3.6 Fishes

Gulf of Alaska Navy Training Activities

Draft Supplemental Environmental Impact Statement/ Overseas Environmental Impact Statement

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3.6.1 Introduction

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

The purpose of the fishes section in this SEIS/OEIS is to provide any new or changed information since the 2016 GOA Final SEIS/OEIS that is relevant to an analysis of potential impacts on fish and their habitat associated with the continuation of Navy training activities in the Temporary Maritime Activities Area (TMAA) beyond May 2022. The TMAA is 12 nautical miles (NM) or greater from shore, outside of the U.S. Territorial Sea. The current National Marine Fisheries Service (2017) Biological Opinion for Navy training in the TMAA is effective from April 26, 2017, through April 26, 2022, at which time NMFS plans to update the environmental baseline and reassess any changes in fish species status.

The organizational structure of the fish affected environment section varies slightly from that presented in the 2016 GOA Final SEIS/OEIS. Background information in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS for the fish species that occur in the TMAA will not be repeated in this section unless necessary for context in support of new information and emergent relevant best available science. This supplement includes fish status updates, recent GOA fish research studies, new groundfish harvest data, and amendments to Fishery Management Plans (FMPs) and related Essential Fish Habitat (EFH) designations since the 2016 GOA Final SEIS/OEIS. This information is presented in the subsections that follow.

3.6.2 Affected Environment

The predominant fish species and habitat types known to occur in the TMAA have not changed since they were described in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. The TMAA supports two primary categories of fishes: anadromous salmonids (genus *Oncorhynchus*; hereafter referred to as salmonids) and groundfishes. Salmonids found within the Gulf of Alaska include Chinook salmon (*O. tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), pink salmon (*O. gorbuscha*), sockeye salmon (*O. nerka*), and steelhead (*O. mykiss*). The life histories of the dominant species of salmonids and groundfishes occurring in the Gulf of Alaska are described in the 2011 GOA Final EIS/OEIS, with some updated information on salmonid distribution and migration patterns provided herein.

In the subsequent sections, updated information has been incorporated on the distribution and management status of these fishes within the Gulf of Alaska and TMAA. Further, a discussion of the Endangered Species Act (ESA)-listed Southern Distinct Population Segment (DPS) of green sturgeon (*Acipenser medirostris*) has been included based on additional information suggesting that it may occur within the continental shelf portion of the TMAA. This SEIS/OEIS describes a few species status changes and fisheries management updates that have occurred since the 2016 GOA Final SEIS/OEIS. With the exception of these changes, the information and analysis presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid.

The TMAA overlaps a portion of the continental shelf/slope but is mostly located within offshore pelagic (open ocean) habitats that include the abyssal plain and various seamounts. These habitats are influenced by the Alaska Coastal Current and the Alaska Gyre. With the exception of Montague Island located over 12 NM from the northern point of the TMAA, the nearest shoreline (Kenai Peninsula) is located approximately 24 NM north of the TMAA's northern boundary (Figure 3.6-1). The GOA shelf is dominated by gravel, sand, silt, and mud, punctuated by areas of hard rock (Fautin et al., 2010). There are numerous banks and reefs with coarse, rocky bottoms, but much of the shelf is covered by glacial silt from the Copper River and the Bering and Malaspina glaciers (Mundy, 2005). Habitat types and their characteristics within the TMAA were described in the 2011 GOA Final EIS/OEIS.

3.6.2.1 General Background

3.6.2.1.1 Endangered Species Act-Listed Species in the Temporary Maritime Activities Area

Many ESA-listed fish species (including various salmonids and green sturgeon) from the U.S. West Coast may occur within the TMAA. Following a review of Federal Register (FR) publications (National Marine Fisheries Service, 2020b) since the 2016 GOA Final SEIS/OEIS, the most current federal status of threatened, endangered, and candidate fish species is presented in Table 3.6-1. Abundance data and trends for all Pacific salmonid Evolutionarily Significant Units (ESUs)/DPSs are incorporated by reference in National Marine Fisheries Service (2016b). Candidate species are any species that are undergoing a status review that NMFS has announced through an FR notice (71 FR 61022). Candidate species do not carry any procedural or substantive protections under the ESA (71 FR 61022). Table 3.6-1 indicates ESA-listed salmonid species that originate from rivers in Washington, Oregon, and California that have been confirmed to be, or may be, present in the TMAA during certain periods of their life cycle. Salmon and steelhead that originate from Alaskan rivers may be present in the TMAA, but since they are not listed under the ESA, they are not included in the table.

In addition, green sturgeon have occasionally been documented in Alaskan waters as far north as Unalaska Island, and two fish from the ESA-listed southern DPS have been identified at Graves Harbor in Southeast Alaska (Environmental Protection Information Center et al., 2001) (74 FR 52300). Although a few green sturgeon have been documented in the Gulf of Alaska, they were not identified to a DPS so it is unclear whether they were part of the ESA-listed Southern DPS. However, based on their migration patterns, it is possible that ESA-listed green sturgeon could be present within the on-shelf portion of the TMAA. However, as described in Section 3.6.2.8.2 (Distribution), they are not expected to be found within the offshore portion.

On October 4, 2019, NMFS announced that they plan to initiate five-year reviews of 28 Pacific salmonid species listed under the ESA (84 FR 53117). The purpose of these reviews is to ensure the accuracy of their listing classifications. The five-year reviews will be based on the best scientific and commercial data available at the time of the reviews; NMFS accepted comments until March 27, 2020. Based on the results of these five-year reviews, NMFS will make the requisite determinations under the ESA.

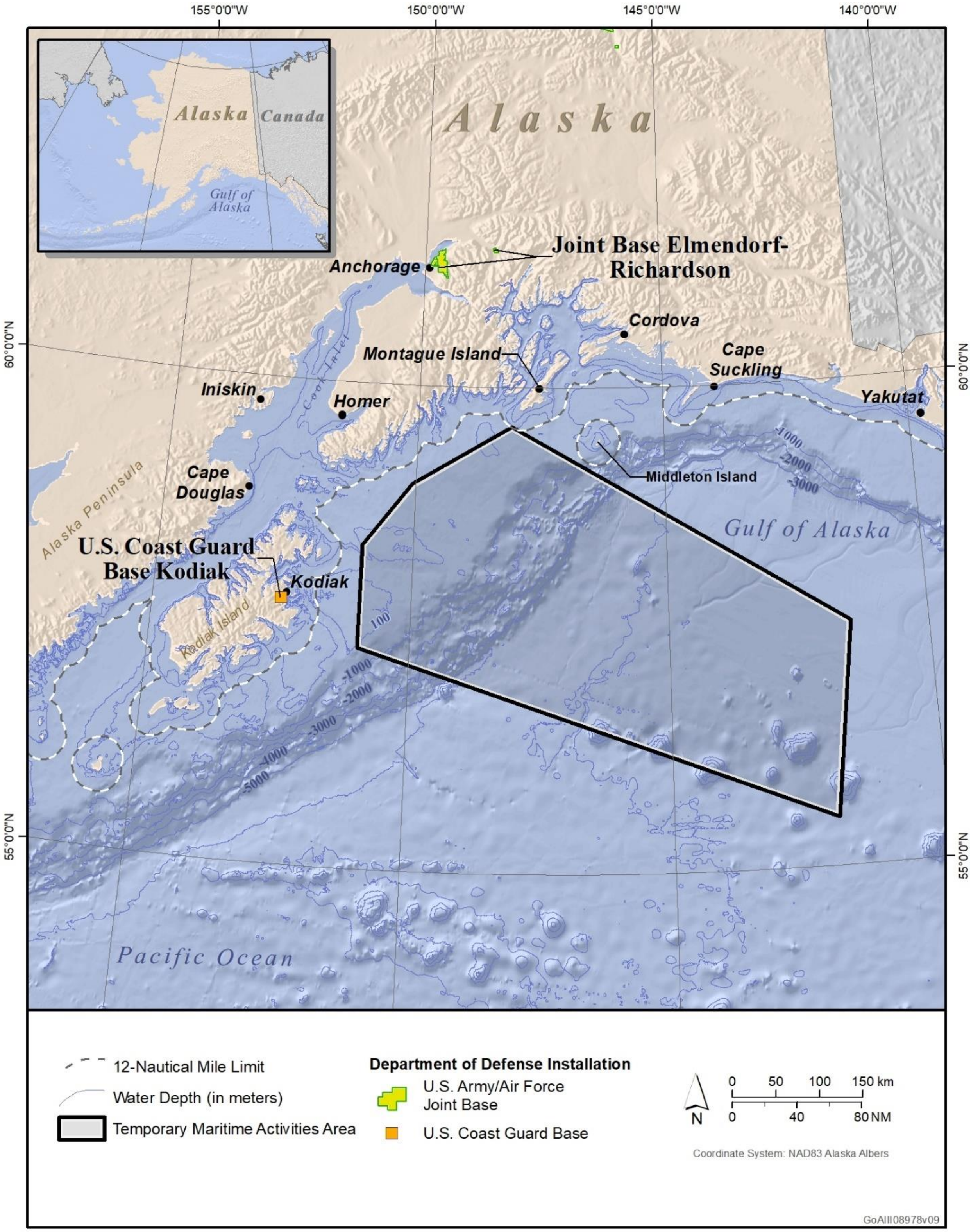

Figure 3.6-1: Gulf of Alaska Navy Training Activities Study Area

Table 3.6-1: Status and Presence of ESA-Listed Fish Species and their Designated Critical Habitat and Candidate Species Found in the TMAA within the Gulf of Alaska

