Study Area is described in the 2015 MITT Final EIS/OEIS. Overfishing and associated factors, such as bycatch, fisheries-induced evolution, and intrinsic vulnerability to overfishing were described. Pollution, including the effect of oceanic circulation patterns scattering coastal pollution throughout the open ocean, was described. The effects of organic and inorganic pollutants to fishes, including bioaccumulation of pollutants, behavioral and physiological changes, or genetic damage, were described, as well as entanglement in abandoned commercial and recreational fishing gear. Other human-caused stressors on fishes described were the introduction of non-native species, climate change, aquaculture, energy production, vessel movement, and underwater noise. Neither the extent or any other threats have changed since it was last described in the 2015 MITT Final EIS/OEIS. Therefore, the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.1.3 Endangered Species Act Species

The scalloped hammerhead shark (*Sphyrna lewini*), oceanic whitetip shark (*Carcharhinus longimanus*), and giant manta ray (*Manta birostris*) are the only ESA-listed fish species in the Study Area (Table 3.9-1). Two species of concern, the humphead wrasse (*Cheilinus undulates)* and bumphead parrotfish (*Bolbometopon muricatum*), also occur in the Study Area (Table 3.9-1). The National Marine Fisheries Service (NMFS) has some concerns regarding status and threats for species of concern, but insufficient information is available to indicate a need to list the species under the ESA. Species of concern status does not carry any procedural or substantive protections under the ESA. All the species listed in Table 3.9-1 are declining because of impacts from fishing (including night spear fishing, bycatch, and illegal fishing activities) and habitat degradation.

Table 3.9-1: Endangered Species Act Listed and Special Status Fish Species in the Mariana Islands Study Area

Species Name and Regulatory Status			Presence in Study Area	
Common Name	**Scientific Name**	**Endangered Species Act Status**	**Open Ocean/ Transit Corridor**	**Coastal Ocean**
Scalloped hammerhead shark (Indo-West Pacific Distinct Population Segment)	*Sphyrna lewini*	Threatened	Yes	Yes
Oceanic whitetip shark	*Carcharhinus longimanus*	Threatened	Yes	Yes
Giant manta ray	*Manta birostris*	Threatened	Yes	Yes
Humphead wrasse	*Cheilinus undulatus*	Species of Concern	No	Yes
Bumphead parrotfish	*Bolbometopon muricatum*	Species of Concern	No	Yes

3.9.1.3.1 Scalloped Hammerhead Shark (*Sphyrna lewini*)

A literature review found that the information on the scalloped hammerhead shark in the Study Area has not substantially changed from what is included in the 2015 MITT Final EIS/OEIS. Therefore, the information presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.1.3.1.1 Status and Management

In 2013, NMFS determined that two distinct population segments, the Central and Southwest Atlantic and Indo-West Pacific, warrant listing as threatened. The Indo-West Pacific distinct population segment is the only one located within the Study Area. Following a review of recent literature, the status and management of this species has not changed since the publication of the 2015 MITT Final EIS/OEIS. As such, the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid. No critical habitat has been designated for this species.

3.9.1.3.1.2 Habitat and Geographic Range

The habitat and geographic range of scalloped hammerhead sharks is described in the 2015 MITT Final EIS/OEIS. Following a review of recent literature, information related to habitat and the geographic range of this species has not changed since the publication of the 2015 MITT Final EIS/OEIS. As such, the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.1.3.1.3 Population and Abundance

As indicated in the 2015 MITT Final EIS/OEIS, information on population and abundance of scalloped hammerhead sharks is limited. Following a review of recent literature, information related to population and abundance estimates for this species has not changed since the publication of the 2015 MITT Final EIS/OEIS. As such, the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.1.3.1.4 Predator and Prey Interactions

A new study by Brown et al. (2016) found that juvenile scalloped hammerhead sharks in the Rewa River estuary on Fiji consumed primarily estuarine and marine prawns, stomatopoda (mantis shrimps), estuarine eels, and various bony fish, with prawns being found in half of the stomachs sampled, which is consistent with other available information. However, this new information does not appreciably change the information and analysis that was presented in the 2015 MITT Final EIS/OEIS.

3.9.1.3.1.5 Species-Specific Threats

Primary threats to scalloped hammerhead sharks are from direct take, especially by the foreign commercial shark fin fishery (National Marine Fisheries Service, 2011), as described in the 2015 MITT Final EIS/OEIS. Following a review of recent literature, information on threats to this species has not changed since the publication of the 2015 MITT Final EIS/OEIS. As such, the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.1.3.2 Oceanic Whitetip Shark (*Carcharhinus longimanus*)

3.9.1.3.2.1 Status and Management

Since the publication of the 2015 MITT Final EIS/OEIS, NMFS proposed on December 29, 2016 to list the oceanic whitetip shark as a threatened species under the ESA (81 Federal Register [FR] 96304). On January 30, 2018, NMFS published the Final Rule listing this species as threatened and concluded that critical habitat is not determinable because data sufficient to perform the required analyses are lacking (83 FR 4153). Because this species was proposed as threatened, and subsequently listed as threatened under the ESA after the publication of the 2015 MITT Final EIS/OEIS, the impact analysis included in Section 3.9.2 (Environmental Consequences) is new.

3.9.1.3.2.2 Habitat and Geographic Range

Oceanic whitetip sharks are found worldwide in warm tropical and subtropical waters between 30° North and 35° South latitude near the surface of the water column (Young et al., 2016). Oceanic whitetips occur throughout the Central Pacific. This species has a clear preference for open ocean waters, with abundances decreasing with greater proximity to continental shelves. Preferring warm waters near or over 20°Celsius (68°Fahrenheit), and offshore areas, the oceanic whitetip shark is known to undertake seasonal movements to higher latitudes in the summer (National Oceanic and Atmospheric Administration, 2016b) and may regularly survey extreme environments (deep depths, low temperatures) as a foraging strategy (Young et al., 2016).

3.9.1.3.2.3 Population and Abundance

Population trend information is not clear or available. Information shows that the population has declined and there is evidence of decreasing average weights of the sharks. Unstandardized nominal catch data from the Inter-American Tropical Tuna Commission in the eastern Pacific tropical tuna purse seine fisheries show trends of decreasing catch (Inter-American Tropical Tuna Commission, 2015). In addition, Rice & Harvey (2012) found catch, catch per unit effort, and size composition data for oceanic whitetip sharks in the western and central Pacific all show consistent declines.

3.9.1.3.2.4 Predator and Prey Interactions

Oceanic whitetip sharks are large, often reaching a maximum length of 345 centimeters (cm) (Ebert et al., 2015), can live up to nine years (Joung et al., 2016), and are one of the major apex predators in the tropical open ocean waters. This species feeds on fishes, stingrays, sea turtles, birds, and cephalopods, and has no known predators.

3.9.1.3.2.5 Species-Specific Threats

Threats include pelagic longline and drift net fisheries bycatch, targeted fisheries (for the shark fin trade), and destruction or modification of its habitat and range (Baum et al., 2015; Defenders of Wildlife, 2015b). Legal and illegal fishing activities have caused significant population declines for the oceanic whitetip shark caught as bycatch in tuna and swordfish longlines throughout its range.

3.9.1.3.3 Giant Manta Ray (*Manta birostris*)

3.9.1.3.3.1 Status and Management

Since the publication of the 2015 MITT Final EIS/OEIS, NMFS proposed on January 12, 2017 to list the giant manta ray as a threatened species under ESA (82 FR 3694). Based on the best scientific and commercial information available, including the status review report (Miller & Klimovich, 2016), and after taking into account efforts being made to protect these species, NMFS determined that the giant manta ray is likely to become an endangered species within the foreseeable future throughout a significant portion of its range. On January 22, 2018, NMFS published the Final Rule listing this species as threatened and concluded that critical habitat was not determinable because data sufficient to perform the required analyses are lacking (83 FR 2916). Because this species was proposed as threatened and subsequently listed as threatened under the ESA after the publication of the 2015 MITT Final EIS/OEIS, the impact analysis presented below in Section 3.9.2 (Environmental Consequences) is new.

3.9.1.3.3.2 Habitat and Geographic Range

Giant manta rays are visitors to productive coastlines with regular upwelling, including oceanic island shores, and offshore pinnacles and seamounts. They utilize sandy bottom habitat and seagrass beds, as well as shallow reefs, and the ocean surface both inshore and offshore. The species ranges globally and is distributed in tropical, subtropical, and temperate waters. They migrate seasonally usually more than 1,000 kilometers (km) (621.4 miles), however not likely across ocean basins (National Oceanic and Atmospheric Administration, 2016a).

3.9.1.3.3.3 Population and Abundance

No stock assessments exist for the giant manta ray. Most estimates of subpopulations are based on anecdotal observations by divers and fishermen, with current populations estimated between 100 and 1,500 individuals (Miller & Klimovich, 2016). In general, giant manta ray populations have declined, except in areas where they are specifically protected, such as the Hawaiian Islands (National Oceanic and Atmospheric Administration, 2016a). Giant manta rays reach maturity at age 10 and have one pup every two to three years (National Oceanic and Atmospheric Administration, 2016a).

3.9.1.3.3.4 Predator and Prey Interactions

Manta rays prey exclusively on plankton (Defenders of Wildlife, 2015a). The gill plates of the giant manta ray filters the water as they swim, straining out any plankton that is larger than a grain of sand (Defenders of Wildlife, 2015a).

3.9.1.3.3.5 Species-Specific Threats

Threats to giant manta rays include fisheries and bycatch, destruction or modification of habitat, and disease and predation. The international market highly values the gill plates of the giant manta ray for use in traditional medicines. They also trade their cartilage and skins and consume the manta ray meat or use it for local bait. Bycatch occurs in purse seine, gillnet, and trawl fisheries as well (National Oceanic and Atmospheric Administration, 2016a). Fisheries exist outside the Study Area in Indonesia, Sri Lanka, India, Peru, Mexico, China, Mozambique, and Ghana (Food and Agriculture Organization of the United Nations, 2013). Other potential threats include degradation of coral reefs, interaction with marine debris, marine pollution, and boat strikes (Food and Agriculture Organization of the United Nations, 2013).

3.9.1.4 Federally Managed Species

The Magnuson-Stevens Fishery Conservation and Management Act (MSA) (see Section 3.0.1.1, Federal Statutes, in the 2015 MITT Final EIS/OEIS) established eight fishery management councils that share authority with NMFS to manage and conserve the fisheries in federal waters. Together with NMFS, the councils maintain fishery management plans for species or species groups to regulate commercial and recreational fishing within their geographic regions. The Study Area is under the jurisdiction of the Western Pacific Regional Fishery Management Council. Sections 3.3 (Marine Habitats), 3.7 (Marine Vegetation), and 3.8 (Marine Invertebrates) analyze impacts on habitats within the Study Area.

The Mariana Archipelago Fishery Ecosystem Plan (FEP), which includes fishery management measures for Guam and the Commonwealth of the Northern Mariana Islands, was approved in 2009 and codified in 2010. The Western Pacific Regional Fishery Management Council is currently working on an update to the FEP (Western Pacific Regional Fishery Management Council, 2016). Federally managed fish species listed in the 2015 MITT Final EIS/OEIS and in Table 3.9-2 have not changed since the publication of the EIS/OEIS and the information and analysis presented in the 2015 MITT Final EIS/OEIS remains valid.

The 2015 NMFS stock assessment report for the bottomfish fishery in Guam and the Commonwealth of the Northern Mariana Islands (CNMI) concluded that the fishery was not overfished through 2013, and modeled projections predicted that the fishery was very unlikely to become overfished by 2017 (Yau et al., 2016). However, coral reef fisheries, which support most traditional fishing in the Study Area, have declined over the past 30 years (Weijerman et al., 2016). However, the catch from the non-commercial reef fish fishery in the CNMI, which supports most traditional fishing, has historically been underestimated, yet has clearly been in decline since the late 1970s (Cuetos-Bueno & Houk, 2014). Detailed information on overfished stocks is presented in Section 3.12.1.2 (Commercial and Recreational Fishing).

Table 3.9-2: Federally Managed Fish Species Within the Mariana Islands Study Area for Each Fishery Management Unit in the FEP

Western Pacific Regional Fishery Management Council	
Marianas Bottomfish Management Unit	
Common Name	**Scientific Name**
Amberjack	*Seriola dumerili*
Black trevally/jack	*Caranx lugubris*
Blacktip grouper	*Epinephelus fasciatus*
Blueline snapper	*Lutjanus kasmira*
Giant trevally/jack	*Caranx ignobilis*
Gray snapper	*Aprion virescens*
Lunartail grouper	*Variola louti*
Pink snapper	*Pristipomoides filamentosus*
Pink snapper	*Pristipomoides flavipinnis*
Red snapper/silvermouth	*Aphareus rutilans*
Red snapper/buninas agaga	*Etelis carbunculus*
Red snapper/buninas	*Etelis coruscans*
Redgill emperor	*Lethrinus rubrioperculatus*
Snapper	*Pristipomoides zonatus*
Yelloweye snapper	*Pristipomoides flavipinnis*
Yellowtail snapper	*Pristipomoides auricilla*
Marianas Coral Reef Ecosystem Management Unit	
Banded goatfish	*Parupeneus spp.*
Bantail goatfish	*Upeneus arge*
Barred flag-tail	*Kuhlia mugil*
Barred thicklip	*Hemigymnus fasciatus*
Bigeye	*Priacanthus hamrur*
Bigeye scad	*Selar crumenophthalmus*
Bignose unicornfish	*Naso vlamingii*
Bigscale soldierfish	*Myripristis berndti*
Black tongue unicornfish	*Naso hexacanthus*
Black triggerfish	*Melichthys niger*
Blackeye thicklip	*Hemigymnus melapterus*
Blackstreak surgeonfish	*Acanthurus nigricauda*
Blacktip reef shark	*Carcharhinus melanopterus*
Blotcheye soldierfish	*Myripristis murdjan*

Table 3.9-2: Federally Managed Fish Species Within the Mariana Islands Study Area for Each Fishery Management Unit in the FEP (continued)

Western Pacific Regional Fishery Management Council	
Marianas Coral Reef Ecosystem Management Unit (continued)	
Common Name	**Scientific Name**
Blue-banded surgeonfish	*Acanthurus lineatus*
Blue-lined squirrelfish	*Sargocentron tiere*
Bluespine unicornfish	*Naso unicornus*
Brick soldierfish	*Myripristis amaena*
Bronze soldierfish	*Myripristis adusta*
Cigar wrasse	*Cheilio inermis*
Clown triggerfish	*Balistoides conspicillum*
Convict tang	*Acanthurus triostegus*
Crown squirrelfish	*Sargocentron diadema*
Dash-dot goatfish	*Parupeneus barberinus*
Dogtooth tuna	*Gymnosarda unicolor*
Doublebar goatfish	*Parupeneus bifasciatus*
Engel's mullet	*Moolgarda engeli*
Floral wrasse	*Cheilinus chlorourus*
Forktail rabbitfish	*Siganus aregentus*
Fringelip mullet	*Crenimugil crenilabis*
Galapagos shark	*Carcharhinus galapagensis*
Giant moray eel	*Gymnothorax javanicus*
Glasseye	*Heteropriacanthus cruentatus*
Golden rabbitfish	*Siganus guttatus*
Gold-spot rabbitfish	*Siganus punctatissimus*
Gray unicornfish	*Naso caesius*
Great barracuda	*Sphyraena barracuda*
Grey reef shark	*Carcharhinus amblyrhynchos*
Heller's barracuda	*Sphyraena helleri*
Humphead parrotfish	*Bolbometopon muricatum*
Humpnose unicornfish	*Naso tuberosus*
Longface wrasse	*Hologynmosus doliatus*
Mackerel scad	*Decapterus macarellus*
Mimic surgeonfish	*Acanthurus pyroferus*
Multi-barred goatfish	*Parupeneus multifaciatus*
Napoleon wrasse	*Cheilinus undulates*

Table 3.9-2: Federally Managed Fish Species Within the Mariana Islands Study Area for Each Fishery Management Unit in the FEP (continued)

Western Pacific Regional Fishery Management Council	
Marianas Coral Reef Ecosystem Management Unit (continued)	
Common Name	**Scientific Name**
Orange-spot surgeonfish	*Acanthurus olivaceus*
Orangespine unicornfish	*Naso lituratus*
Orangestriped triggerfish	*Balistapus undulates*
Pacific longnose parrotfish	*Hipposcarus longiceps*
Parrotfish	*Scarus spp.*
Pearly soldierfish	*Myripristis kuntee*
Pinktail triggerfish	*Melichthys vidua*
Razor wrasse	*Xyrichtys pavo*
Red-breasted wrasse	*Cheilinus fasciatus*
Ring-tailed wrasse	*Oxycheilinus unifasciatus*
Ringtail surgeonfish	*Acanthurus blochii*
Rudderfish	*Kyphosus biggibus*
Rudderfish	*Kyphosus cinerascens*
Rudderfish	*Kyphosus vaigienses*
Saber or long jaw squirrelfish	*Sargocentron spiniferum*
Scarlet soldierfish	*Myripristis pralinia*
Scribbled rabbitfish	*Siganus spinus*
Side-spot goatfish	*Parupeneus pleurostigma*
Silvertip shark	*Carcharhinus albimarginatus*
Spotfin squirrelfish	*Neoniphon spp.*
Spotted unicornfish	*Naso brevirostris*
Stareye parrotfish	*Calotomus carolinus*
Striped bristletooth	*Ctenochaetus striatus*
Stripped mullet	*Mugil cephalus*
Surge wrasse	*Thalassoma purpureum*
Tailspot squirrelfish	*Sargocentron caudimaculatum*
Threadfin	*Polydactylus sexfilis*
Three-spot wrasee	*Halicoeres trimaculatus*
Titan triggerfish	*Balistoides viridescens*
Triple-tail wrasee	*Cheilinus trilobatus*
Twospot bristletooth	*Ctenochaetus binotatus*
Undulated moray eel	*Gymnothorax undulatus*
Vermiculate rabbitfish	*Siganus vermiculatus*

Table 3.9-2: Federally Managed Fish Species Within the Mariana Islands Study Area for Each Fishery Management Unit in the FEP (continued)

Western Pacific Regional Fishery Management Council	
Marianas Coral Reef Ecosystem Management Unit (continued)	
Common Name	**Scientific Name**
Violet soldierfish	*Myripristis violacea*
White-lined goatfish	*Parupeneus ciliatus*
White-spotted surgeonfish	*Acanthurus guttatus*
Whitebar surgeonfish	*Acanthurus leucopareius*
Whitecheek surgeonfish	*Acanthurus nigricans*
Whitemargin unicornfish	*Naso annulatus*
Whitepatch wrasse	*Xyrichtys aneitensis*
Whitetip reef shark	*Triaenodon obesus*
Whitetip soldierfish	*Myripristis vittata*
Yellow goatfish	*Mulloidichthys spp.*
Yellow tang	*Zebrasoma flavescens*
Yellowfin goatfish	*Mulloidichthys vanicolensis*
Yellowfin soldierfish	*Myripristis chryseres*
Yellowfin surgeonfish	*Acanthurus xanthopterus*
Yellowmargin moray eel	*Gymnothorax flavimarginatus*
Yellowsaddle goatfish	*Parupeneus cyclostomas*
Yellowstripe goatfish	*Mylloidichthys flaviolineatus*
Guam and Northern Mariana Islands Pelagic Fisheries	
Dogtooth tuna	*Gymnosarda unicolor*
Double-lined mackerel	*Grammatorcynus bilineatus*
Kawakawa	*Euthynnus affinis*
Mahi	*Coryphaena hippurus*
Oilfish	*Ruvettus pretiosus*
Pacific blue marlin	*Makaira mazara*
Rainbow runner	*Elagatis bipinnulatus*
Skipjack tuna	*Katsuwonus pelamis*
Wahoo	*Acanthocybium solandri*
Yellowfin tuna	*Thunnus albacares*

3.9.1.5 Taxonomic Group Descriptions

A literature review found that the information on the taxonomic groups of fishes in the Study Area has not substantially changed from what is included in the 2015 MITT Final EIS/OEIS. Therefore, the information presented in the 2015 MITT Final EIS/OEIS remains valid.