Species and Regulatory Status				*Presence in the GOA and TMAA*	
Common Name (Scientific Name)	***Distinct Population Segment (DPS)/ Evolutionarily Significant Unit (ESU)***	***Federal Status***	***Critical Habitat Designation***	***Documented Presence in the GOA/TMAA[1]***	***Likelihood of Presence in the TMAA***
Chinook Salmon (*Oncorhynchus tshawytscha*)	Puget Sound ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Upper Columbia River Spring-run ESU	E	Designated (Not in TMAA)	X/X	Confirmed
	Lower Columbia River ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Snake River Spring/Summer-run ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Snake River Fall-run ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Upper Willamette River ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Oregon Coast Spring-Run ESU[2]	C	Not Designated	-/-	Potential
	Upper Klamath-Trinity River ESU[2]	C	Not Designated	-/-	Potential
	California Coastal ESU	T	Designated (Not in TMAA)	-/-	Potential
	Sacramento River Winter-run ESU	E	Designated (Not in TMAA)	-/-	Potential
	Central Valley Spring-run ESU	T	Designated (Not in TMAA)	X/-	Potential
Coho Salmon (*Oncorhynchus kisutch*)	Lower Columbia River ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Oregon Coast ESU	T	Designated (Not in TMAA)	X/X	Confirmed
	Southern Oregon/Northern California Coasts ESU	T	Designated (Not in TMAA)	-/-	Potential
	Central California Coast ESU	E	Designated (Not in TMAA)	-/-	Potential
Chum Salmon (*Oncorhynchus keta*)	Hood Canal Summer-run ESU	T	Designated (Not in TMAA)	-/-	Likely
	Columbia River ESU	T	Designated (Not in TMAA)	-/-	Likely

Table 3.6-1: Status and Presence of ESA-Listed Fish Species and their Designated Critical Habitat and Candidate Species Found in the TMAA within the Gulf of Alaska (continued)

Species and Regulatory Status				***Presence in the GOA and TMAA***	
Common Name (Scientific Name)	***Distinct Population Segment (DPS)/ Evolutionarily Significant Unit (ESU)***	***Federal Status***	***Critical Habitat Designation***	***Documented Presence in the GOA/TMAA***[1]	***Likelihood of Presence in the TMAA***
Sockeye Salmon (*Oncorhynchus nerka*)	Snake River ESU	E	Designated (Not in TMAA)	-/-	Likely
	Ozette Lake ESU	T	Designated (Not in TMAA)	-/-	Likely
Steelhead (*Oncorhynchus mykiss*) ~~(continued)~~	Puget Sound DPS	T	Designated (Not in TMAA)	X/-	Likely
	Upper Columbia River DPS	T	Designated (Not in TMAA)	X/-	Likely
	Middle Columbia River DPS	T	Designated (Not in TMAA)	X/-	Likely
	Lower Columbia River DPS	T	Designated (Not in TMAA)	X/-	Likely
	Snake River Basin DPS	T	Designated (Not in TMAA)	X/-	Likely
	Upper Willamette River DPS	T	Designated (Not in TMAA)	X/-	Likely
	Northern California DPS	T	Designated (Not in TMAA)	-/-	Potential
	Northern California Summer-Run DPS[2]	C	Not Designated	-/-	Potential
	California Central Valley DPS	T	Designated (Not in TMAA)	-/-	Potential
	Central California Coast DPS	T	Designated (Not in TMAA)	X/-	Potential
	South-Central California Coast DPS	T	Designated (Not in TMAA)	-/-	Potential
	Southern California DPS	E	Designated (Not in TMAA)	-/-	Unlikely
Green Sturgeon (*Acipenser medirostris*)	Southern DPS	T	Designated (Not in TMAA)	-/-	Potential

[1] Presence based on coded wire tag studies (see Section 3.6.2.7, Site-Specific Information on Endangered Species Act-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area)
[2] New/updated species status since the 2016 GOA Final SEIS/OEIS.
Notes: Federal Status: C = Candidate, E = Endangered, T = Threatened; "X" = Documented; "-" = Not Documented
Sources: (National Marine Fisheries Service, 2016a, 2020b)

3.6.2.1.2 Endangered Species Act-Listed Species Unlikely to be Present in the Temporary Maritime Activities Area

The Southern California Steelhead DPS is the only ESA-listed fish species addressed in this document that is considered unlikely to be present in the TMAA. In their southern range, steelhead tend to migrate north and south along the continental shelf, a pattern that may be related to the shorter time these stocks spend in saltwater (Barnhart, 1991; Busby et al., 1996; Moyle et al., 2017). There is no evidence suggesting that these fish migrate as far north as the Gulf of Alaska. Many steelhead stocks in the northern range are known to make extensive offshore migrations. For example, Oregon, Washington, and British Columbia steelhead are commonly captured in Alaskan waters (Barnhart, 1991). Although California stocks were not previously known to occur in the Gulf of Alaska, coded wire tag (CWT) data indicates that some individuals from the Central California Coast Steelhead DPS and Central Valley spring-run Chinook Salmon ESU have been found in the Gulf of Alaska and may occur in the TMAA (Hayes et al., 2011). Thus, there is potential for ESA-listed fish from Washington south to Central California to occur in the TMAA.

3.6.2.1.3 Hearing and Vocalization

A summary of fish hearing and vocalizations is described in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Due to the availability of new literature, including revised sound exposure criteria, the information provided below will supplant the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/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 & Schilt, 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, 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 levels 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 B, 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 of 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 higher frequency hearing capabilities and better sound pressure detection.

Although many researchers have investigated hearing and vocalizations in fish species, hearing capability data only exist for just over 100 of the currently known 34,000 marine and freshwater fish species (Eschmeyer & Fong, 2016). 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; Popper & Hastings, 2009b). 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, 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, possess anatomical specializations, and are capable of sound pressure detection at frequencies up to 10 kHz to over 100 kHz (not present in Study Area).

The quantitative literature review conducted by Wiernicki et al. (2020), the x-ray and image processing performed by Schulz-Mirbach et al. (2020), and hearing measurements and dissections of black sea bass by Stanley et al. (2020) continue to support the above hearing group classifications. Additional research is still needed to better understand species-specific frequency detection capabilities and overall sensitivity to sound.

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 or shads) 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. (1998; 1997) 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).

ESA-listed species with the potential to occur within the TMAA include various salmonid species, as well as green sturgeon (see Table 3.6-1 for details). There are no available data on the hearing capabilities of these specific ESA-listed populations. Instead, each species is considered to be part of a hearing group described above based on data from similar species, and knowledge of that species' physiology. As discussed above, most marine fishes investigated to date lack hearing capabilities greater than 1,000 Hz. Notably, this includes sturgeon and salmonid species, fishes with a swim bladder that is not involved in hearing. Although it is assumed that sturgeon and salmon species can detect frequencies up to 1,000 Hz, available hearing data has only tested these species up to about 600 Hz (Hawkins & Johnstone, 1978; Kane et al., 2010; Lovell et al., 2005; Meyer et al., 2010). There are no ESA-listed species that occur in the TMAA that have a swim bladder that is involved in hearing, or that have high-frequency hearing (the two most sensitive hearing groups).

Many fishes are known to produce sound. Bony fishes can produce sounds in a number of ways and use them for a variety 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 referenced to 1 micropascal (dB re 1 μPa). Female midshipman fish detect and locate the "hums" (approximately 90 to 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 decibels (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 monitoring by divers) 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., 2020; Rowell et al., 2018).

3.6.2.1.4 General Threats

General threats to fish species within the TMAA were not addressed in the 2011 GOA Final EIS/OEIS or the 2016 GOA Final SEIS/OEIS. The major threats to fish species that were described in the 2015 Biological Evaluation and 2017 GOA Biological Opinion are summarized and updated below. Much of the Climate Change discussion below was summarized from Johnson (2016).

Climate Change

The Intergovernmental Panel on Climate Change's Fifth Assessment Synthesis Reports conclude that climate change is unequivocal (Intergovernmental Panel on Climate Change, 2013, 2014). The reports indicate that oceans have warmed, with the greatest warming occurring near the surface. Over the last 60 years Alaska has warmed more than twice as rapidly as most of the United States. The U.S. Environmental Protection Agency reports that average annual temperatures in Alaska have risen 3.4 degrees Fahrenheit (°F) (winter temperatures have risen 6.2°F) during that period, and some projections call for another 2–4°F increase by the middle of this century (Chapin III et al., 2014; Johnson, 2016). It is expected that long-term warming trends will override inter-annual or multi-decadal climate variability (Johnson, 2016).

Potential consequences of climate change on fish in the Gulf of Alaska include temperature and salinity stratification; changes to primary productivity and prey base; ocean acidification; decreased ocean oxygen levels; invasive species; and harmful algal blooms (Johnson, 2016). Climate change has the potential to impact species abundance, geographic distribution (both laterally and vertically), migration patterns, timing of seasonal activities (Intergovernmental Panel on Climate Change, 2014), and species viability into the future.

Climate change may affect food web processes in the Gulf of Alaska through changes in oceanic stratification. Phytoplankton form the basis of the oceanic food web and require sunlight energy and nutrient mixing to support a phytoplankton bloom. If summer temperatures are too warm thermal stratification occurs, which blocks deeper nutrients from reaching phytoplankton near the surface. Timing and intensity of phytoplankton blooms must match the abundance of zooplankton, and the eggs and larvae of fish and crustaceans, for maximum fisheries productivity (Johnson, 2016). Further, a warming climate may cause winter precipitation to shift from a snow to rain-dominated system on the Gulf of Alaska coast. As such, the spring phytoplankton bloom may occur earlier and may not be available to zooplankton, which would reduce zooplankton productivity and result in a subsequent decrease in fisheries production (Johnson, 2016).

Studies indicate that sustained periods of warming can elevate metabolic costs to organisms, reduce available energy to higher trophic level fishes, and ultimately change the trophic structure of the ecosystem (Anderson & Piatt, 1999; Brodeur & Daly, 2019; Clark et al., 2010; Johnson, 2016; National Oceanic and Atmospheric Administration, 2020a; Overland & Wang, 2007; Schwing et al., 2010; von Biela et al., 2019; Zador et al., 2019). The 1977 shift to a warmer climate regime in the North Pacific (Pacific Decadal Oscillation) was accompanied by an increase in zooplankton, salmon, cod, and pollock production, but it also brought steep declines in forage fish, crab, and shrimp (Johnson, 2016). For many years, these type of Pacific Decadal Oscillation regime shifts served as useful indices for understanding climate variability and predicting fish productivity and distribution patterns. However, as described below, climate change appears to be causing more extreme variations of ocean temperatures and wind patterns and are making correlations between Pacific Decadal Oscillation regime changes and biological variables more difficult to predict (National Oceanic and Atmospheric Administration, 2020b).

Over the past several years, the mass of warm water in the Gulf of Alaska (called the “Blob”) that impacted marine fish species and ecosystems the entire length of the U.S. West Coast, reduced the availability of phytoplankton and zooplankton. Scientists have associated a marked absence of pollock larvae surviving into late summer with those portions of the Gulf of Alaska affected by the Blob (Johnson, 2016). Although some scientists believe that the Blob could be associated with a particularly warm Pacific Decadal Oscillation phase, based on the different mechanism involved it is more likely a factor of human-caused climate change and potentially representative of future climate change impacts (Freedman & Tierney, 2019; Liang et al., 2017). The warm surface waters inhibit nutrients from being mixed into the surface layer to fuel production of phytoplankton. Furthermore, warm-water currents off the Pacific Coast bring southern varieties of zooplankton, particularly copepods, which have low lipid (fat) content and are less nutritious to fish and birds than the normally available northern varieties of copepods and krill. Increases in euphausiid (krill) abundance have been strongly correlated with cold temperatures in the Bering Sea (National Oceanic and Atmospheric Administration, 2019; Ressler et al., 2014), but not in the Gulf of Alaska (Simonsen et al., 2016). However, “cold water” copepods have been found to be more abundant during cold-water periods (Keister et al., 2011), so it remains to be seen how climate change will affect the production of northern copepods and krill in the Gulf of Alaska that provide high-energy nutrition to pollock and salmon.

The Blob has also significantly reduced the Pacific cod *(Gadus microcephalus)* population in the Gulf of Alaska through an increase in metabolic demand and reduced prey supply (Gisclair, 2019). With cod stocks falling, the North Pacific Fishery Management Council (NPFMC) set an 80 percent reduction in the catch limit in 2018 and an additional reduction of 5 percent in 2019. The NPFMC completely closed the directed fishery in 2020 and reduced cod bycatch limits for other fisheries. These rules were enacted to reduce overfishing, avoid long-term population-level effects, and protect Steller sea lions, which rely on cod for prey (Gisclair, 2019).

Ocean acidification, a climate change related process where increasing atmospheric carbon dioxide concentrations are reducing ocean pH and carbonate ion concentrations, may have serious impacts on fish development and behavior (Raven et al., 2005). Ocean acidification is expected to progress faster and more severely in Alaska than lower latitudes because cold Alaskan waters hold more carbon dioxide year-round and have a high baseline concentration of carbon dioxide (Alaska Ocean Acidification Network, 2019). Acidification of seawater reduces the amount of calcium carbonate minerals needed for shell-building organisms to build and maintain their shells, which poses a danger for species such as crab, clams, pelagic calcifying snails (pteropods) and some types of zooplankton. Changes in ocean chemistry can also affect fish. For instance, higher acidity water has been shown to reduce the ability for some fish to detect predators (Alaska Ocean Acidification Network, 2019).

Pteropods are a key food source for salmon, herring, and other fish in the Gulf of Alaska (Alaska Ocean Acidification Network, 2019; Johnson, 2016). Extensive shell dissolution has been documented in pteropods in both the Gulf of Alaska and the Bering Sea (Alaska Ocean Acidification Network, 2019). It has been estimated that a 10 percent decrease in pteropods could cause a 20 percent decrease in body weight of adult pink salmon (Chapin III et al., 2014; Johnson, 2016).

Azumaya and Urawa (2019) found that the distribution of chum salmon in the North Pacific in summer has shifted northward and the area of chum distribution has decreased approximately 5 percent during the last 36 years due to recent warming trends. A National Oceanic and Atmospheric Administration study found that Pacific cod shift abundance to deeper water in warm years (Johnson, 2016). Predation, competition, and disease are likely to have a greater negative impact as northern seas warm. While researchers have found that naturally occurring salmon sharks *(Lamna ditropis)* regularly contribute to high mortality rates of Chinook salmon in the Bering Sea (Seitz et al., 2019), more southern species of sharks (e.g., great white sharks [*Carcharodon carcharias*], common thresher sharks (*Alopias vulpinus*]) tend to occur more frequently in the Gulf of Alaska during particularly warm years and are very effective predators on salmon and herring (Johnson, 2016). Pacific pomfret (*Brama japonica*), and possibly Pacific mackerel (*Scomber japonicus*), have appeared in Alaska waters; they prey on juvenile salmonids and are aggressive competitors for the same prey resources (Johnson, 2016).

Due to the dynamic factors associated with climate change, effects on salmonids are difficult to predict. Studies and modeling have shown that climate change could result in a range of beneficial and adverse effects. The impacts on climate change on West Coast freshwater spawning and rearing habitats have been studied thoroughly and are expected to present significant challenges for salmonids (Crozier & Siegel, 2017). However, salmonids that use the Gulf of Alaska may benefit from increased primary productivity in the ocean, even though results of some research suggest that higher quality prey, like the more lipid-rich copepods that predominate during cooler water phases, tend to produce higher juvenile salmon survival. This is particularly the case if migration timing and plankton bloom timing are in phase (Johnson, 2016).

Since the 1977 regime shift to a warmer phase, pink, chum, and sockeye have been more productive, while coho and Chinook did not respond so positively. This is likely because these salmonids migrate to the ocean early in their life when they are small and, thus, feed on lower trophic levels for a longer period of time than do Chinook and coho. Therefore, almost all their biomass is accumulated in the marine environment (Irvine & Fukuwaka, 2011). Some stocks may spread into or become more firmly established in arctic waters with warming temperatures. For example, Larson et al. (2013) used genetic stock identification to show that a significant portion of stocks from California to Southeast Alaska overwinter in the Gulf of Alaska, then travel northward to the continental shelf region of the eastern Bering Sea during spring and summer. This migration pattern is thought to be driven by warm summer temperatures in the Gulf of Alaska, which promote northward movement towards the cooler and more productive Alaskan continental shelf. With temperatures rising in the Gulf of Alaska due to climate change, it is possible that this region will become even less hospitable to salmonids during the summer months, increasing the proportion of salmon stocks that spend the summer in the Bering Sea (Abdul-Aziz et al., 2011; Myers et al., 2007). Future research incorporating similar data could provide direct evidence of shifting salmonid migration patterns in response to climate change.

High-seas salmon have also shown the ability to adapt to climate-induced changes in their prey resources by switching their diets either within or between trophic levels (Brodeur & Daly, 2019; Fergusson et al., 2019; Kaeriyama et al., 2004). However, on the other hand, an extreme interpretation of models based on climate change scenarios predicts that by 2100 the ocean winter habitat of Pacific Northwest sockeye salmon would decrease by 38 percent and summer habitat for Chinook by 86 percent, sockeye by 45 percent, 30 percent for coho, 30 percent for pink, and 29 percent for chum (Abdul-Aziz et al., 2011; Johnson, 2016). Projected losses would be greatest in the Gulf of Alaska and may include nearly complete loss of habitat for sockeye (Abdul-Aziz et al., 2011). Recent and ongoing field work and modeling by the National Oceanic and Atmospheric Administration suggests that the manifestations of warming in the Gulf of Alaska will continue, highlighting the need for continued research and monitoring of conditions and biological responses to these changes (National Oceanic and Atmospheric Administration, 2020a; Zador et al., 2019).