3.9.2 Environmental Consequences

In the Proposed Action for this SEIS/OEIS, there have been some modifications to the quantity and type of acoustic and explosive stressors under the two action alternatives. There are also additional species listed under the ESA that are considered. In addition, within the stressor framework presented in the

2015 MITT Final EIS/OEIS, high-energy lasers are being analyzed as a new energy stressor, as detailed in Section 3.0.4.3.2.2 (High-Energy Lasers).

The 2015 MITT Final EIS/OEIS considered training and testing activities that currently occur in the Study Area and considered all potential stressors related to fishes. The potential impacts on fishes in the Study Area from Navy training and testing activities is presented in detail for ESA-listed and federally managed species, as well as generally for taxonomic groups.

The stressors vary in intensity, frequency, duration, and location within the Study Area. The stressors applicable to fishes in the Study Area are the same stressors analyzed in the 2015 MITT Final EIS/OEIS and include:

- **Acoustic** (sonar and other transducers, vessel noise, aircraft noise, and weapon noise)
- **Explosives** (in-air explosions and in-water explosions)
- **Energy** (in-water electromagnetic devices and high-energy lasers)
- **Physical disturbance and strikes** (vessels, in-water devices, military expended materials, and seafloor devices)
- **Entanglement** (wires and cables, decelerators/parachutes)
- **Ingestion** (military expended materials – munitions, military expended materials other than munitions)
- **Secondary** (impacts associated with sediments and water quality)

This section evaluates how and to what degree potential impacts on fishes from stressors described in Section 3.0 (Introduction) may have changed since the analysis presented in the 2015 MITT Final EIS/OEIS was completed. Table 2.5-1 and Table 2.5-2 in Chapter 2 (Description of Proposed Action and Alternatives) list the proposed training and testing activities and include the number of times each activity would be conducted annually and the locations within the Study Area where the activity would typically occur under each alternative. The tables also present the same information for activities described in the 2015 MITT Final EIS/OEIS so that the proposed levels of training and testing under this SEIS/OEIS can be easily compared.

The Navy conducted a review of federal and state regulations and standards relevant to fishes and reviewed scientific literature published since 2015 for new information on fishes that could update the analysis presented in the 2015 MITT Final EIS/OEIS. The analysis presented in this section also considers standard operating procedures, which are discussed in Section 2.3.3 (Standard Operating Procedures) of this SEIS/OEIS, and mitigation measures that are described in Chapter 5 (Mitigation). The Navy would implement these measures to avoid or reduce potential impacts on fishes from stressors associated with the proposed training and testing activities. Mitigation for ESA-listed fishes will be coordinated with NMFS through the ESA consultation process.

3.9.2.1 Acoustic Stressors

The analysis of effects to fishes follows the concepts outlined in Section 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). This section begins with a summary of relevant data regarding acoustic impacts on fishes in Section 3.9.2.1.1 (Background). This is followed by an analysis of estimated impacts on fishes due to specific Navy acoustic stressors (sonar and other transducers, vessel noise, aircraft noise, and weapon noise). Additional explanations of the acoustic terms and sound energy concepts used in this section are found in Appendix H (Acoustic and Explosive Concepts).

The Navy will rely on the 2015 MITT Final EIS/OEIS analysis for the analysis of vessel noise and weapon noise, as there has been no substantive or otherwise meaningful change in the action, although new applicable and emergent science in regard to these sub-stressors is presented in the sections that follow. Due to available new literature, adjusted sound exposure criteria, and new acoustic effects modeling, the analysis provided in Section 3.9.2.1.2 (Impacts from Sonar and Other Transducers) and Section 3.9.2.1.4 (Impacts from Aircraft Noise) of this SEIS/OEIS supplants the 2015 MITT Final EIS/OEIS for fishes, and changes estimated impacts for some species since the 2015 MITT Final EIS/OEIS.

3.9.2.1.1 Background

Effects of human-generated sound on fishes have been examined in numerous publications (Hastings & Popper, 2005; Hawkins et al., 2015; Ladich & Popper, 2004; Lindseth & Lobel, 2018; Mann, 2016; Mickle & Higgs, 2018; National Research Council, 1994, 2003; Neenan et al., 2016; Popper et al., 2004; Popper & Hawkins, 2019; Popper, 2003, 2008; Popper & Hastings, 2009b; Popper et al., 2014; Popper et al., 2016; Popper & Hawkins, 2018). The potential impacts from Navy activities are based on the analysis of available literature related to each type of effect. Where applicable, thresholds and relative risk factors presented in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) were used to assist in the analysis of effects on fishes from Navy activities.

There are limited studies of fish responses to aircraft and weapon noise. Based on the general characteristics of these sound types, for stressors where data is lacking (such as aircraft noise), studies of the effects of similar non-impulsive/continuous noise sources (such as sonar or vessel noise) are used to inform the analysis of fish responses. Similarly, studies of the effects from impulsive sources (such as air guns or pile driving) are used to inform fish responses to other impulsive sources (such as weapon noise). Non-impulsive or continuous sources may be presented as a proxy source to better understand potential reactions from fish where data from sonar and vessel noise exposures are limited. Additional information on the acoustic characteristics of these sources can be found in Appendix H (Acoustic and Explosive Concepts).

3.9.2.1.1.1 Injury

Injury refers to the direct effects on the tissues or organs of a fish. Moderate- to low-level noise from vessels, aircraft, and weapons use are described in Section 3.0.4.1 (Acoustic Stressors) and lacks the amplitude and energy to cause any direct injury. Section 3.0.4.7 (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 Impulsive Sound Sources

Impulsive sounds, such as those produced by seismic air guns and impact pile driving, may cause injury or mortality in fishes. Although air guns and pile driving would not occur in the Study Area, this information aids in the analysis of other impulsive sources (i.e., weapons noise or in some cases, explosions). Mortality and potential damage to the cells of the lateral line have been observed in fish larvae, fry, and embryos after exposure to single shots from a seismic air gun within close proximity to the sound source (0.1 to 6 meters [m]) (Booman et al., 1996; Cox et al., 2012). However, exposure of adult fish to a single shot from an air gun array (four air guns) within similar ranges (6 m), has not resulted in any signs of mortality within seven days after exposure (Popper et al., 2016). Although injuries occurred in adult fishes, they were similar to injuries seen in control subjects (i.e., fishes that were not exposed to the air gun) so there is little evidence that the air gun exposure solely contributed to the observed effects.

Injuries, such as ruptured swim bladders, hematomas, and hemorrhaging of other gas-filled organs, have been reported in fish exposed to a large number of simulated impact pile driving strikes with cumulative sound exposure levels up to 219 decibels referenced to 1 micropascal squared seconds (dB re 1 µPa^2-s) under highly controlled settings where fish were unable to avoid the source (Casper et al., 2012b; Casper et al., 2013a; Casper et al., 2013b; Halvorsen et al., 2011; Halvorsen et al., 2012a; Halvorsen et al., 2012b). However, it is important to note that these studies exposed fish to 900 or more strikes, as the studies aimed to evaluate the equal energy hypothesis, which suggests that the effects of a large single pulse of energy is equivalent to the effects of energy received from many smaller pulses (as discussed in Smith & Gilley, 2008). Halvorsen et al. (2011) and Casper et al. (2017) found that the equal energy hypothesis does not apply to effects of pile driving; rather, metrics relevant to injury could include, but not be limited to, cumulative sound exposure level, single strike sound exposure level, and number of strikes (Halvorsen et al., 2011). Furthermore, Casper et al. (2017) found the amount of energy in each pile strike and the number of strikes determines the severity of the exposure and the injuries that may be observed. For example, hybrid striped bass (white bass *Morone chrysops* x striped bass *M. saxaltilis*) exposed to fewer strikes with higher single strike sound exposure values resulted in a higher number of, and more severe, injuries than bass exposed to an equivalent cumulative sound exposure level that contained more strikes with lower single strike sound exposure values. This is important to consider when comparing data from pile driving studies to potential effects from an explosion. Although single strike peak sound pressure levels were reported during these experiments (at average levels of 207 dB re 1 µPa), the injuries were only observed during exposures to multiple strikes, therefore, it is anticipated that a peak value much higher than the measured values would be required to lead to injury in fishes exposed to a single strike, or explosion.

These studies included species both with and without swim bladders. The majority of fish that exhibited injuries were those with swim bladders. Lake sturgeon (*Acipenser fulvescens*), a physostomous fish, was found to be less susceptible to injury from impulsive sources than Nile tilapia (*Oreochromis niloticus*) or hybrid striped bass, physoclistous fishes (Casper et al., 2017; Halvorsen et al., 2012a). As reported by Halvorsen et al. (2012a), the difference in results is likely due to the type of swim bladder in each fish. Physostomous fishes have an open duct connecting the swim bladder to their esophagus and may be able to quickly adjust the amount of gas in their body by gulping or releasing air. Physoclistous fishes do not have this duct; instead, gas pressure or glands regulate gas pressure in the swim bladder. There were no mortalities reported during these experiments, and in the studies where recovery was observed, the majority of exposure related injuries healed within a few days in a laboratory setting. In many of these controlled studies, neutral buoyancy was determined in the fishes prior to exposure to the simulated pile driving. However, fishes with similar physiology to those described in these studies that are exposed to actual pile driving activities may show varying levels of injury depending on their state of buoyancy.

By exposing caged juvenile European sea bass (*Dicentrarchus labrax*) to actual pile driving operations, Debusschere et al. (2014) largely confirmed the results discussed in the paragraph above. No differences in mortality were found between control and experimental groups at similar levels tested in the experiments described in the paragraph above (sound exposure levels up to 215–222 dB re 1 µPa^2-s), and many of the same types of injuries occurred (Casper et al., 2012b; Casper et al., 2013a; Casper et al., 2013b; Halvorsen et al., 2011; Halvorsen et al., 2012a; Halvorsen et al., 2012b). Fishes with injuries from impulsive sources such as these may not survive in the wild due to harsher conditions and risk of predation.

Other potential effects from exposure to impulsive sound sources include potential bubble formation and neurotrauma. It is speculated that high sound pressure levels may also cause bubbles to form from micronuclei in the blood stream or other tissues of animals, possibly causing embolism damage (Hastings & Popper, 2005). Fishes have small capillaries where these bubbles could be caught and lead to the rupturing of the capillaries and internal bleeding. It has also been speculated that this phenomena could take place in the eyes of fish due to potentially high gas saturation within the eye tissues (Popper & Hastings, 2009b). Additional research is necessary to verify if these speculations apply to exposures to non-impulsive sources such as sonars. These phenomena have not been well studied in fishes and are difficult to recreate under real-world conditions.

As summarized in the *ANSI Sound Exposure Guideli*ne technical report (Popper et al., 2014), exposure to high intensity and long duration impact pile driving or air gun shots did not cause mortality, and fishes typically recovered from injuries in controlled laboratory settings. Species tested to date can be used as viable surrogates for investigating injury in other species exposed to similar sources (Popper et al., 2014).

Injury due to Sonar and Other Transducers

Non-impulsive sound sources (e.g., sonar, acoustic modems, and sonobuoys) have not been known to cause direct injury or mortality to fish under conditions that would be found in the wild (Halvorsen et al., 2012a; Kane et al., 2010; Popper et al., 2007). Potential direct injuries (e.g., barotrauma, hemorrhage or rupture of organs or tissue) from non-impulsive sound sources, such as sonar, are unlikely because of slow rise times,[1] lack of a strong shock wave such as that associated with an explosive, and relatively low peak pressures. General categories and characteristics of Navy sonar systems are described in Section 3.0.4.1.1 (Sonar and Other Transducers).

The effects of mid-frequency sonar-like signals (1.5–6.5 kHz) on larval and juvenile Atlantic herring (*Clupea harengus*), Atlantic cod (*Gadus morhua*), saithe (*Pollachius virens*), and spotted wolffish (*Anarhichas minor*) were examined by Jørgensen et al. (2005).Researchers investigated potential effects on survival, development, and behavior in this study. Among fish kept in tanks and observed for one to four weeks after sound exposure, no significant differences in mortality or growth-related parameters between exposed and unexposed groups were observed. Examination of organs and tissues from selected herring experiments did not reveal obvious differences between unexposed and exposed groups. However, two (out of 42) of the herring groups exposed to sound pressure levels of 189 dB re 1 µPa and 179 dB re 1 µPa had a post-exposure mortality of 19 and 30 percent, respectively. It is not clear if this increased mortality was due to the received level or to other unknown factors, such as exposure to the resonance frequency of the swim bladder. Jørgensen et al. (2005) estimated a resonant frequency of 1.8 kHz for herring and saithe ranging in size from 6.3 to 7.0 cm, respectively, which lies within the range of frequencies used during sound exposures and therefore may explain some of the noted mortalities.

Individual juvenile fish with a swim bladder resonance in the frequency range of the operational sonars may be more susceptible to injury or mortality. Past research has demonstrated that fish species, size,

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[1] Rise time: the amount of time for a signal to change from static pressure (the ambient pressure without the added sound) to high pressure. Rise times for non-impulsive sound typically have relatively gradual increases in pressure, while impulsive sound has near-instantaneous rise to a high peak pressure. For more detail, see Appendix H (Acoustic and Explosive Concepts).

and depth influences resonant frequency (Løvik & Hovem, 1979; McCartney & Stubbs, 1971). At resonance, the swim bladder, which can amplify vibrations that reach the fish's hearing organs, may absorb much of the acoustic energy in the impinging sound wave. It is suspected that the resulting oscillations may cause mortality, harm the auditory organs or the swim bladder (Jørgensen et al., 2005; Kvadsheim & Sevaldsen, 2005b). However, damage to the swim bladder and to tissues surrounding the swim bladder was not observed in fishes exposed to sonar at their presumed swim bladder resonant frequency (Jørgensen et al., 2005). Sonar is expected to physiologically affect adult fish less than juveniles because adult fish are in a more robust stage of development, and their swim bladder resonant frequencies would be lower than that of mid-frequency active sonar. Additionally, adult fish have more ability to move from an unpleasant stimulus (Kvadsheim & Sevaldsen, 2005a). Lower frequencies (i.e., generally below 1 kHz) are expected to produce swim bladder resonance in adult fishes from about 10 to 100 centimeters (McCartney & Stubbs, 1971). Fish, especially larval and small juveniles, are more susceptible to injury from swim bladder resonance when exposed to continuous signals within the resonant frequency range.

Hastings (1991; 1995) tested the limits of acoustic exposure on two freshwater fish species. Hastings found "acoustic stunning" (loss of consciousness) in blue gouramis (*Trichogaster trichopterus*) following an eight-minute continuous exposure in captivity to a 150 Hz pure tone with a sound pressure level of 198 dB re 1 µPa (Hastings, 1995). This species of fish has an air bubble in the mouth cavity directly adjacent to the animal's braincase that may have caused this injury. Hastings (1991; 1995) also found that goldfish (*Carassius auratus*) exposed to a 250 Hz continuous wave sound with peak pressures of 204 dB re 1 µPa for two hours and blue gourami exposed to a 150 Hz continuous wave sound at a sound pressure level of 198 dB re 1 µPa for 0.5 hour did not survive. These studies illustrate the highest-known levels tested on fishes with hearing specializations. These high levels of noise were also projected for relatively long durations of time and in a small tank test environment, therefore direct comparisons to results in natural settings should be treated with caution. Stunning and mortality due to exposure to non-impulsive sound exposure has not been observed in other studies.

Three freshwater species of fish, the rainbow trout (*Oncorhynchus mykiss*), channel catfish (*Ictalurus punctatus*), and the hybrid sunfish (*Lepomis* sp.), were exposed to both low- and mid-frequency sonar (Kane et al., 2010; Popper et al., 2007). Low-frequency exposures with received sound pressure levels of 193 dB re 1 µPa occurred for either 324 or 648 seconds. Mid-frequency exposures with received sound pressure levels of 210 dB re 1 µPa occurred for 15 seconds. No fish mortality resulted from either experiment, and during necropsy after test exposures, both studies found that none of the subjects showed signs of tissue damage related to exposure (Kane et al., 2010; Popper et al., 2007).

As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), although fish have been injured and killed due to intense, long-duration, non-impulsive sound exposures, fish exposed under more realistic conditions have shown no signs of injury. Those species tested to date can be used as viable surrogates for estimating injury in other species exposed to similar sources.

3.9.2.1.1.2 Hearing Loss

Researchers have examined the effects on hearing in fishes from sonar-like signals, tones, and different non-impulsive noise sources. Section 3.0.4.7 (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.

Exposure to high-intensity sound can cause hearing loss, also known as a noise-induced threshold shift, or simply a threshold shift (Miller, 1974). A temporary threshold shift (TTS) is a temporary, recoverable loss of hearing sensitivity. A TTS may last several minutes to several weeks, and the duration may be related to the intensity of the sound source and the duration of the sound (including multiple exposures). A permanent threshold shift (PTS) is non-recoverable, results from the destruction of tissues within the auditory system, permanent loss of hair cells, or damage to auditory nerve fibers (Liberman, 2016), and can occur over a small range of frequencies related to the sound exposure. However, the sensory hair cells of the inner ear in fishes are regularly replaced over time when they are damaged, unlike in mammals where sensory hair cells loss is permanent (Lombarte et al., 1993; Popper et al., 2014; Smith et al., 2006). Consequently, PTS has not been known to occur in fishes, and any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al., 2005; Popper et al., 2014; Smith et al., 2006). (Popper et al., 2005; Popper et al., 2014; Smith et al., 2006). As with TTS, the animal does not become deaf but requires a louder sound stimulus, relative to the amount of PTS, to detect a sound within the affected frequencies. Although available data for some terrestrial mammals have shown signs of nerve damage after severe threshold shifts (e.g., Kujawa & Liberman, 2009; Lin et al., 2011), it is not known if damage to auditory nerve fibers could also occur in fishes and, if so, whether fibers would recover during this process. One example that demonstrated a lack of damage to sensory receptors when TTS occurred was in a study on hearing loss in zebrafish (*Danio rerio,* a freshwater species with a swim bladder involved in hearing). This was one of the first studies to look at both auditory threshold shifts and potential physical effects on the inner ear. However, marine species have yet to be tested and future research should evaluate other potential mechanisms of cellular or structural damage if in fact physical damage occurs in fishes with the onset of a threshold shift (Breitzler et al., 2020).

Hearing Loss due to Impulsive Sound Sources

Popper et al. (2005) examined the effects of a seismic air gun array on a fish with a swim bladder that is involved in hearing, the lake chub (*Couesius plumbeus*), and two species that have a swim bladder that is not involved in hearing, the northern pike (*Esox lucius*) and the broad whitefish (*Coregonus nasus*), a salmonid. In this study, the lowest received cumulative sound exposure level at which effects were noted was 186 dB re 1 µPa^2-s (5 shots with a mean sound pressure level of 177 dB re 1 µPa). The results showed temporary hearing loss for both lake chub and northern pike to both 5 and 20 air gun shots, but not for the broad whitefish. Hearing loss was approximately 20 to 25 dB at some frequencies for both species, and full recovery of hearing took place within 18 hours after sound exposure. Examination of the sensory surfaces of the ears after allotted recovery times (one hour for five shot exposures, and up to 18 hours for 20 shot exposures) showed no damage to sensory hair cells in any of the fish from these exposures (Song et al., 2008).

McCauley et al. (2003) and McCauley and Kent (2012) showed loss of a small percent of sensory hair cells in the inner ear of caged fish exposed to a towed air gun array simulating a passing seismic vessel. Pink snapper (*Pagrus auratus*), a species that has a swim bladder that is not involved in hearing, were exposed to multiple air gun shots for up to 1.5 hours (McCauley et al., 2003) where the maximum received sound exposure levels exceeded 180 dB re 1 µPa^2-s. The loss of sensory hair cells continued to increase for up to at least 58 days post exposure to 2.7 percent of the total cells. Gold band snapper (*Pristipomoides multidens*) and sea perch (*Lutjanis kasmira*), both fishes with a swim bladder involved in hearing, were also exposed to a towed air gun array simulating a passing seismic vessel (McCauley & Kent, 2012). Although received levels for these exposures have not been published, hair cell damage increased as the range of the exposure (i.e., range to the source) decreased. Again, the amount of damage was considered small in each case (McCauley & Kent, 2012). It is not known if this hair cell loss would result in hearing loss since fish have tens or even hundreds of thousands of sensory hair cells in the inner ear and only a small portion were affected by the sound (Lombarte & Popper, 1994; Popper & Hoxter, 1984). A reason McCauley and Kent (2012) found damage to sensory hair cells while Popper et al. (2005) did not may be in their distinct methodologies. Their studies had many differences, including species and the precise sound source characteristics.

Hastings et al. (2008) exposed a fish with a swim bladder that is involved in hearing, the pinecone soldierfish (*Myripristis murdjan*), and three species that have a swim bladder that is not involved in hearing, the blue green damselfish (*Chromis viridis*), the saber squirrelfish (*Sargocentron spiniferum*), and the bluestripe seaperch (*Lutjanus kasmira*), to an air gun array. Fish in cages were exposed to multiple air gun shots with a cumulative sound exposure level of 190 dB re 1 µPa^2-s. The authors found no hearing loss in any fish examined up to 12 hours after the exposures.

In an investigation of another impulsive source, Casper et al. (2013b) found that some fishes may actually be more susceptible to barotrauma (e.g., swim bladder ruptures, herniations, and hematomas) than hearing effects when exposed to simulated impact pile driving. Hybrid striped bass (white bass x striped bass) and Mozambique tilapia (*Oreochromis mossambicus*), two species with a swim bladder not involved in hearing, were exposed to sound exposure levels between 213 and 216 dB re 1 µPa^2-s. The subjects exhibited barotrauma, and although researchers began to observe signs of inner ear hair cell loss, these effects were small compared to the other non-auditory injuries incurred. Researchers speculated that injury might occur prior to signs of hearing loss or TTS. These sound exposure levels may present the lowest threshold at which hearing effects may begin to occur.

Overall, PTS has not been known to occur in fishes tested to date. Any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al., 2005; Popper et al., 2014; Smith et al., 2006). The lowest sound exposure level at which TTS has been observed in fishes with a swim bladder involved in hearing is 186 dB re 1 µPa^2-s. As reviewed in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), fishes without a swim bladder, or fishes with a swim bladder that is not involved in hearing, would be less susceptible to hearing loss (i.e., TTS) than fishes with swim bladders involved in hearing, even at higher levels and longer durations.

Hearing Loss due to Sonar and Other Transducers

Several studies have examined the effects of the sound exposures from low-frequency sonar on fish hearing (i.e., Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). Hearing was measured both immediately post exposure and for up to several days thereafter (Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). Maximum received sound pressure levels were 193 dB re 1 µPa for 324 or 648 seconds (a cumulative sound exposure level of 218 or 220 dB re 1 µPa^2-s, respectively) at frequencies ranging from 170 to 320 Hz (Kane et al., 2010; Popper et al., 2007), and 195 dB re 1 µPa for 324 seconds (a cumulative sound exposure level of 215 dB re 1 µPa^2-s) in a follow-on study (Halvorsen et al., 2013). Two species with a swim bladder not involved in hearing, the largemouth bass (*Micropterus salmoides*) and yellow perch (*Perca flavescens*), showed no loss in hearing sensitivity from sound exposure immediately after the test or 24 hours later. Channel catfish, a fish with a swim bladder involved in hearing, and some specimens of rainbow trout, a fish with a swim bladder not involved in hearing, showed a threshold shift (up to 10–20 dB of hearing loss) immediately after exposure to the low-frequency sonar when compared to baseline and control animals. Small thresholds shifts were detected for up to 24 hours after the experiment in some channel catfish. Although some rainbow trout in one test group showed signs of hearing loss, rainbow trout in another group showed no hearing loss. The different results between rainbow trout test groups are difficult to understand, but may be due to development or genetic differences in the various groups of fish. Catfish hearing returned to, or close to, normal within about 24 hours after exposure to low-frequency sonar. Examination of the inner ears of the fish during necropsy revealed no differences from the control groups in ciliary bundles or other features indicative of hearing loss. The maximum time fish were held post exposure before sacrifice was 96 hours (Kane et al., 2010).

The same investigators examined the potential effects of mid-frequency active sonar on fish hearing and the inner ear (Halvorsen et al., 2012c; Kane et al., 2010). The maximum received sound pressure level was 210 dB re 1 µPa at a frequency of 2.8 to 3.8 kHz for a total duration of 15 seconds (cumulative sound exposure level of 220 dB re 1 μPa^2-s). Out of the species tested (rainbow trout and channel catfish), only one test group of channel catfish showed any hearing loss after exposure to mid-frequency active sonar. The investigators tested catfish during two different seasons and found that the group tested in October experienced TTS, which recovered within 24 hours, but fish tested in December showed no effect. It was speculated that the difference in hearing loss between catfish groups might have been due to the difference in water temperature during the testing period or due to differences between the two stocks of fish (Halvorsen et al., 2012c). Any effects on hearing in channel catfish due to sound exposure appeared to be short-term and non-permanent (Halvorsen et al., 2012c; Kane et al., 2010).

Some studies have suggested that there may be some loss of sensory hair cells due to high-intensity sources, indicating a loss in hearing sensitivity; however, none of those studies concurrently investigated the subjects' actual hearing range after exposure to these sources. Enger (1981) found loss of ciliary bundles of the sensory cells in the inner ears of Atlantic cod following one to five hours of exposure to pure tone sounds between 50 and 400 Hz with a sound pressure level of 180 dB re 1 µPa. Hastings (1995) found auditory hair-cell damage in goldfish, a freshwater species with a swim bladder that is involved in hearing. Goldfish were exposed to 250 Hz and 500 Hz continuous tones with maximum peak sound pressure levels of 204 dB re 1 µPa and 197 dB re 1 µPa, respectively, for about two hours. Similarly, Hastings et al. (1996) demonstrated damage to some sensory hair cells in oscars (*Astronotus ocellatus*) observed one to four days following a one-hour exposure to a pure tone at 300 Hz with a sound pressure level of 180 dB re 1 µPa, but no damage to the lateral line was observed. Both studies found a relatively small percentage of total hair cell loss from hearing organs despite long duration exposures. Effects from long-duration noise exposure studies are generally informative; however, they are not necessarily a direct comparison to intermittent short-duration exposures produced during Navy activities involving sonar and other transducers.

As noted in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), some fish species with a swim bladder that is involved in hearing may be more susceptible to TTS from high-intensity non-impulsive sound sources, such as sonar and other transducers, depending on the duration and frequency content of the exposure. Fishes with a swim bladder involved in hearing and fishes with high-frequency hearing may exhibit TTS from exposure to low- and mid-frequency sonar, specifically at cumulative sound exposure levels above 215 dB re 1 μPa^2-s. However, fishes without a swim bladder and fishes with a swim bladder that is not involved in hearing would be unlikely to detect mid-frequency or other higher-frequency sonars and would likely require a much higher sound exposure level to exhibit the same effect from exposure to low-frequency active sonar.

Hearing Loss due to Vessel Noise

Little data exist on the effects of vessel noise on hearing in fishes. However, TTS has been observed in fishes exposed to elevated background noise and other non-impulsive sources (e.g., white noise). Caged studies on pressure-sensitive fishes (i.e., fishes with a swim bladder involved in hearing and those with high-frequency hearing) show some hearing loss after several days or weeks of exposure to increased background sounds, although the hearing loss seems to recover (e.g., Scholik & Yan, 2002b; Smith et al., 2004b; Smith et al., 2006). Smith et al. (2004b; 2006) exposed goldfish, to noise with a sound pressure level of 170 dB re 1 µPa and found a clear relationship between the amount of hearing loss and the duration of exposure until maximum hearing loss occurred at about 24 hours of exposure. A 10-minute exposure resulted in 5 dB of TTS, whereas a three-week exposure resulted in a 28 dB TTS that took over two weeks to return to pre-exposure baseline levels (Smith et al., 2004b). Recovery times were not measured by investigators for shorter exposure durations. It is important to note that these exposures were continuous and subjects were unable to avoid the sound source for the duration of the experiment.

Scholik and Yan (2001) demonstrated TTS in fathead minnows (*Pimephales promelas*) after a 24-hour continuous exposure to white noise (0.3–2.0 kHz) at 142 dB re 1 µPa that took up to 14 days post-exposure to recover. This is the longest recorded time for a threshold shift to recover in a fish. The same authors also found that the bluegill sunfish (*Lepomis macrochirus*), a species that primarily detects particle motion and lacks specializations for hearing, did not show significant elevations in auditory thresholds when exposed to the same stimulus (Scholik & Yan, 2002a). This demonstrates again that fishes with a swim bladder involved in hearing and those with high-frequency hearing may be more sensitive to hearing loss than fishes without a swim bladder or those with a swim bladder not involved in hearing.

Breitzler et al. (2020) exposed zebrafish (a freshwater fish with a swim bladder involved in hearing) to 24 hours of white noise at various frequencies and sound levels. This is one of the first studies that measured hearing thresholds, physical damage (i.e., loss of hair cells), and recovery post-exposure. Overall, results were similar to those from previous studies. As the noise level increased, the amount of TTS observed in zebrafish also increased and frequencies that were most affected were those within the fishes best hearing sensitivity. Breitzler et al. (2020) also observed an increase in response latency in fish with TTS (i.e., the fish were slower to respond to auditory stimuli during hearing tests). Threshold shifts in fish exposed to 130 dB and 140 dB recovered within three days whereas it took up to 14 days for fish exposed to the highest exposure level (150 dB) to return to pre-exposure levels. Similarly, response latency was time dependent and sometimes took up to 14 days to recover to pre-exposure levels. The highest threshold shifts recorded also resulted in hair cell loss but, similar to the other effects measured in this study, hair cells returned to baseline levels within seven days post-exposure. This further demonstrates the ability for fish to rejuvenate hair cells and for hearing thresholds to recover to baseline levels without any evidence of PTS.

When reviewing results from these three studies, it is important to note that the fish were unable to avoid the sound source (e.g., held stationary in tubs or tanks) and were subject to long, continuous duration exposures (e.g., days to weeks). A direct comparison of these results to fish exposed to continuous sound sources in natural settings should be treated with caution. For example, fishes that are exposed to vessel noise in their natural environment, even in areas with high levels of vessel movement, would only be exposed for periods of short durations (e.g., seconds or minutes) as vessels pass by. Therefore, overall effects would not likely rise to the level of impact demonstrated in laboratory studies. As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), some fish species with a swim bladder that is involved in hearing may be more susceptible to TTS from long duration continuous noise, such as broadband[2] white noise, depending on the duration of the exposure (thresholds are proposed based on continuous exposure of 12 hours). However, it is not likely that TTS would occur in fishes with a swim bladder not involved in hearing or in fishes without a swim bladder.

3.9.2.1.1.3 Masking

Masking refers to the presence of a noise that interferes with a fish's ability to hear biologically important sounds, including those produced by prey, predators, or other fishes. Masking occurs in all vertebrate groups and can effectively limit the distance over which an animal can communicate and detect biologically relevant sounds. Human-generated continuous sounds (e.g., some sonar, vessel or aircraft noise, and vibratory pile driving) have the potential to mask sounds that are biologically important to fishes. Researchers have studied masking in fishes using continuous masking noise, but masking due to intermittent, short-duty cycle sounds has not been studied. Section 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on masking and the framework used to analyze this potential impact.

Masking is likely to occur in most fishes due to varying levels of ambient or natural noise in the environment such as wave action, precipitation, or other animal vocalizations (Popper et al., 2014). Ambient noise during higher sea states in the ocean has resulted in elevated thresholds in several fish species (Chapman & Hawkins, 1973; Ramcharitar & Popper, 2004). Although the overall intensity or loudness of ambient or human-generated noise may result in masking effects in fishes, masking may be most problematic when human-generated signals or ambient noise levels overlap the frequencies of biologically important signals (Buerkle, 1968, 1969; Popper et al., 2014; Tavolga, 1974).

Wysocki and Ladich (2005) investigated the influence of continuous white noise exposure on the auditory sensitivity of two freshwater fish with notable hearing specializations for sound pressure detection, the goldfish and the lined Raphael catfish (*Platydoras costatus*), and a freshwater fish without notable specializations, the pumpkinseed sunfish (*Lepomis gibbosus*). For the goldfish and catfish, baseline thresholds were lower than masked thresholds. Continuous white noise with a sound pressure level of approximately 130 dB re 1 µPa at 1 m resulted in an elevated threshold of 23–44 dB within the subjects' region of best sensitivity between 500 and 1,000 Hz. There was less evidence of masking in the sunfish during the same exposures, with only a shift of 11 dB. Wysocki and Ladich (2005) suggest that ambient sound regimes may limit acoustic communication and orientation, especially in animals with notable hearing specializations for sound pressure detection.

Masking could lead to potential fitness costs depending on the severity of the reaction and the animal's ability to adapt or compensate during an exposure (de Jong et al., 2020; Radford et al., 2014; Slabbekoorn et al., 2010). For example, masking could result in changes in predator-prey relationships, potentially inhibiting a fish's ability to detect predators and therefore increase its risk of predation (Astrup, 1999; Mann et al., 1998; Simpson et al., 2015; Simpson et al., 2016). Masking may also limit the distance over which fish can communicate or detect important signals (Alves et al., 2016; Codarin et al.,

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[2] A sound or signal that contains energy across multiple frequencies.

2009; Ramcharitar et al., 2001; Ramcharitar et al., 2006), including sounds emitted from a reef for navigating larvae (Higgs, 2005; Neenan et al., 2016). If the masking signal is brief (a few seconds or less), biologically important signals may still be detected, resulting in little effect to the individual. If the signal is longer in duration (minutes or hours) or overlaps with important frequencies for a particular species, more severe consequences may occur such as the inability to attract a mate and reproduce. Holt and Johnston (2014) were the first to demonstrate the Lombard effect in one species of fish, a potentially compensatory behavior where an animal increases the source level of its vocalizations in response to elevated noise levels. The Lombard effect is currently understood to be a reflex that may be unnoticeable to the animal or may lead to increased energy expenditure during communication.

The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) highlights a lack of data that exists for masking by sonar but suggests that the narrow bandwidth and intermittent nature of most sonar signals would result in only a limited probability of any masking effects. In addition, most sonars (mid-, high-, and very high-frequency) are above the hearing range of most marine fish species, eliminating the possibility of masking for these species. In most cases, the probability of masking would further decrease with increasing distance from the sound source.

In addition, no data are available on masking by impulsive signals (e.g., impact pile driving and air guns) (Popper et al., 2014). Impulsive sounds are typically brief, lasting only fractions of a second, where masking could occur only during that brief duration of sound. Biological sounds can typically be detected between pulses within close distances to the source unless those biological sounds are similar to the masking noise, such as impulsive or drumming vocalizations made by some fishes (e.g., cod or haddock). Masking could also indirectly occur because of repetitive impulsive signals where the repetitive sounds and reverberations over distance may create a more continuous noise exposure.

Although there is evidence of masking because of exposure to vessel noise, the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) does not present numeric thresholds for this effect. Instead, relative risk factors are considered, and it is assumed the probability of masking occurring is higher at near to moderate distances from the source (up to hundreds of meters) but decreases with increasing distance (Popper et al., 2014).

3.9.2.1.1.4 Physiological Stress

Section 3.0.4.7 (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. A fish must first be able to detect a sound above its hearing threshold and above the ambient noise level before a physiological stress reaction can occur. The initial response to a stimulus is a rapid release of stress hormones into the circulatory system, which may cause other responses such as elevated heart rate and blood chemistry changes. Although an increase in background sound has been shown to cause stress in humans and animals, only a limited number of studies have measured biochemical responses by fishes to acoustic stressors (e.g., Goetz et al., 2015; Madaro et al., 2015; Remage-Healey et al., 2006; Smith et al., 2004a; Wysocki et al., 2006; Wysocki et al., 2007), and the results have varied. Stimuli that have been used to study physiological stress responses in fishes include predator vocalizations, non-impulsive or continuous, and impulsive noise exposures.

A stress response that has been observed in fishes is the production of cortisol (a stress hormone) when exposed to sounds such as boat noise, tones, or predator vocalizations. Nichols et al. (2015) found that giant kelpfish (*Heterostichus rostratus*) had increased levels of cortisol with increased sound level and intermittency of boat noise playbacks. Cod exposed to a short-duration upsweep (a tone that sweeps upward across multiple frequencies) across 100–1,000 Hz had increases in cortisol levels, which returned to normal within one hour post-exposure (Sierra-Flores et al., 2015). Remage-Healey et al. (2006) found elevated cortisol levels in Gulf toadfish (*Opsanus beta*) exposed to low-frequency bottlenose dolphin sounds, but they observed no physiological change when they exposed toadfish to low-frequency "pops" produced by snapping shrimp.

A sudden increase in sound pressure level (i.e., presentation of a sound source) or an increase in overall background noise levels can increase hormone levels and alter other metabolic rates indicative of a stress response, such as increased ventilation and oxygen consumption (Pickering, 1981; Popper & Hastings, 2009a; Radford et al., 2016; Simpson et al., 2015; Simpson et al., 2016; Smith et al., 2004a, 2004b; Spiga et al., 2017). Similarly, reef fish embryos exposed to boat noise have demonstrated changes in morphological development and increases in heart rate, another indication of a physiological stress response, although survival rates were unchanged (Fakan & McCormick, 2019; Jain-Schlaepfer et al., 2018). Although results have varied, it has been shown that chronic or long-term (days or weeks) exposures of continuous man-made sounds can lead to a reduction in embryo viability (Sierra-Flores et al., 2015) and decreased growth rates (Nedelec et al., 2015).

However, not all species tested to date show these reactions. Smith et al. (2004a) found no increase in corticosteroid, a class of stress hormones, in goldfish exposed to a continuous, band-limited noise (0.1–10 kHz) with a sound pressure level of 170 dB re 1 µPa for one month. Wysocki et al. (2007) exposed rainbow trout to continuous band-limited noise with a sound pressure level of about 150 dB re 1 µPa for nine months with no observed stress effects. Growth rates and effects on the trout's immune systems were not significantly different from control animals held at a sound pressure level of 110 dB re 1 µPa.

Fishes may have physiological stress reactions to sounds that they can hear. Generally, stress responses are more likely to occur in the presence of potentially threatening sound sources, such as predator vocalizations, or the sudden onset of impulsive signals rather than from non-impulsive or continuous sources such as vessel noise or sonar. If an exposure is short, the stress responses are typically brief (a few seconds to minutes). In addition, research shows that fishes may habituate (i.e., learn to tolerate) to the noise that is being presented after multiple exposures or longer duration exposures that prove to be non-threatening. However, exposure to chronic noise sources can lead to more severe impacts over time, such as reduced growth rates which can lead to reduced survivability for an individual. It is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

3.9.2.1.1.5 Behavioral Reactions

Section 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on behavioral reactions and the framework used to analyze this potential impact. Behavioral reactions in fishes have been observed due to a number of different types of sound sources. The majority of research has been performed using air guns (including large-scale seismic surveys), sonar, and vessel noise. Fewer observations have been made on behavioral reactions to impact pile driving noise, although fish are likely to show similar behavioral reactions to any impulsive noise within or outside the zone for hearing loss and injury.

As with masking, a fish must first be able to detect a sound above its hearing threshold and above the ambient noise level before a behavioral reaction can occur. Most fishes can only detect low-frequency sounds, with the exception of a few species that can detect some mid and high frequencies (above 1 kHz).

Fish studies have identified the following behavioral reactions to sound: alteration of natural behaviors (e.g., startle or alarm), and avoidance (LGL Ltd Environmental Research Associates et al., 2008; McCauley et al., 2000; Pearson et al., 1992). In the context of this SEIS/OEIS, and to remain consistent with available behavioral reaction literature, the terms "startle," "alarm," "response," and "reaction" will be used synonymously.

In addition, observed behavioral effects to fish could include disruption or alteration of natural activities such as swimming, schooling, feeding, breeding, and migrating. Sudden changes in sound level can cause fish to dive, rise, or change swimming direction. However, some fish may learn to tolerate or habituate to repeated exposures and noise that seems threatening (e.g., Bruintjes et al., 2016; Currie et al., 2020; Nedelec et al., 2016b; Radford et al., 2016).

Research on behavioral reactions can be difficult to understand and interpret. For example, behavioral responses often times vary depending on the type of exposure and sound source present. Changes in sound intensity may be more important to a fishes' behavior than the maximum sound level. Some studies show that sounds that fluctuate in level or have intermittent pulse rates tend to elicit stronger responses from fish than even stronger sounds with a continuous level (Currie et al., 2020; Neo et al., 2014; Schwarz & Greer, 1984). It has also been suggested that unpredictable sounds that last for long durations may have the largest impact on behavioral responses (de Jong et al., 2020). Interpreting behavioral responses can also be difficult due to species-specific behavioral tendencies, motivational state (e.g., feeding or mating), an individual's previous experience, how resilient a species is to changes in their environment, and whether or not the fish are able to avoid the source (e.g., caged versus free-swimming subjects). Results from caged studies may not provide a clear understanding of how free-swimming fishes may react to the same or similar sound exposures (Hawkins et al., 2015).