Commercial and Recreational Fishing

For information on updated data for commercial and recreational fishing in the state of Alaska see Section 3.11 (Socioeconomic Resources and Environmental Justice). Commercial and recreational fishing can adversely affect fish populations, other species, and habitats. Potential impacts of fishing include overfishing of targeted species, bycatch, entanglement, and habitat modification. Bycatch is the capture of fish, marine mammals, sea turtles, seabirds, and other nontargeted species that occur incidentally to normal fishing operations. Fisheries bycatch has been identified as a primary driver of population declines in several marine species, including sharks, mammals, seabirds, and sea turtles (Wallace et al., 2010). Use of mobile fishing gear such as bottom trawls disturb the seafloor and may reduce habitat structural complexity. Indirect impacts of trawls were described in the 2011 Final GOA EIS/OEIS and include increased turbidity, alteration of surface sediment, removal of prey (leading to declines in predator abundance), and removal of predators (Hamilton Jr., 2000). Lost gill nets, purse seines, and long-lines may foul and disrupt bottom habitats and have the potential to entangle or be ingested by marine animals (National Marine Fisheries Service, 2017).

In addition to being subject to capture in fisheries closer to their natal rivers, federally listed salmonids are caught in several fisheries that operate in GOA waters. These fisheries include the following: groundfish fisheries managed by NMFS under the FMP for Groundfish of the Gulf of Alaska (North Pacific Fishery Management Council, 2019); salmon fisheries under the FMP for the Salmon Fisheries in the Exclusive Economic Zone (EEZ) off Alaska (North Pacific Fishery Management Council et al., 2018); Pacific salmon fisheries that operate under the Pacific Salmon Treaty between the United States and Canada (Pacific Salmon Commission, 2020); and State of Alaska-managed commercial, recreational (personal use), sport, and subsistence fisheries for Pacific salmon that operate in the Gulf of Alaska. State fisheries do not operate in the TMAA so are not further discussed. Updates to the FMPs are provided in Section 3.6.2.9 (Essential Fish Habitat).

Groundfish fisheries do occur in the TMAA and are known to incidentally capture ESA-listed salmonids (Dorn et al., 2019; Guthrie III et al., 2019; Guthrie III et al., 2020; Masuda, 2019; Masuda et al., 2019; Schnaittacher & Narita, 2019, 2020). Annual prohibited species catch limits in groundfish fisheries are established by the NPFMC for Chinook salmon in the central and western GOA. The National Marine Fisheries Service (2017b) has indicated that only a small percentage of these fish would be expected to be from ESA-listed populations.

Marine Debris and Pollution

Marine debris is any anthropogenic object intentionally or unintentionally discarded, disposed of, or abandoned in the marine environment. Common types of marine debris include various forms of plastic and abandoned fishing gear, as well as clothing, metal, glass, and other debris. Marine debris degrades marine habitat quality and poses ingestion and entanglement risks to marine life (National Marine Fisheries Service, 2006).

Plastic marine debris is a major concern because it degrades slowly and many plastics float, allowing the debris to be transported by currents throughout the oceans. Currents in the oceanic convergence zone in the North Pacific Subtropical Gyre act to accumulate the floating plastic marine debris. These debris-carrying currents include the south-flowing California Current and the north-flowing GOA Current. These currents distribute debris throughout the TMAA.

A major concern associated with plastic waste is degradation into microplastics, which are consumed by zooplankton and various filter feeders (e.g., oysters) and then bioaccumulate. Some fish and marine organisms have been shown to fill up their stomachs with indigestible material and then starve to death because they feel full but haven't received the nutrition they require (Jambeck, 2018; Prinz & Korez, 2019). Microplastics can also alter the behavior of fish, with those that ingest the pollutants likely to be bolder, more active, and swim in risky areas, which can lead to mortality (McCormick et al., 2020).

Additionally, plastic waste in the ocean chemically attracts hydrocarbon pollutants such as polychlorinated biphenyl (PCB) and dichlorodiphenyltrichloroethane (DDT), which accumulate up to one million times more in plastic than in ocean water (Mato et al., 2001). Marine animals can mistakenly consume these wastes, which contain elevated levels of toxins, instead of their prey. In the North Pacific Subtropical Gyre, it is estimated that the fishes in this area are ingesting 12,000–24,000 U.S. tons (10,886,216–21,772,433 kilograms) of plastic debris a year (Davison & Asch, 2011).

Debris that sinks to the seafloor is also a concern for ingestion and entanglement of fish and contributes to marine habitat degradation. West Coast groundfish bottom trawl surveys in 2007/2008 found anthropogenic debris at depths of 55–1,280 m, and the density increased with depth. The majority of debris was plastic and metallic while the rest was composed of fabric and glass (Keller et al., 2010).

Offshore petroleum production and local, transitory pollution events such as oil spills pose some degree of risk. Offshore petroleum production and large-scale transport of petroleum occurs in the Alaska EEZ, although at this time there is no offshore production of petroleum in the commercial troll area of the EEZ (North Pacific Fishery Management Council et al., 2018). Offshore oil and gas development and transport will inevitably result in some oil entering the environment at levels exceeding background amounts. The Exxon Valdez oil spill was shown to have direct effects on the survival, fitness, and habitats of pink salmon and herring (Rosen, 2017). The herring population in Prince William Sound crashed in 1993, just four years after the Exxon Valdez oil spill, and has yet to recover. Scientists have not been able to determine if and how the spill played a role in the collapse of the herring population (Robertson & Pegau, 2018; Rosen, 2017). Chinook salmon were not directly affected, because of their different habitat utilization in the spill area (North Pacific Fishery Management Council, 2018b). In general, the early life history stages of fish are more susceptible to oil pollution than juveniles or adults (North Pacific Fishery Management Council, 2018b).

3.6.2.2 Chinook Salmon (*Oncorhynchus tshawytscha*)

3.6.2.2.1 Status and Management

Since the 2016 GOA Final SEIS/OEIS, NMFS has responded to petitions to list the Upper Klamath-Trinity River Chinook Salmon ESU (83 FR 8410) and Oregon Coast spring-run Chinook Salmon ESU (85 FR 20476) as threatened or endangered species under the ESA (Table 3.6-1). NMFS determined that the petitions present substantial scientific information indicating that the petitioned actions may be warranted and plans to conduct status reviews of both Chinook salmon ESUs. There have been no other listing status changes to ~~other~~ Chinook salmon ESUs since 2016.

3.6.2.2.2 Distribution

Chinook salmon distribution in marine waters varies seasonally and inter-annually due to a variety of environmental factors (Pacific Fishery Management Council, 2014). However, there are general migration and ocean distribution patterns characteristic of populations in specific geographic areas (North Pacific Fishery Management Council et al., 2018). Chinook populations originating from river systems north of Cape Blanco, Oregon, tend to migrate north and westward along the Pacific coast, whereas those originating south of Cape Blanco tend to migrate west and south to forage in waters off Oregon and California (Masuda, 2019; North Pacific Fishery Management Council, 2018b; Quinn & Myers, 2005; Sharma, 2009b). As such, southern stocks (south of Cape Blanco) are less likely to use habitats in the Gulf of Alaska than northern stocks. However, as described in Section 3.6.2.7 (Site-Specific Information on Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area), juveniles from southern ESUs have been documented in the Gulf of Alaska, so it is possible that some individuals from southern populations could migrate into the TMAA.

Listed spring-run Chinook salmon from northern West Coast ESUs that originate from the Columbia River Basin are more likely to migrate into the Gulf of Alaska and the TMAA than other listed Chinook salmon. Upper Columbia River spring-run and Snake River spring/summer-run ESUs are more common off the continental shelf and in the open ocean during their marine residence (Masuda, 2019; Quinn, 2018; Sharma, 2009a), whereas other spring-run ESUs are primarily distributed on the shelf (Sharma, 2009a).

Listed fall- and summer-run Chinook salmon from West Coast ESUs tend to be primarily distributed along the continental shelf during their marine residence, remaining in coastal water throughout their ocean life (Sharma, 2009b). After emigrating from their natal streams, juveniles spend several months rearing in nearshore estuarine habitat, before moving onto the continental shelf. A recent study has shown that most juvenile Chinook captured off the Southeast Alaska coast originate from Columbia River spring-run stocks (Van Doornik et al., 2019). Columbia River fall Chinook generally undertake a rapid northward migration, but very few are recovered north of Vancouver Island (Trudel et al., 2009).