Behavioral Reactions due to Impulsive Sound Sources

It is assumed that most species would react similarly to impulsive sources such as weapons noise and explosions. However, it is important to note that most data on behavioral reactions to impulsive sources is collected from studies using air guns and impact pile driving, sources that do not occur in the Study Area. Reactions include startle or alarm responses and increased swim speeds at the onset of impulsive sounds (Fewtrell & McCauley, 2012; Pearson et al., 1992; Roberts et al., 2016a; Spiga et al., 2017). Data on fish behavioral reactions exposed to impulsive sound sources is mostly limited to studies using caged fishes and the use of seismic air guns (Løkkeborg et al., 2012). Several species of rockfish (*Sebastes* species) in a caged environment exhibited startle or alarm reactions to seismic air gun pulses between peak-to-peak sound pressure levels of 180 dB re 1 µPa and 205 dB re 1 µPa (Pearson et al., 1992). More subtle behavioral changes were noted at lower sound pressure levels, including decreased swim speeds. At the presentation of the sound, some species of rockfish settled to the bottom of the experimental enclosure and reduced swim speed. Trevally (*Pseudocaranx dentex*) and pink snapper also exhibited alert responses as well as changes in swim depth, speed, and schooling behaviors when exposed to air gun noise (Fewtrell & McCauley, 2012). Both trevally and pink snapper swam faster and closer to the bottom of the cage at the onset of the exposure. However, trevally swam in tightly cohesive groups at the bottom of the test cages while pink snapper exhibited much looser group cohesion. These behavioral responses were seen during sound exposure levels as low as 147 up to 161 dB re 1 μPa^2-s but habituation occurred in all cases, either within a few minutes or within 30 minutes after the final air gun shot (Fewtrell & McCauley, 2012; Pearson et al., 1992).

Some studies have shown a lack of behavioral reactions to air gun noise. Herring exposed to an approaching air gun survey (from 27 to 2 km over six hours), resulting in single pulse sound exposure levels of 125–155 dB re 1 µPa^2-s, did not react by changing direction or swim speed (Pena et al., 2013). Although these levels are similar to those tested in other studies which exhibited responses (Fewtrell & McCauley, 2012), the distance of the exposure to the test enclosure, the slow onset of the sound source, and a strong motivation for feeding may have affected the observed response (Pena et al., 2013). In another study, Wardle et al. (2001) observed marine fish on an inshore reef before, during, and after an air gun survey at varying distances. The air guns were calibrated at a peak level of 210 dB re 1 µPa at 16 m and 195 dB re 1 µPa at 109 m from the source. Other than observed startle responses and small changes in the position of pollack, when the air gun was located within close proximity to the test site (within 10 m), they found no substantial or permanent changes in the behavior of the fish on the reef throughout the course of the study. Behavioral responses to impulsive sources are more likely to occur within near and intermediate (tens to hundreds of meters) distances from the source as opposed to far distances (thousands of meters) (Popper et al., 2014).

Unlike the previous studies, Slotte et al. (2004) used fishing sonar (38 kHz echo sounder) to monitor behavior and depth of blue whiting (*Micromesistius poutassou*) and Norwegian spring herring (*Claupea harengus L.*) spawning schools exposed to air gun signals. They reported that fishes in the area of the air guns appeared to go to greater depths after the air gun exposure compared to their vertical position prior to the air gun usage. Moreover, the abundance of animals 30–50 km away from the air guns increased during seismic activity, suggesting that migrating fish left the zone of seismic activity and did not re-enter the area until the activity ceased. It is unlikely that either species was able to detect the fishing sonar. However, it should be noted that these behavior patterns may have also been influenced by other variables such as motivation for feeding, migration, or other environmental factors (e.g., temperature, salinity, etc.) (Slotte et al., 2004).

Alterations in natural behavior patterns due to exposure to pile driving noise have not been studied as thoroughly, but reactions noted thus far are similar to those seen in response to seismic surveys. These changes in behavior include startle responses, changes in depth (in both caged and free-swimming subjects), increased swim speeds, changes in ventilation rates, changes in attention and anti-predator behaviors, and directional avoidance (e.g., Hawkins et al., 2014; Mueller-Blenkle et al., 2010; Neo et al., 2015; Roberts et al., 2016a; Spiga et al., 2017). The severity of response varied greatly by species and received sound pressure level of the exposure. For example, some minor behavioral reactions such as startle responses were observed during caged studies with a sound pressure level as low as 140 dB re 1 µPa (Neo et al., 2014). However, only some free-swimming fishes avoided pile driving noise at even higher sound pressure levels between 152 and 157 dB re 1 µPa (Iafrate et al., 2016). In addition, Roberts et al. (2016a) observed that although multiple species of free swimming fish responded to simulated pile driving recordings, not all responded consistently. In some cases, only one fish would respond while the others continued feeding from a baited remote underwater video. In other instances, various individual fish would respond to different strikes. The repetition rate of pulses during an exposure may also have an effect on what behaviors were noted and how quickly these behaviors recovered as opposed to the overall sound pressure or exposure level (Neo et al., 2014). Neo et al. (2014) observed slower recovery times in fishes exposed to intermittent sounds (similar to pile driving) compared to continuous exposures.

As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), species may react differently to the same sound source depending on a number of variables, such as the animal's life stage or behavioral state (e.g., feeding, mating). Without specific data, it is assumed that fishes react similarly to all impulsive sounds outside the zone for hearing loss and injury. Observations of fish reactions to large-scale air gun surveys are informative, but not necessarily directly applicable to analyzing impacts from the short-term, intermittent use of all impulsive sources. It is assumed that fish have a high probability of reacting to an impulsive sound source within near and intermediate distances (tens to hundreds of meters), and a decreasing probability of reaction at increasing distances (Popper et al., 2014).

Behavioral Reactions due to Sonar and Other Transducers

Behavioral reactions to sonar have been studied both in caged and free-swimming fish, although results can oftentimes be difficult to interpret depending on the species tested and the study environment. Jørgensen et al. (2005) showed that caged cod and spotted wolf fish (*Anarhichas minor*) lacked any response to simulated sonar between 1 and 8 kHz. However, within the same study, reactions were seen in juvenile herring. It is likely that the sonar signals were inaudible to the cod and wolf fish (species that lack notable hearing specializations), but audible to herring (a species that has hearing capabilities in the frequency ranges tested).

Doksæter et al. (2009; 2012) and Sivle et al. (2012; 2014) studied the reactions of both wild and captive Atlantic herring to the Royal Netherlands Navy's experimental mid-frequency active sonar ranging from 1 to 7 kHz. The behavior of the fish was monitored in each study either using upward-looking echosounders (for wild herring) or audio and video monitoring systems (for captive herring). The source levels used within each study varied across all studies and exposures with a maximum received sound pressure level of 181 dB re 1 µPa and maximum cumulative sound exposure level of 184 dB re 1 μPa^2s. No avoidance or escape reactions were observed when herring were exposed to any sonar sources. Instead, significant reactions were noted at lower received sound levels of different non-sonar sound types. For example, dive responses (i.e., escape reactions) were observed when herring were exposed to killer whale feeding sounds at received sound pressure levels of approximately 150 dB re 1 µPa (Sivle et al., 2012). Startle responses were seen when the cages for captive herring were hit with a wooden stick and with the ignition of an outboard boat engine at a distance of one meter from the test pen (Doksaeter et al., 2012). It is possible that the herring were not disturbed by the sonar, were more motivated to continue other behaviors such as feeding, or did not associate the sound as a threatening stimulus. Based on these results (Doksaeter et al., 2009; Doksaeter et al., 2012; Sivle et al., 2012), Sivle et al. (2014) created a model in order to report on the possible population-level effects on Atlantic herring from active naval sonar. The authors concluded that the use of naval sonar poses little risk to populations of herring regardless of season, even when the herring populations are aggregated and directly exposed to sonar.

There is evidence that elasmobranchs (cartilaginous fish including sharks and rays) also respond to human-generated sounds. A number of researchers conducted experiments in which they played back sounds (e.g., pulsed tones below 1 kHz) and attracted a number of different shark species to the sound source (e.g., Casper et al., 2012a; Myrberg et al., 1976; Myrberg et al., 1969; Myrberg et al., 1972; Nelson & Johnson, 1972). The results of these studies showed that sharks were attracted to irregularly pulsed low-frequency sounds (below several hundred Hz), in the same frequency range of sounds that might be produced by struggling prey. However, abrupt and irregularly pulsed human-generated noise (0.2–10 kHz, with most energy below 1 kHz) resulted in withdrawal responses of certain shark species (Chapuis et al., 2019). Sharks are not known to be attracted to continuous signals or higher frequencies that they presumably cannot hear (Casper & Mann, 2006; Casper & Mann, 2009).

Only a few species of fishes can detect sonars above 1 kHz (see Section 3.9.1.1, Hearing and Vocalization), meaning that most fishes would not detect most mid-, high-, or very high-frequency Navy sonars. The few marine species that can detect above 1 kHz and have some hearing specializations may be able to better detect the sound and would therefore be more likely to react. However, researchers have found little reaction by adult fish in the wild to sonars within the animals' hearing range (Doksaeter et al., 2009; Doksaeter et al., 2012; Sivle et al., 2012). The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) suggests that fish able to hear sonars would have a low probability of reacting to the source within near or intermediate distances (within tens to hundreds of meters) and a decreasing probability of reacting at increasing distances.

Behavioral Reactions due to Vessel Noise

Vessel traffic also contributes to the amount of noise in the ocean and has the potential to affect fishes. Several studies have demonstrated and reviewed avoidance responses by fishes (e.g., herring and cod) to the low-frequency sounds of vessels (De Robertis & Handegard, 2013; Engås et al., 1995; Handegard et al., 2003). Misund (1997) found that fish that were ahead of a ship and showed avoidance reactions did so at ranges of 50–150 m. When the vessel passed over them, some species of fish responded with sudden escape responses that included lateral avoidance or downward compression of the school.

As mentioned above, behavioral reactions are quite variable and depend on a number of factors such as (but not limited to) the type of fish, its life history stage, behavior, time of day, location, the type of vessel, and the sound propagation characteristics of the water column (Popper et al., 2014; Schwarz & Greer, 1984). Reactions to playbacks of continuous noise or passing vessels generally include basic startle and avoidance responses, as well as evidence of distraction and increased decision-making errors. Other specific examples of observed responses include increased group cohesion, increased distractions or evidence of modified attention, changes in vertical distribution in the water column, changes in swim speeds, distance traveled, as well as changes in feeding efficacy such as reduced foraging attempts and increased mistakes (i.e., lowered discrimination between food and non-food items) (e.g., Bracciali et al., 2012; De Robertis & Handegard, 2013; Handegard et al., 2015; McCormick et al., 2019; Nedelec et al., 2015; Nedelec et al., 2017a; Neo et al., 2015; Payne et al., 2015; Purser & Radford, 2011; Roberts et al., 2016a; Sabet et al., 2016; Simpson et al., 2015; Simpson et al., 2016; Voellmy et al., 2014a; Voellmy et al., 2014b).

Behavioral responses may also be dependent on the type of vessel that fish are exposed to. For example, juvenile damselfish (*Pomacentrus wardi*) exposed to sound from a two-stroke engine resulted in startle responses, reduction in boldness (increased time spent hiding, less time exhibiting exploratory behaviors), and space use (maximum distance ventured from shelter or traveled within the test enclosure), as well as slower and more conservative reactions to visual stimuli analogous to a potential predator. However, damselfish exposed to sound from a four-stroke engine generally displayed similar responses as control fish exposed to ambient noise (e.g., little or no change in boldness) (McCormick et al., 2018; McCormick et al., 2019). Although the two sound sources were very similar, the vessels powered by the four-stroke engine were of lower intensity compared to vessels powered by the two-stroke engine, which may explain the overall reduced response to this engine type.

Vessel noise has also led to changes in anti-predator responses, but these responses vary by species. During exposures to vessel noise, juvenile Ambon damselfish (*Pomacentrus amboinensis*) and European eels showed slower reaction times and lacked startle responses to predatory attacks, and subsequently showed signs of distraction and increased their risk of predation during both simulated and actual predation experiments (Simpson et al., 2015; Simpson et al., 2016). Spiny chromis (*Acanthochromis* *polyacanthus*) exposed to chronic boat noise playbacks for up to 12 consecutive days spent less time feeding and interacting with offspring, and increased defensive acts. In addition, offspring survival rates were also lower at nests exposed to chronic boat noise playbacks versus those exposed to ambient playbacks (Nedelec et al., 2017b). This suggests that chronic or long-term exposures could have more severe consequences than brief exposures.

In contrast, larval Atlantic cod showed a stronger anti-predator response and were more difficult to capture during simulated predator attacks (Nedelec et al., 2015). There are also observations of a general lack of response to shipping and pile driving playback noise by grey mullet (*Chelon labrosus*) and the two-spotted goby (*Gobiusculus flavescens*) (Roberts et al., 2016b). Mensinger et al. (2018) found that Australian snapper located in a protected area showed no change in feeding behavior or avoidance during boat passes, whereas snapper in areas where fishing occurs startled and ceased feeding behaviors during boat presence. This supports that location and past experience also have an influence on whether fishes react.

Although behavioral responses such as those listed above were often noted during the onset of most sound presentations, most behaviors did not last long and animals quickly returned to baseline behavior patterns. In fact, in one study, when given the chance to move from a noisy tank (with sound pressure levels reaching 120–140 dB re 1 μPa) to a quieter tank (sound pressure levels of 110 dB re 1 μPa), there was no evidence of avoidance. The fish did not seem to prefer the quieter environment and continued to swim between the two tanks comparable to control sessions (Neo et al., 2015). However, many of these reactions are difficult to extrapolate to real world conditions due to the captive environment in which testing occurred.

Most fish species should be able to detect vessel noise due to its low-frequency content and their hearing capabilities (see Section 3.9.1.1, Hearing and Vocalization). The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) suggests that fishes have a moderate to high probability of reacting to nearby vessel noise (i.e., within tens of meters) with decreasing probability of reactions with increasing distance from the source (hundreds or more meters).

3.9.2.1.1.6 Long-Term Consequences

Section 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on potential pathways for long-term consequences. Mortality removes an individual fish from the population, while injury reduces the fitness of an individual. Few studies have been conducted on any long-term consequences from repeated hearing loss, stress, or behavioral reactions in fishes due to exposure to loud sounds (Hawkins et al., 2015; Popper & Hastings, 2009a; Popper et al., 2014). Repeated exposures of an individual to multiple sound-producing activities over a season, year, or life stage could cause reactions with costs that can accumulate over time to cause long-term consequences for the individual. These long-term consequences may affect the survivability of the individual, or if impacting enough individuals may have population-level effects, including alteration from migration paths, avoidance of important habitat, or even cessation of foraging or reproductive behavior (Hawkins et al., 2015). Conversely, some animals habituate to or become tolerant of repeated exposures over time, learning to ignore a stimulus that in the past has not accompanied any overt threat. In fact, Sivle et al. (2016) predicted that exposures to sonar at the maximum levels tested would only result in short-term disturbance and would not likely affect the overall population in sensitive fishes such as Atlantic herring (a species which does not occur in the MITT Study Area).

3.9.2.1.2 Impacts from Sonar and Other Transducers

The overall use of sonar and other transducers for training and testing would be similar to what is currently conducted (see Table 2.5-1 and Table 3.0-2 for details). Although individual activities may vary some from those previously analyzed, and some new systems using new technologies would be tested under Alternative 1 and 2, the overall determinations presented in the 2015 MITT Final EIS/OEIS remain valid.

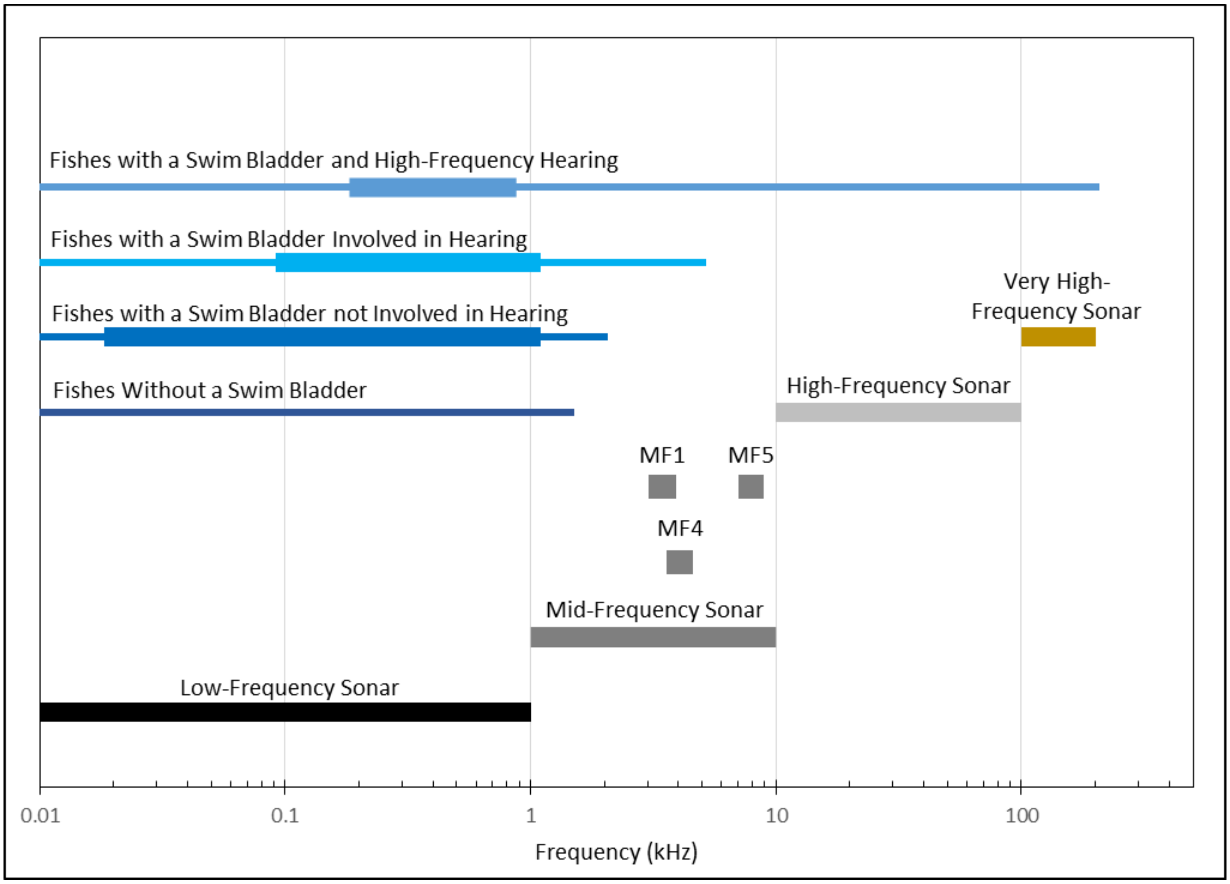
Sonar and other transducers proposed for use are transient in most locations because activities that involve sonar and other transducers take place at different locations and many platforms are generally moving throughout the Study Area. A few activities involving sonar and other transducers occur in inshore waters (within bays and estuaries), including at pierside locations where they reoccur. Sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. General categories and characteristics of these systems and the number of hours these sonars would be operated are described in Section 3.0.4.1.1 (Sonar and Other Transducers). The activities analyzed in this SEIS/OEIS that use sonar and other transducers are described in Appendix A (Training and Testing Activities Descriptions).

As described under Section 3.9.2.1.1.1 (Injury – Injury due to Sonar and Other Transducers), direct injury from sonar and other transducers is highly unlikely because injury has not been documented in fish exposed to sonar (Halvorsen et al., 2012c; Halvorsen et al., 2013; Popper et al., 2007) and therefore is not considered further in this analysis.

Fishes are not equally sensitive to noise at all frequencies. Fishes must first be able to hear a sound in order to be affected by it. As discussed in Section 3.9.1.1 (Hearing and Vocalization), many marine fish species tested to date hear primarily below 1 kHz. For the purposes of this analysis, fish species were grouped into one of four fish hearing groups based on either their known hearing ranges (i.e., audiograms) or physiological features that may be linked to overall hearing capabilities (i.e., swim bladder with connection to, or in close proximity to, the inner ear). Figure 3.9-1 provides a general summary of hearing threshold data from available literature (e.g., Casper & Mann, 2006; Deng et al., 2013; Kéver et al., 2014; Mann et al., 2001; Ramcharitar et al., 2006) to demonstrate the maximum potential range of frequency detection for each hearing group.

Due to data limitations, these estimated hearing ranges may be overly conservative in that they may extend beyond what some species within a given fish hearing group may actually detect. For example, although most sharks are sensitive to lower frequencies, well below 1 kHz, the bull shark has been tested and can detect frequencies up to 1.5 kHz (Kritzler & Wood, 1961; Myrberg, 2001) and therefore represents the uppermost known limit of frequency detection for this hearing group. These upper bounds of each fish hearing groups’ frequency range are outside of the range of best sensitivity for the majority of fishes within that group. As a result, fishes within each group would only be able to detect those upper frequencies at close distances to the source, and from sources with relatively high source levels.

Figure 3.9-1 is not intended as a composite audiogram but rather displays the basic overlap in potential frequency content for each hearing group with Navy defined sonar classes (i.e., low-, mid-, high- and very high-frequency) as discussed under Section 3.0.4.1.1 (Sonar and Other Transducers – Classification of Sonar and Other Transducers).



Notes: Thin blue lines represent the estimated minimum and maximum range of frequency detection for each group. All hearing groups are assumed to hear down to 0.01 kHz regardless of available data. Thicker portions of each blue line represent the estimated minimum and maximum range of best sensitivity for that group. Currently, no data are available to estimate the range of best sensitivity for fishes without a swim bladder. Although each sonar class is represented graphically by the horizontal black, grey and brown bars, not all sources within each class would operate at all the displayed frequencies. Example mid-frequency sources are provided to further demonstrate this. kHz = kilohertz, MF1 = 3.5 kHz, MF4 = 4 kHz, MF5 = 8 kHz.

Figure 3.9-1: Fish Hearing Group and Navy Sonar Bin Frequency Ranges

Systems within the low-frequency sonar class present the greatest potential for overlap with fish hearing. Some mid-frequency sonars and other transducers may also overlap some species' hearing ranges, but to a lesser extent than low-frequency sonars. For example, the only hearing groups that have the potential to be able to detect mid-frequency sources within bins MF1, MF4, and MF5 are fishes with a swim bladder involved in hearing and with high-frequency hearing. It is anticipated that most fishes would not hear or be affected by mid-frequency Navy sonars or other transducers with operating frequencies greater than about 1–4 kHz. Only a few fish species (i.e., fish with a swim bladder and high-frequency hearing specializations) can detect and therefore be potentially affected by high- and very high-frequency sonars and other transducers.

The most probable impacts from exposure to sonar and other transducers are TTS (for more detail see Section 3.9.2.1.1.2, Hearing Loss), masking (for more detail see Section 3.9.2.1.1.3, Masking), physiological stress (for more detail see Section 3.9.2.1.1.4, Physiological Stress), and behavioral reactions (for more detail see Section 3.9.2.1.1.5, Behavioral Reactions). Analysis of these effects are provided below.

3.9.2.1.2.1 Methods for Analyzing Impacts from Sonar and Other Transducers

The Navy performed a quantitative analysis to estimate the range to TTS for fishes exposed to sonar and other transducers used during Navy training and testing activities. Inputs to the quantitative analysis included sound propagation modeling in the Navy Acoustic Effects Model to the sound exposure criteria and thresholds presented below to predict ranges to effects. Although ranges to effect are predicted, density data for fish species within the Study Area are not available; therefore, it is not possible to estimate the total number of individuals that may be affected by sound produced by sonar and other transducers.

Criteria and thresholds to estimate impacts from sonar and other transducers are presented below in Table 3.9-3. Thresholds for hearing loss are typically reported in cumulative sound exposure level so as to account for the duration of the exposure. Therefore, thresholds reported in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) that were presented in other metrics were converted to sound exposure level based on the signal duration reported in the original studies (see Halvorsen et al., 2012c; Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). General research findings from these studies can be reviewed in Section 3.9.2.1.1.2 (Hearing Loss).

Table 3.9-3: Sound Exposure Criteria for TTS from Sonar

Fish Hearing Group	*TTS from Low-Frequency Sonar (SEL_{cum})*	*TTS from Mid-Frequency Sonar (SEL_{cum})*
Fishes without a swim bladder	NC	NC
Fishes with a swim bladder not involved in hearing	> 210	NC
Fishes with a swim bladder involved in hearing	210	220
Fishes with a swim bladder and high-frequency hearing	210	220

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

For mid-frequency sonars, fishes with a swim bladder involved in hearing have shown signs of hearing loss because of mid-frequency sonar exposure at a maximum received sound pressure level of 210 dB re 1 µPa for a total duration of 15 seconds. To account for the total duration of the exposure, the threshold for TTS is a cumulative sound exposure level of 220 dB re 1 µPa^2-s (Halvorsen et al., 2012c; Kane et al., 2010). The same threshold is used for fishes with a swim bladder and high-frequency hearing as a conservative measure, although fishes in this hearing group have not been tested for the same impact. TTS has not been observed in fishes with a swim bladder that is not involved in hearing exposed to mid-frequency sonar. Fishes within this hearing group do not sense pressure well and typically cannot hear at frequencies above 1 kHz (Halvorsen et al., 2012c; Popper et al., 2014). Therefore, no criteria were proposed for fishes with a swim bladder that is not involved in hearing from exposure to mid-frequency sonars, as it is considered unlikely for TTS to occur. Fishes without a swim bladder are even less susceptible to noise exposure; therefore, TTS is unlikely to occur, and no criteria are proposed for this group either.

For low-frequency sonar, as described in Section 3.9.2.1.1.2 (Hearing Loss), exposure of fishes with a swim bladder has resulted in TTS (Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). Specifically, fishes with a swim bladder not involved in hearing showed signs of hearing loss after exposure to a maximum received sound pressure level of 193 dB re 1 μPa for 324 and 648 seconds (cumulative sound exposure level of 218 and 220 dB re 1 μPa^2-s, respectively) (Kane et al., 2010; Popper et al., 2007). In addition, exposure of fishes with a swim bladder involved in hearing to low-frequency sonar at a sound pressure level of 195 dB re 1 μPa for 324 seconds (cumulative sound exposure level of 215 dB re 1 μPa^2-s) resulted in TTS (Halvorsen et al., 2013). Although the results were variable, it can be assumed that TTS may occur in fishes within the same hearing groups at similar exposure levels. As a conservative measure, the threshold for TTS from exposure to low-frequency sonar for all fish hearing groups with a swim bladder was rounded down to a cumulative sound exposure level of 210 dB re 1 μPa^2-s.

Criteria for high- and very-high-frequency sonar were not available in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014); however, only species with a swim bladder involved in hearing and with high-frequency specializations, such as shad, could potentially be affected. The majority of fish species within the Study Area are unlikely to be able to detect these sounds. There is little data available on hearing loss from exposure of fishes to these high-frequency sonars. Due to the lack of available data, and as a conservative measure, effects to these hearing groups from high-frequency sonars would utilize the lowest threshold available for other hearing groups (a cumulative sound exposure level of 210 dB re 1 μPa^2-s), but effects would largely be analyzed qualitatively.

3.9.2.1.2.2 Impact Ranges for Sonar and Other Transducers

The following section provides ranges to specific effects from sonar and other transducers. Ranges are calculated using criteria from Table 3.9-4 and the Navy Acoustic Effects Model. Only ranges to TTS were predicted based on available data. Sonar durations of 1, 30, 60 and 120 seconds were used to calculate the ranges below. However, despite the variation in exposure duration, ranges were almost identical across these durations and therefore were combined and summarized by bin in the table below. General source levels, durations, and other characteristics of these systems are described in Section 3.0.4.1.1 (Sonar and Other Transducers).

Table 3.9-4: Ranges to Temporary Threshold Shift from Four Representative Sonar Bins

Fish Hearing Group	*Range to Effects (meters)*			
	Sonar Bin LF4 Low-frequency	*Sonar Bin MF1 Hull-mounted surface ship sonars (e.g., AN/SQS-53C and AN/SQS-61)*	*Sonar Bin MF4 Helicopter-deployed dipping sonars (e.g., AN/AQS-22)*	*Sonar Bin MF5 Active acoustic sonobuoys (e.g., DICASS)*
Fishes without a swim bladder	NR	NR	NR	NR
Fishes with a swim bladder not involved in hearing	0	NR	NR	NR
Fishes with a swim bladder involved in hearing	0	7 (5–10)	0	0
Fishes with a swim bladder and high-frequency hearing	0	7 (5–10)	0	0

Notes: Ranges to TTS represent modeled predictions in different areas and seasons within the Study Area. The average range to TTS is provided as well as the minimum to the maximum range to TTS in parenthesis. Where only one number is provided the average, minimum, and maximum ranges to TTS are the same.
LF = low-frequency, MF = mid-frequency, NR = no criteria are available and therefore no range to effects are estimated.

3.9.2.1.2.3 Impacts from Sonar and Other Transducers Under Alternative 1

Sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. Use of sonar and other transducers would typically be transient and temporary. General categories and characteristics of sonar systems and the number of hours these sonars would be operated during training and testing activities under Alternative 1 are described in Section 3.0.4.1.1 (Sonar and Other Transducers). Activities using sonars and other transducers would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Training and Testing Activities Descriptions).

Under Alternative 1, training and testing activities including low-frequency sonars within most marine species hearing range (<2 kHz) would take place throughout the Study Area. Unit-level training and major training exercises would fluctuate each year to account for the natural variation of training cycles and deployment schedules. Some unit-level training would be conducted using synthetic means (e.g., simulators) or would be completed through other training exercises. Low-frequency sources are operated more frequently during testing activities than during training activities. Therefore, although the general impacts from sonar and other transducers during testing would be similar in severity to those described during training, there may be more impacts during testing activities as all marine fishes can detect low-frequency sources.

Only a few species of shad within the Clupeidae family, subfamily Alosinae, are known to be able to detect high-frequency sonar and other transducers (greater than 10 kHz) and are considered a part of the fish hearing group for species with a swim bladder that have high-frequency hearing. However, these species are not present in the MITT Study Area. Other marine fishes would probably not detect these sounds and therefore would not experience masking, physiological stress, or behavioral disturbance from exposure to high or very high-frequency sonar and other transducers.

Most marine fish species are not expected to detect sounds in the mid-frequency range (above a few kHz) of most operational sonars. The fish species that are known to detect mid-frequencies (i.e., those with swim bladders including some sciaenids [drum], most clupeids [herring, shad], and potentially deep-water fish such as myctophids [lanternfish]) do not have their best sensitivities in the range of the operational sonars (see Figure 3.9-1). Thus, fishes may only detect the most powerful systems, such as hull-mounted sonar, within a few kilometers; and most other, less powerful mid-frequency sonar systems, for a kilometer or less. Fishes with a swim bladder involved in hearing and with high-frequency hearing are more susceptible to hearing loss due to exposure to mid-frequency sonars. However, as shown in Table 3.9-4, the maximum estimated range to TTS for these fish hearing groups is equal to or less than 10 m for only the most powerful sonar bins. Fishes within these hearing groups would have to be very close to the source and the source levels would have to be relatively high in order to experience this effect.

Most mid-frequency active sonars used in the Study Area would not have the potential to substantially mask key environmental sounds or produce sustained physiological stress or behavioral reactions due to the limited time of exposure due to the moving sound sources and variable duty cycles. However, it is important to note that some mid-frequency sonars have a high duty cycle or are operated continuously. This may increase the risk of masking but only for important biological sounds that overlap with the frequency of the sonar being operated. Furthermore, although some species may be able to produce sound at higher frequencies (greater than 1 kHz), vocal marine fishes, such as sciaenids, largely communicate below the range of mid-frequency levels used by most sonars. Any such effects would be temporary and infrequent as a vessel operating mid-frequency sonar transits an area. As such, mid-frequency sonar use is unlikely to impact individuals. Long-term consequences for fish populations due to exposure to mid-frequency sonar and other transducers are not expected.

All marine fish species can likely detect low-frequency sonars and other transducers. However, low-frequency active sonar use is rare and most low-frequency active operations are typically conducted in deeper, offshore areas. The majority of fish species, including those that are the most highly vocal, exist on the continental shelf and within nearshore, estuarine areas. However, some species may still be present in areas where low-frequency sonar and other transducers are used, including some coastal areas. Most low-frequency sonar sources do not have a high enough source level to cause TTS, as shown in Table 3.9-4. Although highly unlikely, if TTS did occur, it may reduce the detection of biologically significant sounds but would likely recover within a few minutes to days.

The majority of fish species exposed to sonar and other transducers within near (tens of meters) to far (thousands of meters) distances of the source would be more likely to experience; mild physiological stress; brief periods of masking; behavioral reactions such as startle or avoidance responses, although risk would be low even close to the source; or no reaction. However, based on the information provided in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), the relative risk of these effects at any distance are expected to be low. Due to the transient nature of most sonar operations, overall effects would be localized and infrequent, only lasting a few seconds or minutes. Based on the low level and short duration of potential exposure to low-frequency sonar and other transducers, long-term consequences for fish populations are not expected.

As discussed previously in Section 3.9.1.1 (Hearing and Vocalization) and as shown in Figure 3.9-1, all ESA-listed fish species that occur in the Study Area are capable of detecting sound produced by low-frequency sonars and other transducers. However, scalloped hammerhead sharks, giant manta rays and oceanic whitetip sharks do not have a swim bladder and cannot detect frequencies above 2 kHz therefore impacts from mid-, high- or very high-frequency sonar and other transducers are not expected for any ESA-listed species. The Indo-West Pacific Distinct Population Segment of scalloped hammerhead could occur in nearshore waters, such as bays and estuaries, but is also known to occur in offshore portions of the Study Area. The giant manta ray and oceanic whitetip shark would most likely be exposed to low-frequency sonar in offshore areas throughout the Study Area.

Impacts on ESA-listed fishes, if they occur, would be similar to impacts on fishes in general. As described above, most low-frequency sonar sources do not have a high enough source level to cause TTS, and TTS would not be anticipated in fishes without a swim bladder. Although some shark species have shown attraction to irregularly pulsed low-frequency sounds (below several hundred Hz), they are not known to be attracted to continuous signals or higher frequencies that they presumably cannot hear (Casper & Mann, 2006; Casper & Mann, 2009; Casper et al., 2012a). Due to the short-term, infrequent, and localized nature of these activities, ESA-listed fishes are unlikely to be exposed multiple times within a short period. In addition, physiological and behavioral reactions would be expected to be brief (e.g., seconds to minutes) and infrequent based on the low probability of co-occurrence between training activities and these species. Although individuals may be impacted, long-term consequences for populations would not be expected.

Pursuant to the ESA, the use of sonar and other transducers during training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays.

3.9.2.1.2.4 Impacts from Sonar and Other Transducers Under Alternative 2 (Preferred Alternative)

Sonar and other transducers emit sound waves into the water to detect objects, safely navigate, and communicate. Use of sonar and other transducers would typically be transient and temporary. General categories and characteristics of sonar systems and the number of hours these sonars would be operated during training and testing activities under Alternative 2 are described in Section 3.0.4.1.1 (Sonar and Other Transducers). Activities using sonars and other transducers would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Training and Testing Activities Descriptions).

Under Alternative 2, training and testing activities could occur throughout the Study Area. Training activities include the same type and tempo of training activities as Alternative 1 but also considers additional Fleet exercises (e.g., Valiant Shield type event) every year. Alternative 2 reflects the maximum number of training events that could occur within a given year, and assumes that the maximum number of Fleet exercises would occur every year. However, the types and tempo of testing activities would be the same as those conducted under Alternative 1.

Compared to training and testing activities that use sonar and other transducers that were previously analyzed in the 2015 MITT Final EIS/OEIS under Alternative 2, some training and testing activities would increase, decrease, or stay the same from those currently conducted (see Table 2.5-1 and Table 2.5-2 for details).

Impacts on fishes due to sonar and other transducers are expected to be limited to minor behavioral responses, short-term physiological stress, and brief periods of masking (seconds to minutes at most) for individuals; long-term consequences for individuals and therefore populations would not be expected. Predicted impacts on ESA-listed fish species would not be discernible from those described above in Section 3.9.2.1.2.3 (Impacts from Sonar and Other Transducers under Alternative 1).

Pursuant to the ESA, the use of sonar and other transducers during training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.1.2.5 Impacts from Sonar and Other Transducers Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Acoustic stressors as listed above would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer sonar and other transducers within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for acoustic impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.1.3 Impacts from Vessel Noise

Fishes may be exposed to noise from vessel movement. A detailed description of the acoustic characteristics and typical sound levels of vessel noise are in Section 3.0.4.1.2 (Vessel Noise). Vessel movements involve transits to and from ports to various locations within the Study Area, including commercial ship traffic as well as recreational vessels in addition to U.S. Navy vessels. Many ongoing and proposed training and testing activities within the Study Area involve maneuvers by various types of surface ships, boats, and submarines (collectively referred to as vessels). Activities may vary from those previously analyzed in the 2015 MITT Final EIS/OEIS, but the overall determinations presented remain valid. Increases and decreases shown in Table 2.5-1 and Table 2.5-2 for proposed activities under Alternative 1 and 2 do not appreciably change the impact conclusions presented in the 2015 MITT Final EIS/OEIS.

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Acoustic stressors, as described above, would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in less vessel noise within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for acoustic impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

Pursuant to the ESA, sound produced by vessel movement during training and testing activities, as described under Alternative 1 and Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays. The Navy is consulting with NMFS regarding Alternative 2 as required by Section 7(a)(2) of the ESA.

3.9.2.1.4 Impacts from Aircraft Noise

Fishes that occur near or at the water's surface may be exposed to aircraft noise, although this is considered to be unlikely. Fixed, rotary-wing, and tilt-rotor aircraft are used during a variety of training and testing activities throughout the Study Area. Tilt-rotor impacts would be similar to fixed-wing or rotary-wing (i.e., helicopter) impacts depending which mode the aircraft is in. Most of these sounds would be concentrated around airbases and fixed ranges within the range complex. Aircraft noise could also occur in the waters immediately surrounding aircraft carriers at sea during takeoff and landing. 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 (helicopters) produce low-frequency sound and vibration (Pepper et al., 2003). A detailed description of aircraft noise as a stressor is in Section 3.0.4.1.3 (Aircraft Noise).

Activities may vary from those previously analyzed in the 2015 MITT Final EIS/OEIS. The analysis of impacts from aircraft noise in this Supplemental EIS/OEIS supplants the 2015 MITT Final EIS/OEIS for fishes, and changes estimated impacts for some species since the 2015 MITT Final EIS/OEIS.

3.9.2.1.4.1 Methods for Analyzing Impacts from Aircraft Noise

The amount of sound entering the ocean from aircraft would be very limited in duration, sound level, and affected area. Due to the low level of sound that could enter the water from aircraft activities, hearing loss is not further considered as a potential effect. Potential impacts considered are masking of other biologically relevant sounds, physiological stress, and changes in behavior. Reactions by fishes to these specific stressors have not been recorded; however, fishes would be expected to react to aircraft noise as they would react to other transient sounds (e.g., sonar or vessel noise).

For this analysis, the Navy assumes that some fish at or near the water surface may exhibit startle reactions to certain aircraft noise if aircraft altitude is low. This could mean a hovering helicopter, for which the sight of the aircraft and water turbulence could also cause a response, or a low-flying or super-sonic aircraft generating enough noise to be briefly detectable underwater or at the air-water interface. Because any fixed-wing aircraft noise would be brief, the risk of masking any sounds relevant to fishes is very low. The *ANSI Sound Exposure Guidelines* for fishes did not consider this acoustic stressor (Popper et al., 2014).