The vast majority of juvenile Chinook salmon in the Gulf of Alaska occur on the continental shelf, mostly in the inside waters of the Alexander Archipelago (Echave et al., 2012; National Marine Fisheries Service, 2017), although some Chinook move offshore by late summer (Brodeur et al., 2003). Immature Chinook salmon are also predominantly found on the continental shelf in the Gulf of Alaska, though they are distributed more widely throughout the Gulf of Alaska than juveniles (Echave et al., 2012; National Marine Fisheries Service, 2017). Most mature adults in the Gulf of Alaska are found along the outer coast and inside waters of the Alexander Archipelago. Echave et al. (2012) reported a relatively high abundance of mature Chinook salmon within Southeast Alaska waters (outside of the TMAA), likely because the surveys were conducted when the Chinook were returning to spawn. The offshore distribution off the southern end of the Archipelago was observed during winter sampling, when mature fish are more likely to be offshore in oceanic habitats.

Chinook salmon do not concentrate at the surface, as do other Pacific salmon, but are most abundant at depths of 30–70 meters (m) (North Pacific Fishery Management Council et al., 2018). However, juvenile Chinook salmon tend to be more abundant than adults near the surface, most frequently found at depths of less than 30 m (Fisher & Pearcy, 1995; Orsi & Wertheimer, 1995). Juvenile salmonids are not known to congregate in large schools in marine habitats (Moulton, 1997; Pearcy & Fisher, 1990). Preliminary evidence from the 2019 GOA Expedition suggests that adult salmonids may congregate in schools during the winter months (International Year of the Salmon, 2019).

Site-specific presence of ESA-listed Chinook salmon ESUs in the Gulf of Alaska, including CWT recoveries, is described in Section 3.6.2.7 (Site-Specific Information on ESA-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area). With the exception of the two new candidate ESUs under review, some updated information on Chinook distribution and migratory patterns, and the site-specific presence information, the information presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid. Table 3.6-2 shows the temporal patterns and horizontal/vertical distribution of ESA-listed fish species in the TMAA.

3.6.2.3 Coho Salmon (*Oncorhynchus kisutch*)

3.6.2.3.1 Status and Management

There has been no change in the status or management of coho salmon ESUs since the 2016 GOA Final SEIS/OEIS.

3.6.2.3.2 Distribution

After leaving their natal rivers, juvenile coho tend to use the cool, upwelled waters of the continental shelf for migration and feeding (Bellinger et al., 2015). In the Gulf of Alaska, juvenile coho predominantly occur in coastal waters, throughout the continental shelf and slope (Echave et al., 2012), with some coho moving offshore by late summer (Brodeur et al., 2003; North Pacific Fishery Management Council et al., 2018). Morris et al. (2007) found that juvenile coho from the lower Columbia River and coastal Oregon were recovered in or near the TMAA. Coho juveniles are generally found within the upper 30 m of the water column, with the majority in the top 10–15 m, which is shallower than most Chinook juveniles (North Pacific Fishery Management Council et al., 2018; Orsi & Wertheimer, 1995).

Table 3.6-2: Temporal Patterns and Horizontal/Vertical Distribution of ESA-Listed Fish Species in the Gulf of Alaska and the TMAA

Common Name (Scientific Name)	*Temporal Patterns*	*Horizontal Distribution*	*Vertical Distribution*
Chinook Salmon (*Oncorhynchus tshawytscha*)	Juveniles: Mid-summer to early fall	Juveniles: Majority distributed on continental shelf, mostly in the inside waters of SE Alaska. Smaller abundances found throughout inner and outer shelf waters off Montague Island.	Juveniles: More abundant than adults near the surface, typically at depths less than 30 m (Fisher & Pearcy, 1995; Orsi & Wertheimer, 1995).
	Immature Adults: Year-round	Immature Adults: Mostly distributed on the shelf to just beyond the outer shelf. More widely distributed than juveniles.	Immature Adults: Same as maturing adults.
	Maturing Adults: Mature fish leave in September.	Maturing Adults: Majority within outer coast and inside waters of SE Alaska. Higher relative abundance in SE Alaska waters in summer. More likely to use offshore habitats in winter.	Maturing Adults: Less surface oriented than other Pacific salmon. Deeper depths than juveniles (typically 30–70 m).
Coho Salmon (*Oncorhynchus kisutch*)	Juveniles: June to September	Juveniles: Predominantly occur in coastal waters, throughout the continental shelf and slope. Move offshore by late summer.	Juveniles: Generally shallower than Chinook with majority found at depths of 10–15 m (North Pacific Fishery Management Council, 2018a; Orsi & Wertheimer, 1995).
	Immature/Maturing Adults: Year-round. Mature fish leave in late summer	Immature/Maturing Adults: Continental shelf and beyond into offshore waters.	Immature/Maturing Adults: Primarily within upper 30 m (Walker et al., 2007).
Chum Salmon (*Oncorhynchus keta*)	Juveniles: July to September	Juveniles: Distributed throughout the inner and middle shelf. By the end of their first fall at sea, most fish have moved into offshore waters.	Juveniles: Mostly in top 15 m of water column (Beamish et al., 2007b).
	Immature/Maturing Adults: Year-round. Mature fish leave in early fall.	Immature/Maturing Adults: Distributed throughout the outer portion of the shelf and as far offshore as the U.S. EEZ boundary.	Immature/Maturing Adults: Majority found at 0–30 m depths (Walker et al., 2007).

Table 3.6-2: Temporal Patterns and Horizontal/Vertical Distribution of ESA-Listed Fish Species in the Gulf of Alaska and the TMAA (continued)

Common Name (Scientific Name)	*Temporal Patterns*	*Horizontal Distribution*	*Vertical Distribution*
Sockeye Salmon (*Oncorhynchus nerka*)	Juveniles: Early summer to late winter	Juveniles: Distribution generally contained to the continental shelf.	Juveniles: Shallowest depths of any salmonids (Walker et al., 2007). Mostly found within top 15 m of water column (Beamish et al., 2007a) and within top 5 m in some areas (Walker et al., 2007).
	Immature: Year-round	Immature: Distributed from nearshore waters to the U.S. EEZ boundary.	Immature: Surface-oriented
	Adults: Mature fish leave in early August	Adults: Occur in relatively low abundances extending out to the U.S. EEZ boundary.	Adults: Generally surface oriented (upper 10 m) (Walker et al., 2007).
Steelhead (*Oncorhynchus mykiss*)	Juveniles: Summer to fall	Juveniles: Offshore migration through North Pacific to the western Gulf of Alaska.	Juveniles: Same as adults
	Immature/Maturing Adults: Year-round. Spawners leave in spring/summer	Immature/Maturing Adults: Offshore, widely distributed across North Pacific. May pass through the Gulf of Alaska but migrate south of TMAA when returning to spawn (Light et al., 1989).	Immature/Maturing Adults: Surface-oriented (0–10 m) (Light et al., 1989).
Green Sturgeon (*Acipenser medirostris*)	Subadults and adults: Fall and winter	Subadults and adults: Likely widely distributed over the continental shelf (if present).	Subadults and adults: <200 m depth (primarily 40–110 m) (Erickson & Hightower, 2007; Huff et al., 2012).

Notes: SE = Southeast, EEZ = Exclusive Economic Zone, GOA = Gulf of Alaska, m = meter(s), TMAA = Temporary Maritime Activities Area

Sources: (Echave et al., 2012); (North Pacific Fishery Management Council et al., 2018); (National Marine Fisheries Service, 2017) (unless specified otherwise)

Immature and mature coho in the Gulf of Alaska occur along the continental shelf but also occur in offshore oceanic waters beyond the shelf break (Echave et al., 2012). The 2019 GOA Expedition found relatively high abundances of coho adults in the Gulf of Alaska offshore habitats (International Year of the Salmon, 2019). Immature and mature adults are generally found within the top 50 m but more typically occur at depths shallower than 20 m (Walker et al., 2007).

Updated information on site-specific presence of ESA-listed coho salmon ESUs in the Gulf of Alaska is described in Section 3.6.2.7 (Site-Specific Information on Endangered Species Act-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area). With the exception of the inclusion of updated distribution and migratory patterns and site-specific presence data, the information presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid.

3.6.2.4 Chum Salmon (*Oncorhynchus keta*)

3.6.2.4.1 Status and Management

There have been no listing status changes to chum salmon ESUs since 2016. In 2017, NMFS received a petition to list the winter-run Puget Sound chum salmon in the Nisqually River system and Chambers Creek as a threatened or endangered ESU under the ESA. Following a review completed in 2017, NMFS determined that winter-run chum salmon from these river systems do not qualify as an ESU and was not eligible for listing under the ESA (82 FR 33064).

3.6.2.4.2 Distribution

Chum generally move north and west along the coast upon entering saltwater and move offshore by the end of their first ocean year (Byron & Burke, 2014; Quinn, 2018). Some data suggest that Puget Sound chum, including those in the Hood Canal summer-run ESU, may not make an extended migration into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean (Hartt & Dell, 1986). Myers et al. (1996) documented maturing chum salmon from Washington and the Columbia River in offshore areas of the Gulf of Alaska, though only a small number of coded-wire tag recoveries were observed.

Within the Gulf of Alaska, juvenile chum salmon are distributed throughout the inner and middle shelf along the Gulf coastline between July and September (Echave et al., 2012), but that by the end of their first fall at sea, most fish have moved off the continental shelf into open waters (Quinn, 2018). Immature and mature chum salmon are distributed widely throughout the outer portion of the continental shelf and over oceanic waters as far offshore as the U.S. EEZ boundary (Echave et al., 2012).

Juvenile chum salmon are surface oriented and typically found within the top 15 m of the water column (Beamish et al., 2007b). In Southeast Alaska, juvenile chum salmon were observed near the surface as small aggregations (10–50 fish) but not in large schools (Moulton, 1997). However, immature and mature chum salmon have a deeper vertical distribution (second only to Chinook salmon). The majority are found at 0–30 m depths, but they have been captured as deep as 120 m in the Bering Sea (Walker et al., 2007).

Updated information on site-specific presence of ESA-listed chum salmon ESUs in the Gulf of Alaska is described in Section 3.6.2.7 (Site-Specific Information on Endangered Species Act-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area). With the exception of the inclusion of updated distribution and migratory patterns and site-specific presence data, the information presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid.

3.6.2.5 Sockeye Salmon (*Oncorhynchus nerka*)

3.6.2.5.1 Status and Management

There has been no change in the status or management of sockeye salmon ESUs since the 2016 GOA Final SEIS/OEIS.

3.6.2.5.2 Distribution

Sockeye tend to follow a similar migration pattern as chum once they enter the ocean, moving north and west along the coast, and may move offshore or stay inshore at the end of their first ocean year (Beacham et al., 2014; Byron & Burke, 2014; Quinn, 2018). In the Gulf of Alaska, the distribution of juvenile sockeye salmon is generally contained to the continental shelf (Echave et al., 2012). Immature sockeye are distributed from the nearshore waters to the U.S. EEZ boundary throughout the entire Gulf of Alaska (Echave et al., 2012). Similarly, mature sockeye occur in relatively low abundances extending from coastal waters to the U.S. EEZ boundary (Echave et al., 2012). Myers et al. (1996) documented maturing sockeye salmon from Washington and the Columbia River in offshore areas of the Gulf of Alaska.

Juvenile sockeye are generally found in the top 15 m of the water column (Beamish et al., 2007a). An analysis of juvenile salmonids from 2,968 sets between 0 and 60 m in depth in coastal BC showed that 80–87 percent of sockeye salmon were captured in the top 15 m (Beamish et al., 2007a). Depth data from a limited number of data storage tags in the North Pacific Ocean indicated that sockeye juveniles had the shallowest vertical distribution of any salmonids (regularly found within the top 5 m of the water column) (Walker et al., 2007). Immature and mature sockeye are generally surface oriented (upper 10 m) but have been found up to 80 m in the Bering Sea (Ogura & Ishida, 1995; Quinn et al., 1989; Walker et al., 2007).

The information regarding sockeye salmon presented in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS remains valid. Therefore, no additional updates are required.

3.6.2.6 Steelhead (*Oncorhynchus mykiss*)

3.6.2.6.1 Status and Management

There have been no steelhead listing status changes since the 2016 GOA Final SEIS/OEIS was issued. In February 2020, NMFS responded to a petition to list the Northern California summer-run steelhead as an endangered DPS under the ESA (85 FR 6527) (Table 3.6-1). Based on the best scientific and commercial data available, including the DPS configuration review report, NMFS determined that: (1) listing Northern California summer-run steelhead as an endangered DPS was not warranted; and (2) summer-run steelhead do not meet the criteria to be considered a separate DPS from winter-run steelhead (85 FR 6527). There have been no listing status changes to other steelhead DPSs since 2016.

3.6.2.6.2 Distribution

Steelhead are thought to rely heavily on offshore marine waters for feeding, with high seas tagging programs indicating steelhead make more extensive migrations offshore in their first year than any other Pacific salmonids (Quinn & Myers, 2005). Juveniles migrate rapidly through estuaries, bypass coastal migration routes of other salmonids, and move into oceanic offshore feeding (Daly et al., 2014; Quinn & Myers, 2005). McKinnell et al. (2011) assessed the distribution of North American hatchery steelhead stock in the Gulf of Alaska and Aleutian Islands using CWT mark and recapture data from 1981 through 1994. These data showed that tagged steelhead from hatcheries in the upper, middle, and lower Columbia River, the Snake River basin, coastal Washington, and Puget Sound were recaptured in offshore waters of the northern and southern Gulf of Alaska and the Aleutian Islands.

Tagging and diet studies indicate that adult and juvenile steelhead are surface oriented, spending most of their time in the top 10 m of the surface in oceanic feeding grounds off the continental shelf (Light et al., 1989). Steelhead adults may migrate within 1 m of the surface when returning over the shelf to their natal stream (Light et al., 1989).

Site-specific presence of ESA-listed steelhead DPSs in the Gulf of Alaska is described in Section 3.6.2.7 (Site-Specific Information on Endangered Species Act-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area). With the exception of the inclusion of updated distribution and migratory patterns and site-specific presence data, the information presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid.

3.6.2.7 Site-Specific Information on Endangered Species Act-Listed Salmonids in the Gulf of Alaska and Temporary Maritime Activities Area

Salmon Bycatch in the Groundfish Fishery

Chinook salmon incidentally taken in the pollock fishery historically account for the greatest proportion of Chinook salmon taken in the GOA groundfish fisheries (Schnaittacher & Narita, 2019, 2020). Chum salmon typically account for over 95 percent of the non-Chinook salmon catch, with the remainder consisting of smaller abundances of coho, pink, and sockeye salmon (Schnaittacher & Narita, 2019). These salmonids may comprise ESA-listed and nonlisted fish from Oregon and Washington as well as non-listed fish from British Columbia or Alaska.

Prior to 1998, salmon bycatch was identified to species. Since then, annual estimates of non-Chinook salmon have been combined (Schnaittacher & Narita, 2019). Salmon bycatch generally occurs on vessels fishing with trawl gear. Other gear used to harvest groundfish, such as longline and pot, generally do not catch many salmonids. In the Gulf of Alaska, the majority of salmon bycatch occurs in the pollock trawl fishery, although other target fisheries for flatfish, rockfish, and Pacific cod also can capture Chinook salmon. The incidental harvest of Chinook salmon from federally managed groundfish fisheries in the Gulf of Alaska averaged 21,389 salmon per year from 1990 to 2019, ranging from a low of 8,475 individuals in 2009 to a peak of 54,696 in 2010 (Schnaittacher & Narita, 2019). Comparatively, the number of "other" salmon captured in the GOA groundfish fisheries is relatively low (North Pacific Fishery Management Council, 2020). Over the past six years, non-Chinook bycatch in the Gulf of Alaska ranged from 1,320 (in 2015) to 9,149 (in 2018) salmon and averaged approximately 4,700 salmon (National Marine Fisheries Service, 2020a).

In 2018, the pollock trawl fishery contributed the largest component to Chinook salmon bycatch in the Gulf of Alaska with an estimated 14,820 fish. An additional 2,364 fish from the rockfish trawl and other fisheries increased the Chinook salmon bycatch total to an estimated 17,184 fish (Guthrie III et al., 2019). In 2019, the total incidental catch of Chinook salmon in the Gulf of Alaska from the groundfish fishery was 23,893 individuals and the incidental catch of non-Chinook salmon was 6,407 (Schnaittacher & Narita, 2019).

The estimated prohibited species catch of chum salmon in the Gulf of Alaska (National Marine Fisheries Service, 2016c) is one to two orders of magnitude lower than in the Bering Sea and has been a lower management priority than the typically larger catches of Chinook salmon (Guthrie III et al., 2017). In 2016, chum salmon samples were collected in the Gulf of Alaska, primarily from the pollock trawl fishery, which caught about 56 percent of the chum salmon prohibited species catch. The majority of chum salmon from the non-pollock fisheries were caught in the arrowtooth flounder, sablefish, rockfish, and halibut fisheries (Whittle et al. 2018).

For several years, the Bering Sea pollock industry has been working on developing a Chinook salmon excluder device for trawl gear, which allows salmon to escape from the trawl net underwater, while retaining pollock. The success of such devices relies on the different swimming behaviors and sensory capacities of pollock and Chinook salmon. Through experimental fishery permits authorized by the NPFMC and NMFS, various iterations have been tested, and their voluntary use by pollock skippers is increasing. Recently, the Gulf of Alaska pollock industry has begun to consider how the Bering Sea Chinook salmon excluder might be adapted for the smaller GOA pollock fleet (North Pacific Fishery Management Council, 2020).

In 2012, NMFS implemented Amendment 93 to the GOA Groundfish FMP, which required retention of salmon by all vessels in the GOA pollock fisheries until the catch is delivered to a processing facility where an observer can collect genetic samples and screen for CWTs (77 FR 42629) (National Marine Fisheries Service, 2019). Genetic and CWT data are used for many purposes, including stock contribution studies, in order to better manage harvest rates for conservation of the resource and provide documentation of ESA-listed fish to support ESA Section 7 consultations (Nandor et al., 2010).