3.9.2.1.4.2 Impacts from Aircraft Noise Under Alternative 1

Fishes may be exposed to aircraft-generated noise throughout the Study Area. Characteristics of aircraft noise and the number of training and testing events that include aircraft under Alternative 1 are shown in Section 3.0.4.1.3 (Aircraft Noise). Activities with aircraft would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Training and Testing Activities Descriptions). Aircraft training and testing activities would usually occur adjacent to Navy airfields, installations, and in special use airspace within the Study Area and transit corridor.

Under Alternative 1, activities may vary from those previously analyzed in the 2015 MITT Final EIS/OEIS. Increases and decreases shown in Table 2.5-1 and Table 2.5-2 for proposed activities under Alternative 1 and 2.

In most cases, exposure of fishes to fixed-wing aircraft presence and noise would be brief as the aircraft quickly passes overhead. Fishes would have to be at or near the surface at the time of an overflight to be exposed to appreciable sound levels. Due to the low sound levels in water, it is unlikely that fishes would respond to most fixed-wing aircraft or transiting helicopters. Because most overflight exposure would be brief and aircraft noise would be at low received levels, only startle reactions, if any, are expected in response to low altitude flights. Similarly, the brief duration of most overflight exposures would limit any potential for masking of relevant sounds.

Daytime and nighttime activities involving helicopters may occur for extended periods of time, up to a couple of hours in some areas. During these activities, helicopters would typically transit throughout an area but could also hover over the water. Longer event durations and periods of time where helicopters hover may increase the potential for behavioral reactions, startle reactions, masking, and physiological stress. Low-altitude flights of helicopters during some activities, which often occur under 100 feet (ft.) altitude, may elicit a stronger startle response due to the proximity of a helicopter to the water; the slower airspeed and longer exposure duration; and the downdraft created by a helicopter's rotor.

If fish were to respond to aircraft noise, only short-term behavioral or physiological reactions (e.g., avoidance and increased heart rate) would be expected. Therefore, long-term consequences for individuals would be unlikely and long-term consequences for populations are not expected.

All ESA-listed species that occur in the Study Area are likely capable of detecting aircraft noise as discussed previously in Section 3.9.1.1 (Hearing and Vocalization) and could be exposed to aircraft noise throughout the Study Area. However, due to the small area within which sound could potentially enter the water and the extremely brief window the sound could be present, exposures of fishes to aircraft noise would be extremely rare, and in the event that they did occur, would be very brief (seconds).

Pursuant to the ESA, sound produced by aircraft movement during training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays.

3.9.2.1.4.3 Impacts from Aircraft Noise Under Alternative 2 (Preferred Alternative)

Characteristics of aircraft noise and the number of training and testing events that include aircraft under Alternative 2 are shown in Section 3.0.4.1.3 (Aircraft Noise). Activities with aircraft would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Training and Testing Activities Descriptions). Aircraft training and testing activities would usually occur adjacent to Navy airfields, installations, and in special use airspace within the Study Area and transit corridor.

Under Alternative 2, activities may vary from those previously analyzed in the 2015 MITT Final EIS/OEIS. Increases and decreases shown in Table 2.5-1 and Table 2.5-2 for proposed activities under Alternative 1 and 2.

Activities under Alternative 2 include a minor increase in the number of events that involve aircraft as compared to Alternative 1; however, the training locations, aircraft, and general types of predicted impacts would not be discernible from those described above in Section 3.9.2.1.4.2 (Impacts from Aircraft Noise Under Alternative 1).

Pursuant to the ESA, sound produced by aircraft movement during training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.1.4.4 Impacts from Aircraft Noise Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Acoustic stressors, as described above, would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in less acoustic stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for acoustic impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.1.5 Impacts from Weapon Noise

Fishes may be exposed to sounds caused by the firing of weapons, objects in flight, and 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 (such as those discussed under Section 3.0.4.2, Explosive Stressors) 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. Due to the transient nature of most activities that produce weapon noise, overall effects would be localized and infrequent, only lasting a few seconds or minutes. Reactions by fishes to these specific stressors have not been recorded however, fishes would be expected to react to weapon noise as they would react to other transient sounds (e.g., sonar or vessel noise).

Activities may vary from those previously analyzed in the 2015 MITT Final EIS/OEIS, but the overall determinations presented remain valid. Increases and decreases shown in Table 2.5-1 and Table 2.5-2 for activities proposed under Alternative 1 and 2 do not appreciably change the impact conclusions presented in the 2015 MITT Final EIS/OEIS.

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Acoustic stressors, as described above, would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in less acoustic stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for acoustic impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

Pursuant to the ESA, sound produced by weapon noise during training and testing activities, as described under Alternative 1 and Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays. The Navy is consulting with NMFS regarding Alternative 2 as required by Section 7(a)(2) of the ESA.

3.9.2.2 Explosive Stressors

Explosions in the water or near the water surface can introduce loud, impulsive, broadband sounds into the marine environment. However, unlike other acoustic stressors, explosives release energy at a high rate producing a shock wave that can be injurious and even deadly. Therefore, explosive impacts on fishes are discussed separately from other acoustic stressors, even though the analysis of explosive impacts will rely on data for fish impacts due to impulsive sound exposure where appropriate.

Explosives are usually described by their net explosive weight, which accounts for the weight and type of explosive material. Additional explanation of the acoustic and explosive terms and sound energy concepts used in this section is found in Appendix H (Acoustic and Explosive Concepts).

This section begins with a summary of relevant data regarding explosive impacts on fishes in Section 3.9.2.2.1 (Background). 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 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), and this section follows that framework.

Although air guns and pile driving are not used during MITT training and testing activities, the analysis of some explosive impacts will in part rely on data from fishes exposed to impulsive sources where appropriate. Impulsive sources are further discussed below when applicable data are available for comparison purposes. In addition, there are limited studies of fish responses to weapon noise. For the purposes of this analysis, studies of the effects from air guns, pile driving, and explosives are used to inform fish responses to other impulsive sources (i.e., weapon noise).

Due to the availability of new literature, adjusted sound exposure criteria, and new acoustic effects modeling, the analysis provided in Section 3.9.2.2.2 (Impacts from Explosives) of this SEIS/OEIS supplants the 2015 MITT Final EIS/OEIS for fishes.

3.9.2.2.1 Background

The effects of explosions on fishes have been studied and reviewed by numerous authors (Keevin & Hempen, 1997; O'Keeffe, 1984; O'Keeffe & Young, 1984; Popper et al., 2014). A summary of the literature related to each type of effect forms the basis for analyzing the potential effects from Navy activities. The sections below include a survey and synthesis of best-available science published in peer-reviewed journals, technical reports, and other scientific sources pertinent to impacts on fishes potentially resulting from Navy training and testing activities. Fishes could be exposed to a range of impacts depending on the explosive source and context of the exposure. In addition to acoustic impacts including temporary or permanent hearing loss, auditory masking, physiological stress, or changes in behavior, potential impacts from an explosive exposure can include non-lethal injury and mortality.

3.9.2.2.1.1 Injury

Injury refers to the direct effects on the tissues or organs of a fish. The blast wave from an in-water explosion is lethal to fishes at close range, causing massive organ and tissue damage (Keevin & Hempen, 1997). At greater distance from the detonation point, the extent of mortality or injury depends on a number of factors, including fish size, body shape, depth, physical condition of the fish, and perhaps most importantly, the presence of a swim bladder (Keevin & Hempen, 1997; Wright, 1982; Yelverton et al., 1975; Yelverton & Richmond, 1981). At the same distance from the source, larger fishes are generally less susceptible to death or injury, elongated forms that are round in cross-section are less at risk than deep-bodied forms, and fishes oriented sideways to the blast suffer the greatest impact (Edds-Walton & Finneran, 2006; O'Keeffe, 1984; O'Keeffe & Young, 1984; Wiley et al., 1981; Yelverton et al., 1975). Species with a swim bladder are much more susceptible to blast injury from explosives than fishes without them (Gaspin, 1975; Gaspin et al., 1976; Goertner et al., 1994).

If a fish is close to an explosive detonation, the exposure to rapidly changing high pressure levels can cause barotrauma. Barotrauma is injury due to a sudden difference in pressure between an air space inside the body and the surrounding water and tissues. Rapid compression followed by rapid expansion of airspaces, such as the swim bladder, can damage surrounding tissues and result in the rupture of the airspace itself. The swim bladder is the primary site of damage from explosives (Wright, 1982; Yelverton et al., 1975). Gas-filled swim bladders resonate at different frequencies than surrounding tissue and can be torn by rapid oscillation between high- and low-pressure waves (Goertner, 1978). Swim bladders are a characteristic of most bony fishes with the notable exception of flatfishes (e.g., halibut). Sharks and rays are examples of fishes without a swim bladder. Small airspaces, such as micro-bubbles that may be present in gill structures, could also be susceptible to oscillation when exposed to the rapid pressure increases caused by an explosion. This may have caused the bleeding observed on gill structures of some fish exposed to explosions (Goertner et al., 1994). Sudden very high pressures can also cause damage at tissue interfaces due to the way pressure waves travel differently through tissues with different densities. Rapidly oscillating pressure waves might rupture the kidney, liver, spleen, and sinus and cause venous hemorrhaging (Keevin & Hempen, 1997).

Several studies have exposed fish to explosives and examined various metrics in relation to injury susceptibility. Sverdrup (1994) exposed Atlantic salmon (1–1.5 kg [2–3 lb.]) in a laboratory setting to repeated shock pressures of around 2 megapascals (300 pounds per square inch [psi]) without any immediate or delayed mortality after a week. Hubbs and Rechnitzer (1952) showed that fish with swim bladders exposed to explosive shock fronts (the near-instantaneous rise to peak pressure) were more susceptible to injury when several feet below the water surface than near the bottom. When near the surface, the fish began to exhibit injuries around peak pressure exposures of 40 to 70 psi. However, near the bottom (all water depths were less than 100 ft.) fish exposed to pressures over twice as high exhibited no sign of injury. Yelverton et al. (1975) similarly found that peak pressure was not correlated to injury susceptibility; instead, injury susceptibility of swim bladder fish at shallow depths (10 ft. or less) was correlated to the metric of positive impulse (pascal seconds [Pa-s]), which takes into account both the positive peak pressure, the duration of the positive pressure exposure, and the fish mass, with smaller fish being more susceptible.

Gaspin et al. (1976) exposed multiple species of fish with a swim bladder, placed at varying depths, to explosive blasts of varying size and depth. Goertner (1978) and Wiley (1981) developed a swim bladder oscillation model, which showed that the severity of injury observed in those tests could be correlated to the extent of swim bladder expansion and contraction predicted to have been induced by exposure to the explosive blasts. Per this model, the degree of swim bladder oscillation is affected by ambient pressure (i.e., depth of fish), peak pressure of the explosive, duration of the pressure exposure, and exposure to surface rarefaction (negative pressure) waves. The maximum potential for injury is predicted to occur where the surface reflected rarefaction (negative) pressure wave arrives coincident with the moment of maximum compression of the swim bladder caused by exposure to the direct positive blast pressure wave, resulting in a subsequent maximum expansion of the swim bladder. Goertner (1978) and Wiley et al. (1981) found that their swim bladder oscillation model explained the injury data in the Yelverton et al. (1975) exposure study, and their impulse parameter was applicable only to fishes at shallow enough depths to experience less than one swim bladder oscillation before being exposed to the following surface rarefaction wave.

O’Keeffe (1984) provides calculations and contour plots that allow estimation of the range to potential effects of in-water explosions on fish possessing swim bladders using the damage prediction model developed by Goertner (1978). O’Keeffe’s (1984) parameters include the charge weight, depth of burst, and the size and depth of the fish, but the estimated ranges do not take into account unique propagation environments that could reduce or increase the range to effect. The 10 percent mortality range shown below in Table 3.9-5 is the maximum horizontal range predicted by O'Keeffe (1984) for 10 percent of fish suffering injuries that are expected to not be survivable (e.g., damaged swim bladder or severe hemorrhaging). Fish at greater depths and near the surface are predicted to be less likely to be injured because geometries of the exposures would limit the amplitude of swim bladder oscillations. In addition, detonations at or near the surface (i.e., similar to most Navy activities that utilize bombs and missiles) would result in energy loss at the water air interface, resulting in lower overall ranges to effect than those predicted here.

In contrast to fish with swim bladders, fishes without swim bladders have been shown to be more resilient to explosives (Gaspin, 1975; Gaspin et al., 1976; Goertner et al., 1994). For example, some small (average 116 mm length; approximately 1 oz.) hogchokers (*Trinectes maculatus*) exposed less than 5 ft. from a 10 lb. pentolite charge immediately survived the exposure with slight to moderate injuries, and only a small number of fish were immediately killed; however, most of the fish at this close range did suffer moderate to severe injuries, typically of the gills or around the otolithic structures (Goertner et al., 1994).

Studies that have documented caged fishes killed during planned underwater explosions indicate that most fish that die do so within one to four hours, and almost all die within a day (Yelverton et al., 1975). Mortality in free-swimming (uncaged) fishes may be higher due to increased susceptibility to predation. Fitch and Young (1948) found that the type of free-swimming fish killed changed when blasting was repeated at the same location within 24 hours of previous blasting. They observed that most fish killed on the second day were scavengers, presumably attracted by the victims of the previous day’s blasts.

Table 3.9-5: Range to 10 Percent Mortality from In-water Explosions for Fishes with a Swim Bladder

Weight of Pentolite (lb.) [NEW, lb.][1]	*Depth of Explosion (ft.) [m]*	*10% Mortality Maximum Range (ft.) [m]*		
		1 oz. Fish	*1 lb. Fish*	*30 lb. Fish*
10 [13]	10 [3]	530 [162]	315 [96]	165 [50]
	50 [15]	705 [214]	425 [130]	260 [79]
	200 [61]	905 [276]	505 [154]	290 [88]
100 [130]	10 [3]	985 [300]	600 [183]	330 [101]
	50 [15]	1,235 [376]	865 [264]	590 [180]
	200 [61]	1,340 [408]	1,225 [373]	725 [221]
1,000 [1,300]	10 [3]	1,465 [447]	1,130 [344]	630 [192]
	50 [15]	2,255 [687]	1,655 [504]	1,130 [344]
	200 [61]	2,870 [875]	2,390 [728]	1,555 [474]
10,000 [13,000]	10 [3]	2,490 [759]	1,920 [585]	1,155 [352]
	50 [15]	4,090 [1,247]	2,885 [879]	2,350 [716]
	200 [61]	5,555 [1,693]	4,153 [1,266]	3,090 [942]

[1] Explosive weights of pentolite converted to net explosive weight using the peak pressure parameters in Swisdak (1978).

Notes: ft. = feet, lb. = pounds, m = meters, NEW = net explosive weight, oz. = ounce

Source: Data from O'Keeffe (1984)

Fitch and Young (1948) also investigated whether a significant portion of fish killed would have sunk and not been observed at the surface. Comparisons of the numbers of fish observed dead at the surface and at the bottom in the same affected area after an explosion showed that fish found dead on the bottom comprised less than 10 percent of the total observed mortality. Gitschlag et al. (2000) conducted a more detailed study of both floating fishes and those that were sinking or lying on the bottom after explosive removal of nine oil platforms in the northern Gulf of Mexico. Results were highly variable. They found that 3–87 percent (46 percent average) of the red snapper killed during a blast might float to the surface. Currents, winds, and predation by seabirds or other fishes may be some of the reasons that the magnitude of fish mortality may not have been accurately captured.

There have been few studies of the impact of underwater explosives on early life stages of fish (eggs, larvae, juveniles). Fitch and Young (1948) reported mortality of larval anchovies exposed to underwater blasts off California. Nix and Chapman (1985) found that anchovy and smelt larvae died following the detonation of buried charges. Similar to adult fishes, the presence of a swim bladder contributes to shock wave-induced internal damage in larval and juvenile fish (Settle et al., 2002). Explosive shock wave injury to internal organs of larval pinfish and spot exposed at shallow depths was documented by Settle et al. (2002) and Govoni et al. (2003; 2008) at impulse levels similar to those predicted by Yelverton et al. (1975) for very small fish. Settle et al. (2002) provide the lowest measured received level that injuries have been observed in larval fish. Researchers (Faulkner et al., 2006; Faulkner et al., 2008; Jensen, 2003) have suggested that egg mortality may be correlated with peak particle velocity exposure (i.e., the localized movement or shaking of water particles, as opposed to the velocity of the blast wave), although sufficient data from direct explosive exposures is not available (2003; 2008).

Rapid pressure changes could cause mechanical damage to sensitive ear structures due to differential movements of the otolithic structures. Bleeding near otolithic structures was the most commonly observed injury in non-swim bladder fish exposed to a close explosive charge (Goertner et al., 1994).

As summarized by the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), exposure to explosive energy poses the greatest potential threat for injury and mortality in marine fishes. Fishes with a swim bladder are more susceptible to injury than fishes without a swim bladder. The susceptibility also probably varies with size and depth of both the detonation and the fish. Fish larvae or juvenile fish may be more susceptible to injury from exposure to explosives.

3.9.2.2.1.2 Hearing Loss

There are no direct measurements of hearing loss in fishes 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. PTS in fish has not been known to occur in species tested to date and any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al., 2005; Popper et al., 2014; Smith et al., 2006).

As reviewed in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), fishes without a swim bladder, or fishes with a swim bladder not involved in hearing, would be less susceptible to hearing loss (i.e., TTS), even at higher level exposures. Fish with a swim bladder involved in hearing may be susceptible to TTS within very close ranges to an explosive. General research findings regarding TTS in fishes as well as findings specific to exposure to other impulsive sound sources are discussed in Section 3.9.2.1.1.2 (Hearing Loss).

3.9.2.2.1.3 Masking

Masking refers to the presence of a noise that interferes with a fish's ability to hear biologically important sounds, including those produced by prey, predators, or other fish in the same species (Myrberg, 1980; Popper et al., 2003). This can take place whenever the noise level heard by a fish exceeds the level of a biologically relevant sound. As discussed in Section 3.0.4.7 (Conceptual

Framework for Assessing Effects from Acoustic and Explosive Activities), masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise. Masking may lead to a change in vocalizations or a change in behavior (e.g., cessation of foraging, leaving an area).

There are no direct observations of masking in fishes due to exposure to explosives. The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) highlights a lack of data that exist for masking by explosives but suggests that the intermittent nature of explosions would result in very limited probability of any masking effects, and if masking were to occur it would only occur during the duration of the sound. General research findings regarding masking in fishes due to exposure to sound are discussed in detail in Section 3.9.2.1.1.3 (Masking). Potential masking from explosives is likely to be similar to masking studied for other impulsive sounds such as air guns.

3.9.2.2.1.4 Physiological Stress

Fishes 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 3.0.4.7 (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.

Research on physiological stress in fishes due to exposure to explosive sources is limited. Sverdrup et al. (1994) studied levels of stress hormones in Atlantic salmon after exposure to multiple detonations in a laboratory setting. Increases in cortisol and adrenaline were observed following the exposure, with adrenaline values returning to within normal range within 24 hours. General research findings regarding physiological stress in fishes due to exposure to acoustic sources are discussed in detail in Section 3.9.2.1.1.4 (Physiological Stress). Generally, stress responses are more likely to occur in the presence of potentially threatening sound sources such as predator vocalizations or the sudden onset of impulsive signals. Stress responses may be brief (a few seconds to minutes) if the exposure is short or if fishes habituate or learn to tolerate the noise. It is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

3.9.2.2.1.5 Behavioral Reactions

As discussed in Section 3.0.4.7 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), any stimuli in the environment can cause a behavioral response in fishes, including sound and energy produced by explosions. Behavioral reactions of fishes to explosions have not been recorded. Behavioral reactions from explosive sounds are likely to be similar to reactions studied for other impulsive sounds such as those produced by air guns. 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 or avoidance responses. General research findings regarding behavioral reactions from fishes due to exposure to impulsive sounds, such as those associated with explosions, are discussed in detail in Section 3.9.2.1.1.5 (Behavioral Reactions).

As summarized by the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), species may react differently to the same sound source depending on a number of variables, such as the animal's life stage or behavioral state (e.g., feeding, mating). Without data that are more specific it is assumed that fishes with similar hearing capabilities react similarly to all impulsive sounds outside or within the zone for hearing loss and injury. Observations of fish reactions to large-scale air gun surveys are informative, but not necessarily directly applicable to analyzing impacts from the short-term, intermittent use of all impulsive sources. Fish have a higher probability of reacting when closer to an impulsive sound source (within tens of meters), and a decreasing probability of reaction at increasing distances (Popper et al., 2014).