Genetic Sampling

In 2013, NMFS restructured the North Pacific Observer Program when it implemented Amendment 76 to the GOA Groundfish FMP. Observer coverage and deployment are no longer based on vessel length and processing capacity; rather, NMFS now has the flexibility to decide when and where to deploy observers based on a scientifically defensible sampling design. The design of the new program serves to reduce sources of bias that jeopardized the statistic reliability of catch and bycatch data collected by the North Pacific Observer Program (Schnaittacher & Narita, 2019).

North Pacific fisheries observers enumerate all non-target species bycatch (including salmon) using a whole-haul or systematic subsampling process, as appropriate. Starting in 2013, the Alaska Groundfish Data Bank implemented a census approach whereby genetic samples and biological information were collected from every Chinook salmon encountered as bycatch in the rockfish trawl fisheries. In 2014, the North Pacific Observer Program implemented a simple random sampling protocol for the collection of genetic Chinook salmon samples for the trawl fisheries for walleye pollock in the Gulf of Alaska (Faunce et al., 2014). Since then, there have been many iterations of the sampling design (Faunce, 2015). Now, observers are required to collect a genetic sample from every Chinook and chum specimen encountered in the pollock fishery (Alaska Fisheries Science Center, 2019). The majority of the Chinook and chum salmon bycatch genetic tissue samples are derived from the bottom and midwater pollock trawl fishery (Guthrie III et al., 2020; Whittle et al., 2018).

In 2018, 15 percent of the estimated Chinook salmon bycatch from the pollock fishery were successfully genotyped (Guthrie III et al., 2020). During this year, bycatch samples were collected from trawling conducted off Kodiak Island, just west of the TMAA. Based on analysis of 2,226 Chinook salmon samples from a total bycatch of 14,820 fish, British Columbia (43 percent; 6,433), U.S. West Coast (33 percent; 4,846), and Coastal Southeast Alaska (18 percent; 2,728) stock groups comprised the largest regional contributions. In 2016, 473 chum salmon samples were analyzed from the GOA groundfish fisheries; the highest proportion was from Eastern Gulf of Alaska/Pacific Northwest (93 percent) stocks, similar to previous years (Whittle et al., 2018).

Genetic samples from the GOA rockfish fishery bycatch were also collected in 2018 (Guthrie III et al., 2020). Based on the genotyping of 504 Chinook salmon bycatch samples collected from this fishery in NMFS Statistical Area 630 (central Gulf of Alaska area that overlaps the shelf portion of the TMAA), the U.S. West Coast region had the largest contribution (53 percent: 264) with smaller contributions from British Columbia (28 percent; 141), and Coastal Southeast Alaska (11 percent; 54) regions. The 2018 GOA stock composition estimates for Chinook salmon bycatch in both the trawl and rockfish fisheries follow a similar trend observed in recent years with most (>90 percent) Chinook salmon encountered originating from three large southern regions between coastal Southeast Alaska and northern California. This pattern also holds for samples collected across finer-scale time and area strata within the Gulf of Alaska (Guthrie III et al., 2020).

Two primary factors dictate the observed trends in genetic stock composition of trawl fishery bycatch in the Gulf of Alaska. First, British Columbia and U.S. West Coast systems produce orders of magnitude more Chinook salmon each year than Alaska systems, yielding the much greater proportion of these stocks. Second, the timing of the fisheries may also drive some of these trends. British Columbia and U.S. West Coast stocks have both spring and fall runs of Chinook salmon, which may lead to the presence of greater overlap with trawl fisheries in the Gulf of Alaska, as compared to Alaskan stocks, which are dominated by a spring out-migration of smolts, reducing periods of potential overlap with trawl fleets (Zador et al., 2018).

Recent CWT Studies

CWT studies were reviewed to examine the potential for salmon bycatch captured in the Gulf of Alaska groundfish fishery to include ESA-listed fish (Masuda, 2019; Masuda et al., 2019). In 2019, NMFS prepared an annual report on the stock of origin and CWT data from incidental catch of salmon in 2018 (Masuda, 2019). The report included maps showing the ocean distribution of CWT Chinook salmon from ESA-listed ESUs from the Pacific Northwest. These maps were compiled from the historical database of CWT recoveries (1981–2018) from high seas commercial fisheries and research surveys: GOA groundfish fisheries, GOA rockfish trawl fishery, at-sea Pacific hake trawl fishery off the U.S. West Coast, and the West Coast groundfish trawl fishery, as well as domestic and foreign research surveys in the North Pacific Ocean and the Gulf of Alaska (Masuda, 2019). It should be noted that these fisheries are predominantly on-shelf and, while they may overlap a portion of the nearshore portion of the TMAA, the data will be biased toward those areas where these groundfish fisheries occur, thus providing an incomplete representation of salmonid occurrence in the TMAA.

Since the late 1960s, CWTs have been used in the greater Pacific region (Alaska, British Columbia, Washington, Idaho, Oregon, and California) to mark anadromous salmonids (Nandor et al., 2010). Over 50 million Pacific salmonids with CWTs are released yearly by 54 federal, provincial, state, tribal, and private entities (North Pacific Fishery Management Council, 2018b). Although some tagging of wild stocks occurs (mainly in Alaska), CWTs are used mostly for tagging of hatchery fish. As such, wild stocks of Chinook salmon are generally under-represented by CWTs, especially outside of Alaska.

Despite region-wide usage, CWT sampling coverage does have some limitations (Nandor et al., 2010). Chinook and coho salmon are the only species sampled in commercial and sport fisheries on a coastwide basis. Some sampling does occur for chum, sockeye, pink salmon, and steelhead, but it is much more limited. In such cases, it typically involves agency-specific management objectives in marine terminal areas or limited freshwater areas. Nevertheless, CWTs remain the only stock identification tool that is Pacific coastwide in scope and provides unparalleled information about ocean distribution patterns, fishery impacts, and survival rates for listed Pacific salmon (Nandor et al., 2010). Table 3.6-3 is a summary of CWT recoveries for various adult and juvenile salmonids in the Gulf of Alaska.

As expected, most of the CWT recoveries in the Gulf of Alaska consist of spring-run Chinook from northern West Coast ESUs. The most frequently detected CWT Chinook salmon recovered in the GOA groundfish fisheries have originated from the Upper Willamette River ESU (n=200) and the Lower Columbia River ESU (n=38) (Table 3.6-3). These Chinook have been detected throughout the northern GOA, including offshore areas off Kodiak Island, along the Aleutian Islands, and into the Bering Sea (Masuda, 2019). Relatively high abundances of Snake River spring/summer-run Chinook have also been detected in U.S. research surveys (Table 3.6-3). It was somewhat surprising to find a relatively large number of Snake River fall-run Chinook in the GOA, along with a rare individual from the Central Valley spring-run Chinook ESU. It was not surprising to find a few coho migrating through the Gulf of Alaska, as they tend to utilize offshore areas during their marine residence.

Table 3.6-3: CWT Recoveries of ESA-Listed Salmonids in the Gulf of Alaska

Species	ESU	Federal Status	Adult or Juvenile	Number	Type of Study	Survey Year	Reference
Chinook Salmon	Puget Sound ESU	T	Adult	1	Rockfish trawl fishery	2013–2018	(Masuda, 2019)
			Juvenile	1	NMFS research surveys	1996–2017	(Masuda, 2019)
	Upper Columbia River Spring-run ESU	E	Adult	1	Groundfish fisheries	1981–2018	(Masuda, 2019)
				1	Rockfish trawl fishery	2013–2018	(Masuda, 2019)
			Juvenile	27	NMFS research surveys	1996–2017	(Masuda, 2019)
	Lower Columbia River ESU	T	Adult	38	Groundfish fisheries	1981–2018	(Masuda, 2019)
				2	Rockfish trawl fishery	2013–2018	(Masuda, 2019)
			Juvenile	11	NMFS research surveys	1996–2017	(Masuda, 2019)
	Snake River Spring/Summer-run ESU	T	Adult	1	Groundfish fisheries	1981–2018	(Masuda, 2019)
				2	Rockfish trawl fishery	1981–2018	(Masuda, 2019)
			Juvenile	41	NMFS research surveys	1996–2017	(Masuda, 2019)
	Snake River Fall-run ESU	T	Adult	7	Groundfish fisheries	1981–2018	(Masuda, 2019)
				6	Rockfish trawl fishery	2013–2018	(Masuda, 2019)
			Juvenile	6	NMFS research surveys	1996–2017	(Masuda, 2019)
	Upper Willamette River ESU	T	Adult	200	Groundfish fisheries	1981–2018	(Masuda, 2019)
				28	NMFS research surveys	1996–2017	(Masuda, 2019)
			Juvenile	8	Rockfish trawl fishery	2013–2018	(Masuda, 2019)
	Central Valley Spring-run ESU	T	Adult	3	Groundfish fisheries	1995–1999	(Myers et al., 1999)

Table 3.6-3: CWT Recoveries of ESA-Listed Salmonids in the GOA (continued)