3.9.2.2.1.6 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 3.0.4.7 (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 temporary hearing impairment or chronic masking, which could affect 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 fish species that live for multiple seasons or years. 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.

3.9.2.2.2 Impacts from Explosives

This section analyzes the impacts on fishes due to in-water and in-air explosives that would be used during Navy training and testing activities, synthesizing the background information presented above.

As discussed above, sound and energy from in-water explosions are capable of causing mortality, injury, temporary hearing loss, masking, physiological stress, or a behavioral response, 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 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 affect its ability to reproduce. Temporary threshold shift can also impair an animal’s abilities, although the individual may recover quickly with little significant effect.

The overall use of explosives for training and testing activities would be similar to what is currently conducted and several new testing activities would occur (see Table 2.5-1 and Table 2.5-2 for details). Although individual activities may vary some from those previously analyzed, the overall determinations presented in the 2015 MITT Final EIS/OEIS remain valid, but have been improved upon under this SEIS/OEIS.

3.9.2.2.2.1 Methods for Analyzing Impacts from Explosives

The Navy performed a quantitative analysis to estimate ranges to effect for fishes exposed to underwater explosives during Navy training and testing activities. Inputs to the quantitative analysis included sound propagation modeling in the Navy Acoustic Effects Model to the sound exposure criteria and thresholds presented below. Density data for fish species within the Study Area are not currently available; therefore, it is not possible to estimate the total number of individuals that may be affected by explosive activities.

Criteria and Thresholds used to Estimate Impacts on Fishes from Explosives

Mortality and Injury from Explosives

Criteria and thresholds to estimate impacts from sound and energy produced by explosive activities are presented in Table 3.9-6. In order to estimate the longest range at which a fish may be killed or mortally injured, the Navy based the threshold for mortal injury on the lowest pressure that caused mortalities in the study by Hubbs and Rechnitzer (1952), consistent with the recommendation in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014). As described in Section 3.9.2.2.1.1 (Injury), this threshold likely overestimates the potential for mortal injury. The potential for mortal injury has been shown to be correlated to fish size, depth, and geometry of exposure, which are not accounted for by using a peak pressure threshold. However, until fish mortality models are developed that can reasonably consider these factors across multiple environments, use of the peak pressure threshold allows for a conservative estimate of maximum impact ranges.

Due to the lack of detailed data for onset of injury in fishes exposed to explosives, thresholds from impact pile driving exposures (Halvorsen et al., 2011; Halvorsen et al., 2012a; Halvorsen et al., 2012b) were used as a proxy for the analysis in the Atlantic Fleet Training and Testing FEIS/OEIS (U.S. Department of the Navy, 2018). Upon re-evaluation, it was decided that pile driving thresholds are too conservative and not appropriate to use in the analysis of explosive effects on fishes. Therefore, injury criteria have been revised as follows.

Thresholds for the onset of injury from exposure to explosions are not currently available and recommendations in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) only provide qualitative criteria for consideration. Therefore, available data from existing explosive studies were reviewed to provide a conservative estimate for a threshold to the onset of injury (Gaspin, 1975; Gaspin et al., 1976; Hubbs & Rechnitzer, 1952; Settle et al., 2002; Yelverton et al., 1975).

It is important to note that some of the available literature is not peer-reviewed and may have some caveats to consider when reviewing the data (e.g., issues with controls, limited details on injuries observed, etc.) but this information may still provide a better understanding of where injurious effects would begin to occur specific to explosive activities. The lowest threshold at which injuries were observed in each study were recorded and compared for consideration in selecting criteria. As a conservative measure, the absolute lowest peak sound pressure level recorded that resulted in injury, observed in exposures of larval fishes to explosions (Settle et al., 2002), was selected to represent the threshold to injury.

The injury threshold is consistent across all fish regardless of hearing groups due to the lack of rigorous data for multiple species. As discussed throughout Section 3.9.2.2.1.1 (Injury), it is important to note that these thresholds may be overly conservative, as there is evidence that fishes exposed to higher thresholds than those in Table 3.9-6 have shown no signs of injury (depending on variables such as the weight of the fish, size of the explosion, and depth of the cage (Gaspin, 1975; Gaspin et al., 1976; Hubbs & Rechnitzer, 1952; Settle et al., 2002; Yelverton et al., 1975). It is likely that adult fishes and fishes without a swim bladder would be less susceptible to injury than more sensitive hearing groups (i.e., fishes with a swim bladder) and larval fish.

Table 3.9-6: Sound Exposure Criteria for Mortality and Injury from Explosives

Fish Hearing Group	*Onset of Mortality*	*Onset of Injury*
	SPL_{peak}	*SPL_{peak}*
Fishes without a swim bladder	229	220
Fishes with a swim bladder not involved in hearing	229	220
Fishes with a swim bladder involved in hearing	229	220
Fishes with a swim bladder and high-frequency hearing	229	220

Note: SP_{Lpeak} = Peak sound pressure level.

The number of fish killed by an in-water explosion would depend on the population density near the blast, as well as factors discussed throughout Section 3.9.2.2.1.1 (Injury) such as net explosive weight, depth of the explosion, and fish size. For example, if an explosion occurred in the middle of a dense school of fish, a large number of fish could be killed. However, the probability of this occurring is low, based on the patchy distribution of dense schooling fish. Stunning from pressure waves could also temporarily immobilize fish, making them more susceptible to predation.

Fragments produced by exploding munitions at or near the surface may present a high-speed strike hazard for an animal at or near the surface. In water, however, fragmentation velocities decrease rapidly due to drag (Swisdak & Montanaro, 1992). Because blast waves propagate efficiently through water, the range to injury from the blast wave would likely extend beyond the range of fragmentation risk.

Hearing Loss from Explosives

Criteria and thresholds to estimate TTS from sound produced by explosive activities are presented below in Table 3.9-7. Direct (measured) TTS data from explosives are not available. Criteria used to define TTS from explosives is derived from data on fishes exposed to seismic air gun signals (Popper et al., 2005) as summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014). TTS has not been documented in fishes without a swim bladder from exposure to other impulsive sources (pile driving and air guns). Although it is possible that fishes without a swim bladder could receive TTS from exposure to explosives, fishes without a swim bladder are typically less susceptible to hearing impairment than fishes with a swim bladder. If TTS occurs in fishes without a swim bladder, it would likely occur within the range of injury; therefore, no thresholds for TTS are proposed. General research findings regarding hearing loss in fishes as well as findings specific to exposure to other impulsive sound sources are discussed in Section 3.9.2.2.1.2 (Hearing Loss).

As discussed in Section 3.9.2.2.1.2 (Hearing Loss), exposure to sound produced from seismic air guns at a cumulative sound exposure level of 186 dB re 1 μPa^2-s has resulted in TTS in fishes with a swim bladder involved in hearing (Popper et al., 2005). TTS has not occurred in fishes with a swim bladder not involved in hearing and would likely occur above the given threshold in Table 3.9-7.

Table 3.9-7: Sound Exposure Criteria for Hearing Loss from Explosives

Fish Hearing Group	***TTS (SEL_{cum})***
Fishes without a swim bladder	NC
Fishes with a swim bladder not involved in hearing	> 186
Fishes with a swim bladder involved in hearing	186
Fishes with a swim bladder and high-frequency hearing	186

Notes: TTS = Temporary Threshold Shift, SEL_{cum} = Cumulative sound exposure level (decibel referenced to 1 micropascal squared seconds [dB re 1 µPa2-s]), NC = no criteria are reported, “>” indicates that the given effect would occur above the reported threshold.

3.9.2.2.2.2 Impact Ranges for Explosives

The following section provides estimated range to effects for fishes exposed to sound and energy produced by explosives. Ranges are calculated using criteria from Table 3.9-6 and Table 3.9-7 and the Navy Acoustic Effects Model. Fishes within these ranges would be predicted to receive the associated effect. Ranges may vary greatly depending on factors such as the cluster size, location, depth, and season of the event.

Table 3.9-8 provides range to mortality and injury for all fishes. Only one table (Table 3.9-9) is provided for range to TTS for all fishes with a swim bladder. However, ranges to TTS for fishes with a swim bladder not involved in hearing would be shorter than those reported because this effect has not been observed in fishes without a swim bladder exposed to the described TTS threshold.

3.9.2.2.2.3 Impacts from Explosives Under Alternative 1

The following section provides estimated range to effects for fishes exposed to sound and energy produced by explosives. Ranges are calculated using criteria from Table 3.9-6 and Table 3.9-7 and the Navy Acoustic Effects Model. Most detonations conducted during Navy activities would occur at or near the surface. The Navy Acoustic Effects Model cannot account for the highly non-linear effects of cavitation and surface blow off; therefore, some estimated ranges may be overly conservative. Fishes within these ranges would be predicted to receive the associated effect. Ranges may vary greatly depending on factors such as the cluster size (the number of rounds fired [or buoys dropped] within a very short duration), location, depth, and season of the event.

Table 3.9-8: Range to Mortality and Injury for All Fishes from Explosives

Bin[1]	***Range to Effects (meters)***	
	Onset of Mortality	***Onset of Injury***
	SPL_{peak}	***SPL_{peak}***
E1	50 (45–50)	122 (120–130)
E2	63 (60–65)	156 (110–170)
E3	108 (100–110)	276 (260–280)
E4	141 (140–170)	381 (350–725)
E5	175 (170–250)	433 (410–775)
E6	218 (210–230)	526 (500–625)
E7	330 (330–330)	856 (825–875)
E8	375 (360–410)	920 (850–1,025)
E9	490 (480–500)	1,025 (1,025–1,025)
E10	617 (600–775)	1,388 (1,275–1,775)
E11	785 (700–1,525)	2,111 (1,525–4,775)
E12	770 (750–800)	1,781 (1,775–2,025)

[1] Bin (net explosive weight, lb.): E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: SPLpeak = Peak sound pressure level, NEW = net explosive weight, lb. = pound(s). Range to effects represent modeled predictions in different areas and seasons within the Study Area. Each cell contains the estimated average, minimum and maximum range to the specified effect.

Table 3.9-9: Range to TTS for Fishes with a Swim Bladder from Explosives

Bin[1]	*Cluster Size*	*Range to Effects (meters)*
		TTS
		SEL_{cum}
E1	1	< 50 (45–55)
	18	< 196 (160–230)
E2	1	< 58 (55–60)
E3	1	< 127 (95–160)
	19	< 474 (340–600)
E4	1	< 204 (190–300)
E5	1	< 172 (150–450)
	20	< 674 (525–2,775)
E6	1	< 210 (190–390)
E7	1	< 634 (600–725)
E8	1	< 527 (310–775)
E9	1	< 513 (420–1,025)
E10	1	< 685 (525–1,775)
E11	1	< 1,679 (1,525–2,775)
E12	1	< 815 (675–2,025)

[1] Bin (net explosive weight, lb.): E0 (< 0.1), E1 (0.1 – 0.25), E2 (> 0.25 – 0.5), E3 (> 0.5 – 2.5), E4 (> 2.5 – 5), E5 (> 5 – 10), E7 (> 20 – 60), E8 (> 60 – 100), E10 (> 250 – 500), E11 (> 500 – 650)

Notes: SELcum = Cumulative sound exposure level, TTS = Temporary Threshold Shift, NEW = net explosive weight, lb. = pound(s), "<" indicates that the given effect would occur at distances less than the reported range(s). Range to effects represent modeled predictions in different areas and seasons within the Study Area. Each cell contains the estimated average, minimum and maximum range to the specified effect.

Under Alternative 1, there could be fluctuation in the amount of explosions that could occur annually, although potential impacts would be similar from year to year. The number of impulsive sources in this SEIS/OEIS compared with the totals analyzed in the 2015 MITT Final EIS/OEIS are described in Table 2.5-1 and Table 2.5-2. The number of torpedo testing activities (both explosive and non-explosive) planned under Alternative 1 testing can vary from year to year; however, all other training and testing activities would remain consistent from year to year.

With the exception of mine warfare events which occur at the three established Underwater Detonation ranges, most scheduled training and testing activities involving explosions would occur well offshore (greater than 12 NM), primarily within special use airspace (e.g., W-517). Activities that involve underwater detonations and explosive munitions typically occur more than 3 NM from shore and in the range complexes, rather than in the transit corridor. The Navy will implement mitigation to avoid potential impacts on hammerhead sharks and giant manta rays in the Mariana Islands Range Complex during explosive mine neutralization activities involving Navy divers, as discussed in Section 5.3.3 (Explosive Stressors). In addition to procedural mitigation, the Navy will implement mitigation to avoid impacts from explosives on seafloor resources in mitigation areas throughout the Study Area (see Section 5.4.1, Mitigation Areas for Seafloor Resources), which will consequently also help avoid potential impacts on fishes that shelter and feed on shallow-water coral reefs, live hard bottom, artificial reefs, and shipwrecks.

Sound and energy from explosions could result in mortality and injury, on average, for hundreds to even thousands of meters from some of the largest explosions. Exposure to explosions could also result in temporary hearing loss in nearby fishes. The estimated range to each of these effects based on explosive bin size is provided in Table 3.9-8 and Table 3.9-9. Generally, explosives that belong to larger bins (with large net explosive weights) produce longer ranges within each effect category. However, some ranges vary depending upon a number of other factors (e.g., number of explosions in a single event, depth of the charge, etc.). Fishes without a swim bladder, adult fishes, and larger species would generally be less susceptible to injury and mortality from sound and energy associated with explosive activities than small, juvenile or larval fishes. Fishes that experience hearing loss could miss opportunities to detect predators or prey, or show a reduction in interspecific communication.

If an individual fish were repeatedly exposed to sound and energy from in-water explosions that caused alterations in natural behavioral patterns or physiological stress, these impacts could lead to long-term consequences for the individual such as reduced survival, growth, or reproductive capacity. If detonations occurred close together (within a few seconds), there could be the potential for masking to occur but this would likely happen at farther distances from the source where individual detonations might sound more continuous. Training and testing activities involving explosions are generally dispersed in space and time. Consequently, repeated exposure of individual fishes to sound and energy from in-water explosions over the course of a day or multiple days is not likely and most behavioral effects are expected to be short-term (seconds or minutes) and localized. Exposure to multiple detonations over the course of a day would most likely lead to an alteration of natural behavior or the avoidance of that specific area.

As discussed previously in Section 3.9.1.1 (Hearing and Vocalization), all ESA-listed fish species that occur in the Study Area are capable of detecting sound produced by explosives. In addition, all ESA-listed species that occur in the Study Area may be exposed to explosives associated with training and testing activities. The Indo-West Pacific Distinct Population Segment of scalloped hammerhead could occur in nearshore waters, such as bays and estuaries, but is also known to occur in offshore portions of the

Study Area. The giant manta ray and oceanic whitetip shark would most likely be exposed to low-frequency sonar in offshore areas throughout the Study Area. Overall, impacts on ESA-listed species that encounter explosions would be similar to those discussed above for impacts on fishes in general.

Pursuant to the ESA, the use of explosives during training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays.

3.9.2.2.2.4 Impacts from Explosives Under Alternative 2 (Preferred Alternative)

As described in Chapter 2 (Description of Proposed Action and Alternatives), Section 3.0.4.2 (Explosive Stressors), and Appendix A (Training and Testing Activities Descriptions), the number of activities that use explosives under Alternative 2 are consistent from year-to-year and would increase slightly compared to activities planned under Alternative 1. The differences in the number of events within the range complex across a year is nominal with only minor changes annually; therefore, the locations, tempo, and general types of predicted impacts would not be discernible from those described above in Section 3.9.2.2.2.3 (Impacts from Explosives Under Alternative 1 – Training Activities).

Pursuant to the ESA, the use of explosives during training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment scalloped hammerhead sharks, oceanic whitetip sharks and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.2.2.5 Impacts from Explosives Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Explosive stressors, as described above, would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer explosive stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for explosive impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.3 Energy Stressors

Energy stressors are discussed in Section 3.0.4.3. Energy stressors that may impact fishes include in-water electromagnetic devices and high-energy lasers. While the number of training and testing events would change under this SEIS/OEIS, the analysis presented in the 2015 MITT Final EIS/OEIS, Section 3.9.3.2 (Energy Stressors) remains valid. The changes in training and testing activities are not substantial and would not result in an appreciable change to existing environmental conditions or an increase in the level or intensity of energy stressors within the Study Area. High-energy lasers were not covered in the 2015 MITT Final EIS/OEIS and represent a new stressor analyzed in this SEIS/OEIS.

As discussed in Section 3.0.4.3.2.2 (High-Energy Lasers), high-energy laser weapons are designed to disable surface targets, rendering them immobile. Fish could be exposed to a laser only if the beam missed the target. Should the laser strike the sea surface, individual fish at or near the surface could be exposed. The potential for exposure to a high-energy laser beam decreases as the water depth increases. Most fish are unlikely to be exposed to laser activities because they primarily occur more than a few meters below the sea surface.

3.9.2.3.1 Impacts from In-Water Electromagnetic Devices Under Alternative 1

Under Alternative 1, the number of proposed training and testing events involving the use of in-water electromagnetic devices would decrease in comparison to the 2015 MITT Final EIS/OEIS (Table 3.0-9). The activities would occur in the same locations and in a similar manner as were analyzed previously.

As stated in the 2015 MITT Final EIS/OEIS, in-water electromagnetic devices would not cause any potential risk to fishes because (1) the range of impact (i.e., greater than earth's magnetic field) is small (i.e., 13 ft. from the source), (2) the electromagnetic components of these activities are limited to simulating the electromagnetic signature of a vessel as it passes through the water, and (3) the electromagnetic signal is temporally variable and would cover only a small spatial range during each activity in the Study Area.

ESA-listed scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays are capable of detecting electromagnetic energy. Therefore, energy stressors such as in-water electromagnetic devices could affect these species by causing temporary disturbances in their normal sensory perception during migratory or foraging movements, or avoidance reactions (Kalmijn, 2000). However, electromagnetic signals are temporally variable and would cover only a small spatial range during each activity in the Study Area. Therefore, impacts on fishes under Alternative 1 from in-water electromagnetic devices would be negligible.

Pursuant to the ESA, the use of in-water electromagnetic devices associated with training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

3.9.2.3.2 Impacts from In-Water Electromagnetic Devices Under Alternative 2 (Preferred Alternative)

Under Alternative 2, the number of proposed training and testing events involving the use of in-water electromagnetic devices would decrease in comparison to the 2015 MITT Final EIS/OEIS (Table 3.0-9). The activities would occur in the same locations and in a similar manner as were analyzed previously and above for Alternative 1.

Under Alternative 2, impacts on fishes from in-water electromagnetic devices should not be expected to occur and would be negligible.

Pursuant to the ESA, the use of in-water electromagnetic devices associated with training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.3.3 Impacts from In-Water Electromagnetic Devices Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Energy stressors as listed above would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer energy stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for energy impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.3.4 Impacts from High-Energy Lasers Under Alternative 1

Under Alternative 1, the number of proposed events involving the use of high-energy lasers would be 54 (Table 3.0-10); this is a new substressor that was not analyzed in the 2015 MITT Final EIS/OEIS. As discussed above, the potential for fishes to be exposed to high-energy lasers is extremely low, and impacts from high-energy laser activities proposed under Alternative 1 should not be expected to occur. Therefore, impacts on fishes under Alternative 1 from high-energy lasers, would be negligible.

Pursuant to the ESA, the use of high-energy lasers during training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

3.9.2.3.5 Impacts from High-Energy Lasers Under Alternative 2 (Preferred Alternative)

Under Alternative 2, the number of proposed events involving the use of high-energy lasers would increase from 54 to 60 compared to Alternative 1 (Table 3.0-10) and the 2015 MITT Final EIS/OEIS; however, as discussed above, impacts on fishes from high-energy lasers should not be expected to occur. Therefore, impacts on fishes under Alternative 2 from energy stressors, including high-energy lasers, would be negligible.

Pursuant to the ESA, the use of high-energy lasers during training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.3.6 Impacts from High-Energy Lasers Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Energy stressors, as listed above, would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer energy stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for energy impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.4 Physical Disturbance and Strike Stressors

Physical disturbance and strike stressors are discussed in Section 3.0.4.4. Physical disturbance and strike stressors that may impact fishes include (1) vessels and in-water devices, (2) military expended materials, and (3) seafloor devices. While the number of training and testing events would change under this SEIS/OEIS, the analysis presented in the 2015 MITT Final EIS/OEIS, Section 3.9.3.3 (Physical Disturbance and Strike) remains valid. The changes in training and testing activities are not substantial and would not result in an overall change to existing environmental conditions or an increase in the level or intensity of physical disturbance and strike stressors within the Study Area.

As stated in the 2015 MITT Final EIS/OEIS, with few exceptions, activities involving vessels and in-water devices are not intended to contact the seafloor. There is minimal potential strike impact other than bottom-crawling unmanned underwater vehicles. Physical disturbance and strike stressors from vessels and in-water devices, military expended materials, and seafloor devices have the potential to affect all marine fish groups found within the Study Area, although some fish groups may be more susceptible to strike potential than others. In addition, the potential responses to physical strikes are varied, but include behavioral changes such as avoidance, altered swimming speed and direction, physiological stress, and physical injury or mortality.

3.9.2.4.1 Impacts from Physical Disturbance and Strike Stressors Under Alternative 1

Under Alternative 1, the combined number of proposed training and testing events involving vessels and in-water devices would decrease slightly from those presented in the 2015 MITT Final EIS/OEIS (Table 3.0-12 and Table 3.0-13). Military expended materials (Table 3.0-14, Table 3.0-15, and Table 3.0-16) combined would generally increase, and seafloor devices (Table 3.0-19) would decrease slightly from the number in the 2015 MITT Final EIS/OEIS. Increases in physical disturbance and strike stressors, such as military expended materials, could increase the level of impact on some fishes. Analysis by individual category of expended items indicates that those items having the most potential to affect fishes have decreased. Overall, these changes do not appreciably change the analysis or impact conclusions presented in the 2015 MITT Final EIS/OEIS because the impact analysis was based on the probability of an impact on a resource.

The risk of a strike from vessels and in-water devices used in training and testing activities on an individual fish would be extremely low because (1) most fish can detect and avoid vessel and in-water device movements, and (2) the types of fish that are likely to be exposed to vessel and in-water device strike are limited and occur in low concentrations where vessels and in-water devices are used. Potential impacts of exposure to vessels and in-water devices are not expected to result in substantial changes to an individual’s behavior, fitness, or species recruitment, and are not expected to result in population-level impacts. Therefore, impacts on fish or fish populations would be negligible.

Similar to most other fish species described above, ESA-listed scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays, would be able to sense pressure changes in the water column and swim quickly, and are likely to escape collision with vessels and in-water devices.

Therefore, under Alternative 1, impacts on fishes from the use of vessels and in-water devices, military expended materials, and seafloor devices would be negligible.

Pursuant to the ESA, the use of vessels and in-water devices associated with training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

3.9.2.4.2 Impacts from Physical Disturbance and Strike Stressors Under Alternative 2 (Preferred Alternative)

Under Alternative 2, the combined number of proposed training and testing events involving vessels and in-water devices would decrease slightly from those presented in the 2015 MITT Final EIS/OEIS (Table 3.0-12 and Table 3.0-13). Military expended materials (Table 3.0-14, Table 3.0-15, and Table 3.0-16) combined would generally increase, and seafloor devices (Table 3.0-19) would decrease slightly from the number in the 2015 MITT Final EIS/OEIS. Increases in some physical disturbance and strike stressors such as military expended materials could increase the impact risk on fishes but does not appreciably change the analysis or impact conclusions presented in the 2015 MITT Final EIS/OEIS. Impacts on fishes would be inconsequential for the same reasons detailed above and would have no appreciable change on the impact conclusions for physical disturbance and strike stressors, as presented in the 2015 MITT Final EIS/OEIS and summarized above under Alternative 1.

Therefore, under Alternative 2, impacts on fishes from physical disturbance and strike would be negligible.

Pursuant to the ESA, the use of vessels and in-water devices associated with training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.4.3 Impacts from Physical Disturbance and Strike Stressors Under the No Action Alternative

Under the No Action Alternative, proposed training and testing. Other military activities not associated with this Proposed Action would continue to occur. Physical disturbance and strike stressors as listed above would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer physical disturbance and strike stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for physical disturbance and strike impacts on individual fishes, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.5 Entanglement Stressors

Entanglement stressors are discussed in Section 3.0.4.5. Entanglement stressors considered for fishes include (1) fiber optic cable and guidance wires, and (2) decelerators/parachutes. The annual number of wires and cables and decelerators/parachutes proposed under the alternatives and in comparison to current ongoing activities are presented in Tables 3.0-20, 3.0-21, and 3.0-22. There have been no known instances of any fish being entangled in wires and cables, or decelerators/parachutes associated with Navy training and testing activities prior to or since the 2015 MITT Final EIS/OEIS.

3.9.2.5.1 Impacts from Entanglement Stressors Under Alternative 1

Under Alternative 1, the combined number of fiber optic cables (Table 3.0-22) decrease, guidance wires (Table 3.0-22) increase, and decelerators/parachutes (Table 3.0-24) decrease compared to the number of events proposed in the 2015 MITT Final EIS/OEIS. Decreases in the number of training and testing events would potentially decrease the level of entanglement stressors on fishes in the Study Area.

As stated in the 2015 MITT Final EIS/OEIS, while individual fish susceptible to entanglement would encounter wires and cables, including guidance wires, fiber optic cables, and sonobuoy wires during training and testing activities, the long-term consequences of entanglement are unlikely for either individuals or populations because (1) the encounter rate for wires and cables is low, (2) the types of fishes that are susceptible to these items is limited, (3) there is restricted overlap with susceptible fishes, and (4) the physical characteristics of the wires and cables reduce entanglement risk to fishes compared to monofilament used for fishing gear. Potential impacts from exposure to fiber optic cables and guidance wires are not expected to result in substantial changes to an individual's behavior, fitness, or species recruitment, and are not expected to result in population-level impacts.

As described in the 2015 MITT Final EIS/OEIS, it would be very unlikely that fishes would encounter and become entangled in any decelerators/parachutes or sonobuoy accessories. This is mainly due to the size of the range complexes and the resulting widely scattered decelerators/parachutes. If a few individual fish were to encounter and become entangled in a decelerator/parachute, the growth, survival, annual reproductive success, or lifetime reproductive success of the population as a whole would not be impacted directly or indirectly.

Therefore, impacts on fishes under Alternative 1 from the use of fiber optic cables and guidance wires and decelerators/parachutes would be negligible.

Pursuant to the ESA, the use of fiber optic cables and guidance wires and decelerators/parachutes associated with training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

3.9.2.5.2 Impacts from Entanglement Stressors Under Alternative 2 (Preferred Alternative)

Under Alternative 2, the combined number of entanglement stressors decrease (Table 3.0-22 through Table 3.0-24) compared to the number of events proposed in the 2015 MITT Final EIS/OEIS and would increase or stay the same compared to Alternative 1. However, as stated above for Alternative 1, training and testing activities involving fiber optic cables, guidance wires, and decelerators/parachutes are not expected to impact an individual's behavior, fitness, or species recruitment, and are not expected to result in population-level impacts.

Therefore, impacts on fishes from entanglement stressors such as wires and cables and decelerators/parachutes under Alternative 2 would be negligible.

Pursuant to the ESA, the use of fiber optic cables and guidance wires and decelerators/parachutes associated with training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.5.3 Impacts from Entanglement Stressors Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Entanglement stressors as listed above would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer physical disturbance and strike stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for entanglement impacts on the fishes from entanglement, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally managed under the MSA.

3.9.2.6 Ingestion Stressors

Ingestion stressors (military expended materials – munition and military expended materials – other than munition) are discussed in Section 3.0.4.6. Ingestion stressors that may impact fishes include various types of military expended materials such as munitions and expended materials other than munitions used by the Navy during training and testing activities within the Study Area. While the number of training and testing events would change under this SEIS/OEIS, the analysis presented in the 2015 MITT Final EIS/OEIS, Section 3.9.3.2 (Ingestion Stressors) remains valid. The changes in training and testing activities are not substantial and would not result in an appreciable change to existing environmental conditions or an increase in the amount of ingestion stressors within the Study Area.

3.9.2.6.1 Impacts from Ingestion Stressors Under Alternative 1

Under Alternative 1, the combined number of ingestion stressors would increase compared to the number in the 2015 MITT Final EIS/OEIS (see Table 3.0-14, Table 3.0-15, Table 3.0-16, Table 3.0-25, and Table 3.0-26). However, increases in the number of ingestion stressors do not appreciably change the impact analysis or conclusions presented in the 2015 MITT Final EIS/OEIS.

As presented in the 2015 MITT Final EIS/OEIS, open-ocean predators and open-ocean planktivores are most likely to ingest materials in the water column, while coastal bottom-dwelling predators and estuarine bottom-dwelling predators could ingest materials from the seafloor. Open-ocean predators such as tunas and sharks may eat floating or sinking expended materials, while open-ocean planktivores, such as sardines and filter-feeding species such as whale sharks, may ingest floating expended materials incidentally as they feed in the water column. Other fish species such as skates and rays forage on the seafloor and may ingest expended materials on the seafloor. Encounter rates for all of these feeding guilds would be extremely low, but may result in injury or death to individuals; however, population-level effects are not anticipated.

Potential impacts of ingestion on some adult fishes are different than for other life stages (eggs, larvae, and juveniles) because early life stages for some species are too small to ingest any military expended materials except for chaff, which has been shown to have limited effects on fishes in the concentration levels that it is released at (Arfsten et al., 2002; U.S. Department of the Air Force, 1997; U.S. Department of the Navy, 1999). Therefore, with the exception of later stage larvae and juveniles that could ingest microplastics, no ingestion potential impacts on early life stages are expected.

Overall, the potential impacts of ingesting expended military materials such as munitions or other expended materials, such as chaff and flare end caps and pistons, would be limited to individual fish that might suffer a negative response from a given ingestion event. While ingestion of military expended materials could result in sublethal or lethal effects to a small number of individuals, the likelihood of a fish encountering an expended item is dependent on where that species feeds and the amount of material expended. Furthermore, an encounter may not lead to ingestion, as a fish might "taste" an item, then expel it (Felix et al., 1995), in the same manner that a fish would take a lure into its mouth then spit it out.

Therefore, the number of fishes potentially impacted by ingestion of military expended materials such as munitions and other expended materials would be negligible.

Pursuant to the ESA, the use of military expended materials associated with training and testing activities, as described under Alternative 1, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays.

3.9.2.6.2 Impacts from Ingestion Stressors Under Alternative 2 (Preferred Alternative)

Under Alternative 2, the combined number of ingestion stressors would increase compared to the number proposed for use in the 2015 MITT Final EIS/OEIS and above for Alternative 1 (see Table 3.0-14, Table 3.0-15, Table 3.0-16, Table 3.0-25, and Table 3.0-26). However, these increases do not appreciably change the impact analysis or conclusions presented in the 2015 MITT Final EIS/OEIS and presented above under Alternative 1.

Therefore, impacts on fishes from ingestion of military expended materials under Alternative 2 would be negligible.

Pursuant to the ESA, the use of military expended materials associated with training and testing activities, as described under Alternative 2, may affect ESA-listed Indo-West Pacific Distinct Population Segment of scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. The Navy is consulting with NMFS as required by Section 7(a)(2) of the ESA.

3.9.2.6.3 Impacts from Ingestion Stressors Under the No Action Alternative

Under the No Action Alternative, proposed training and testing activities would not occur. Other military activities not associated with this Proposed Action would continue to occur. Ingestion stressors as listed above would not be introduced into the marine environment. Therefore, existing environmental conditions would either remain unchanged or would improve slightly after cessation of ongoing training and testing activities.

Discontinuing the training and testing activities would result in fewer ingestion stressors within the marine environment where training and testing activities have historically been conducted. Therefore, discontinuing training and testing activities under the No Action Alternative would lessen the potential for ingestion impacts on the fishes from ingestion of military expended material, but would not measurably improve the status of fish populations or subpopulations, including those listed under ESA and those federally-managed under the MSA.

3.9.2.7 Secondary Stressors

Secondary stressors from training and testing activities that could pose secondary or indirect impacts on fishes via habitat, prey, sediment, and water quality include (1) explosives and byproducts; (2) metals; (3) chemicals; (4) other materials such as targets, chaff, and plastics; and (5) impacts on fish habitat. While the number of training and testing events would change under this SEIS/OEIS, the analysis presented in the 2015 MITT Final EIS/OEIS, Section 3.9.3.6 (Secondary Stressors) remains valid. The changes in training and testing activities are not substantial and would not result in an appreciable change to existing environmental conditions or an increase in the level or intensity of energy stressors within the Study Area.

As stated in the 2015 MITT Final EIS/OEIS, indirect impacts of explosives and unexploded ordnance on fishes via water could not only cause physical impacts, but prey might also have behavioral reactions to underwater sound. For example, the sound from underwater explosions might induce startle reactions and temporary dispersal of schooling fishes if they are within close proximity. The abundances of fish and invertebrate prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Secondary impacts from underwater explosions would be temporary, and no lasting impact on prey availability or the pelagic food web would be expected. Indirect impacts of underwater detonations and explosive ordnance use under the

Proposed Action would not result in a decrease in the quantity or quality of fish populations or fish habitats in the Study Area.

Indirect impacts of explosives and unexploded ordnance to fishes via sediment is possible in the immediate vicinity of the ordnance. Degradation of explosives proceeds via several pathways discussed in Section 3.1 (Sediments and Water Quality). Degradation products of Royal Demolition Explosive are not toxic to marine organisms at realistic exposure levels (Rosen & Lotufo, 2010). TNT and its degradation products impact developmental processes in fishes and are acutely toxic to adults at concentrations similar to real-world exposures (Halpern et al., 2008; Rosen & Lotufo, 2010). It is likely that various lifestages of fishes could be impacted by the indirect impacts of degrading explosives within a very small radius of the explosive (1–6 ft.), but these impacts are expected to be short term and localized.

Certain metals are harmful to fishes at concentrations above background levels (e.g., cadmium, chromium, lead, mercury, zinc, copper, manganese, and many others) (Wang & Rainbow, 2008). Metals are introduced into seawater and sediments as a result of Navy training and testing activities involving vessel hulks, targets, ordnance, munitions, and other military expended materials. Indirect impacts of metals to fishes via sediment and water involve concentrations that are several orders of magnitude lower than concentrations achieved via bioaccumulation. Fishes may be exposed by contact with the metal, contact with contaminants in the sediment or water, and ingestion of contaminated sediments. Concentrations of metals in sea water are orders of magnitude lower than concentrations in marine sediments. It is extremely unlikely that fishes would be indirectly impacted by toxic metals via the water.

Several training and testing activities introduce potentially harmful chemicals into the marine environment; principally, flares and propellants for rockets, missiles, and torpedoes. The greatest risk to fishes from flares, missile, and rocket propellants is perchlorate, which is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals. Fishes may be exposed by contact with contaminated water or ingestion of contaminated sediments. Since perchlorate is highly soluble, it does not readily absorb to sediments. Therefore, missile and rocket fuel pose no risk of indirect impact on fishes via sediment. In contrast, the principal toxic components of torpedo fuel, propylene glycol dinitrate and nitrodiphenylamine, adsorbs to sediments, has relatively low toxicity, and is readily degraded by biological processes. It is conceivable that various lifestages of fishes could be indirectly impacted by propellants via sediment in the immediate vicinity of the object (e.g., within a few inches), but these potential impacts would diminish rapidly as the propellant degrades.

As described in the 2015 MITT Final EIS/OEIS, some military expended materials (e.g., decelerators/parachutes) could become remobilized after their initial contact with the sea floor (e.g., by waves or currents) and could be reintroduced as an entanglement or ingestion hazard for fishes. In some bottom types (without strong currents, hard-packed sediments, and low biological productivity), items such as projectiles might remain intact for some time before becoming degraded or broken down by natural processes. While these items remain intact sitting on the bottom, they could potentially remain ingestion hazards. These potential impacts may cease only (1) when the military expended materials is too massive to be mobilized by typical oceanographic processes, (2) if the military expended materials become encrusted by natural processes and incorporated into the seafloor, or (3) when the military expended materials become permanently buried. In this scenario, a parachute could initially sink to the seafloor, but then be transported laterally through the water column or along the seafloor, increasing the opportunity for entanglement. In the unlikely event that a fish would become entangled, injury or mortality could result. The entanglement stressor would eventually cease to pose an entanglement risk as it becomes encrusted or buried, or degrades.

Secondary stressors can also involve impacts on habitat (sediment or water quality) or prey (i.e., impacting the availability or quality of prey) that have the potential to affect fish species, including ESA-listed scalloped hammerhead sharks, oceanic whitetip sharks, and manta rays. Secondary stressors that may affect ESA-listed species only include those related to the use of explosives. Secondary effects on prey and habitat from the release of metals, chemicals, and other materials into the marine environment during training and testing activities are not anticipated. In addition to directly impacting ESA-listed species, underwater explosives could impact other species in the food web, including those that these species prey upon. The impacts of explosions would differ depending upon the type of prey species in the area of the blast. In addition to physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that might include swimming to the surface or scattering away from the source. This startle and flight response is the most common secondary defense among animals. The abundances of prey species near the detonation point could be diminished for a short period of time, affecting prey availability for ESA-listed species feeding in the vicinity. Any effects to prey, other than prey located within the impact zone when the explosive detonates, would be temporary. The likelihood of direct impacts on fishes and mobile invertebrates is low, as described in this section. No lasting effects on prey availability or the pelagic food web would be expected.

3.9.3 Public Comments

The public raised a number of issues during the scoping period in regards to fishes. The issues are summarized in the list below. Comments received from the public during the Draft SEIS/OEIS commenting period related to fishes are addressed in Appendix K (Public Comment Responses).

- **Acoustic and explosive disturbance to fish and EFH –** As described in the 2015 MITT Final EIS/OEIS, and documented in Section 3.9.2.1 (Acoustic Stressors), Navy training and testing activities may affect individual fish by causing some minor behavioral reactions. However, these activities would not cause a population-level impact. For federally managed fish species and habitats under the MSA, those impacts are detailed in Chapter 6. The Navy would also use mitigation measures detailed in Chapter 5 (Mitigation) to reduce potential impacts on less than significant levels. For example, during Explosive Mine Neutralization Activities involving Navy divers, divers will notify their supporting small boat or Range Safety Officer of hammerhead shark sightings (of any hammerhead species, due to the difficulty of differentiating species) at the detonation location. The Navy will delay fuse initiations or detonations until the shark is observed exiting the detonation location.
- **Direct and cumulative impacts from military-expended material and debris on marine biology –** As described in the 2015 MITT Final EIS/OEIS and above, military expended material may affect marine biological resources such as fishes through physical disturbance and strike, entanglement, ingestion, and have a cumulative effect on these resources. However, due to the low potential for interaction between biological resources and entanglement, ingestion, and strike stressors for reasons discussed above and in the 2015 MITT Final EIS/OEIS, military expended materials are not expected to pose a significant risk to the marine resources, including fishes.

- **Direct and cumulative impacts on fish populations –** As described in the 2015 MITT Final EIS/OEIS and in most sections above, impacts on fish from acoustic and explosive stressors (Section 3.9.2.1, Acoustic Stressors, and Section 3.9.2.2, Explosive Stressors) may injure or kill a few individuals but are unlikely to have measurable impacts on overall stocks or populations, including ESA-listed scalloped hammerhead sharks, oceanic whitetip sharks, and giant manta rays. As stated in the 2015 MITT Final EIS/OEIS, if an underwater explosion occurred in an area of high fish density, then more fish would be impacted; however, the probability of this occurring is low based on the patchy distribution of dense schooling fish. In addition, near shore areas used for underwater seafloor detonations are areas that have been previously disturbed and unlikely to support large schools or groups of fish. Cumulative impacts may affect individual fish, but would not have population-level impacts.
- **Impacts on marine species from the metals in the water (copper and lead) (see Section 3.9.2.7, Secondary Stressors) –** As described in the 2015 MITT Final EIS/OEIS and above, metals would be introduced into seawater and sediments as a result of Navy training and testing activities involving vessel hulks, targets, ordnance, munitions, and other military expended materials. Fishes may be exposed by contact with the metal, contact with contaminants in the sediment or water, and ingestion of contaminated sediments. Concentrations of metals in sea water are orders of magnitude lower than concentrations in marine sediments. It is extremely unlikely that fishes would be indirectly impacted by toxic metals via the water.

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