Species	ESU	Federal Status	Adult or Juvenile	Number	Type of Study	Survey Year	Reference
Coho salmon	Lower Columbia River ESU	T	Juvenile	1	NMFS research surveys	1996–2017	(Masuda, 2019)
				17	Fisheries and Oceans Canada and NMFS research surveys	1995–2004	(Morris et al., 2007)
	Oregon Coast ESU	T	Juvenile	2	Canadian Research surveys	1981–2005	(Myers et al., 2005)
				3	Fisheries and Oceans Canada and NMFS research surveys	1995–2004	(Morris et al., 2007)
Steelhead	Puget Sound DPS	T	Mixed	NA	Groundfish fisheries	1956–1989	(Burgner et al., 1992)
	Upper Columbia River DPS	T	Mixed	NA	Groundfish fisheries	1956–1989	(Burgner et al., 1992)
	Middle Columbia River DPS	T	Mixed	NA	Groundfish fisheries	1956–1989	(Burgner et al., 1992)
	Lower Columbia River DPS	T	Mixed	NA	Groundfish fisheries	1956–1989	(Burgner et al., 1992)
	Snake River Basin DPS	T	Adult	1	Canadian Research surveys	1981–2005	(Myers et al., 2005)
	Upper Willamette River DPS	T	Mixed	NA	Groundfish fisheries	1956–1989	(Burgner et al., 1992)
	Central California Coast DPS	T	Adult	3	Archival tagging study	2004–2008	(Hayes et al., 2011)

Notes: DPS = Distinct Population Segment, ESU = Evolutionarily Significant Unit, NA= not available, E = Endangered, T = Threatened

Although chum and sockeye were not identified in the Gulf of Alaska (likely due to few CWT fish), it is likely that some fish from listed West Coast ESUs may be present in the Gulf of Alaska in low numbers. Studies have shown that steelhead from Washington and the Columbia River Basin are distributed throughout the high seas fishery with the distribution varying by season and age class, but the studies do not provide origins of individually tagged fish (Burgner et al., 1992; Myers et al., 2005). Since Oregon only tags Columbia River Basin steelhead, no Oregon Coast steelhead were detected in the Gulf of Alaska. There were no apparent differences in distribution in the Gulf of Alaska between coastal and interior stocks of steelhead.

Although Oregon and Washington steelhead are well represented in the Gulf of Alaska, California steelhead are not (Burgner et al., 1992; Light et al., 1989). California uses CWTs extensively for hatchery-released steelhead; however, no CWT recoveries from California steelhead have been reported in the Gulf of Alaska (Burgner et al., 1992; Masuda, 2019). The only presumed California steelhead presence in the Gulf of Alaska was based on archival tags (using water temperature data), which determined that Scott Creek kelts (from the Central California Coast DPS) migrated into the Gulf of Alaska (Hayes et al., 2011). Hayes et al. (2011) suggested that steelhead from the larger Sacramento-San Joaquin basin stay in coastal waters, while fish from the central to north California coast may be well represented in the high seas, but just not bearing CWTs. Southern steelhead populations tend to have a more southern offshore distribution. It is not surprising that no ESA-listed Southern California steelhead CWTs have been detected in the Gulf of Alaska. These stocks have very low abundance, few historically marked fish, and rarely leave the continental shelf of California (Barnhart, 1991). As such, the probability that Southern California stocks would be present in the Gulf of Alaska and the TMAA is very low.

2019 International GOA Expedition

Scientists estimate that one-third of all Pacific salmon overwinter in the Gulf of Alaska (International Year of the Salmon, 2019). Since there have been limited surveys, the factors influencing the declines and booms are not well known. In February–March 2019, Dr. Richard Beamish led an international research team to study the mechanisms affecting salmon in the Gulf of Alaska (International Year of the Salmon, 2019; Pakhomov et al., 2019). The primary goal of the International GOA Expedition was to evaluate whether salmon abundance is mostly determined by the end of the first ocean winter, as fish that grow faster in their first year tend to survive better. The expedition used DNA technology to identify the stock-specific rearing areas for all five species of salmon and determine their abundances and condition.

The initial findings are summarized below (International Year of the Salmon, 2019; Pakhomov et al., 2019):

- A preliminary abundance estimate calculated 55 million salmon in the Expedition study area.
- The study area ranged geographically from 47 degrees North (°N) to 57°N; the northern part of the study area overlapped the far southern, offshore portion of the TMAA.
    - Salmon species differed substantially in their distributions with some showing potential links to environmental conditions. Trawl net surveys captured a total of 425 salmon throughout the study area. The frequency of occurrence in trawl catches for all salmon species was 83 percent and individually as follows: chum (64 percent), coho (38 percent), sockeye (31 percent), pink (17 percent), and Chinook (5 percent).
    - Using a tested abundance catchability coefficient of 0.3 for adult salmon, the study estimated the following abundances: chum (27.7 million), coho (13.6 million), sockeye (9 million), pink (4.2 million), and Chinook (0.4 million).
    - Sockeye in the northern portion of the study area were associated with cooler waters compared to pink salmon that were captured in southern, warmer waters. Catches of sockeye were somewhat lower than expected. It is possible that some sockeye salmon over-winter farther west of the study area.
    - The GOA survey covered roughly 12 percent of potential pink salmon wintering area, but estimated abundance reached only 0.3 percent of estimated total pink salmon abundance. Radchenko (2020) speculates that pink salmon may be overwintering in the South Bering Sea which has experienced warmer ocean waters in recent years.

  - Chum salmon were most broadly distributed and caught in the majority of sets. Chum salmon were represented by all marine-age groups including fish of first marine year. It is likely that many of these fish originated from Asia (Dunagan, 2019).
  - Coho salmon were found at relatively high abundances and captured over 1,000 kilometers (km) (621 miles) offshore; they were previously thought to have a more coastal distribution.
  - Few Chinook salmon were captured, presumably because these fish are found in deeper waters than where trawling typically occurs(Dunagan, 2019).
  - It is likely that no steelhead were captured because they tend to be more surface oriented, and the trawl nets were deployed at depths too deep to capture them.
  - Fish condition varied over the study area and even within a single set. Fish condition was positively related to stomach fullness. Chum exhibited a range of conditions (from skinny to robust) within a single set. DNA analysis will help determine if the variability is due to stock origin.
- At sea-genetic sequencing provided real-time stock composition. Coho caught ranged from SE Alaska to the Columbia River, with the majority originating from British Columbia.
- Trawl net videos provided preliminary evidence indicating that some adult salmon tend to exhibit schooling rather than solitary feeding behavior, which was previously thought to be more prevalent in the Gulf of Alaska during the winter months.
- Stomach analyses to examine diet was conducted on all salmon captured during the survey. Key diet categories (by volume) included euphausiids, pteropods, larval fish, and squid.
- Few salmon predators were observed during the 2019 GOA Expedition, which is consistent with previous winter surveys; eDNA results will indicate whether major predators were present but not captured during the trawl surveys (Weitkamp, 2020).
- In the Gulf of Alaska, squid are particularly important in the diet of higher trophic level species (coho, Chinook, steelhead), while occupying an important trophic position as intra-guild prey of pink and sockeye salmon (Katugin et al., 2019). During the 2019 GOA Expedition spring trawl surveys, several pelagic squid species were regularly encountered but at different abundance levels. One potentially abundant species (*Okutania anonycha*) was absent from trawl catches, but it occurred exclusively in salmon stomachs, indicating that the surveys may have occurred too late in the season or at depths that were too shallow (Katugin et al., 2019). The 2019 GOA Expedition also found large aggregations of northern sea nettles (*Chrysaora melanaster*), a scyphozoan jellyfish, in the Gulf of Alaska, including the southern portion of the TMAA (Hunt, 2019). This is the first documented occurrence of *Chrysaora* in the Gulf of Alaska, which is notable because they may present competition for food resources for juvenile salmonids.
- Although the February–March study timeframe of this expedition doesn't overlap with the timing of the proposed activity (April to October), the study does encompass a portion of the TMAA and provides baseline information on salmonid stock presence and relative abundance within deep water offshore habitats similar to those found in the TMAA.

In March 2020, researchers continued their study on the winter ecology of Pacific salmon by returning to the Gulf of Alaska for a second expedition (International Year of the Salmon, 2020). A similar trawl net was used for this study to ensure comparable results across expeditions. The 2020 Expedition was more focused on the southern GOA (outside the TMAA) but found generally higher abundances of salmon than in 2019. Surveys are planned to continue in 2021 throughout the entire North Pacific Ocean (International Year of the Salmon, 2020).

GOA Integrated Ecosystem Research Program – Salmon Studies

Although the 2010–2014 North Pacific Research Board GOA Integrated Ecosystem Research Program was focused on studying GOA groundfish (as further described in Section 3.6.2.9, Essential Fish Habitat), the researchers collected incidental information on salmonids as well. Ecologically important juvenile groundfishes and salmon co-occur in the upper water column of the eastern Gulf of Alaska during the summer, a period when growth is critical to their survival. Daly et al. (2019a) quantified fine-scale spatial and trophic overlap of juvenile groundfishes (arrowtooth flounder [*Atheresthes stomias*], Pacific cod [*Gadus macrocephalus*], walleye pollock [*Gadus chalcogrammus*], and rockfish) and salmon (piscivorous coho and Chinook as well as planktivorous pink, chum, and sockeye) to examine trophic structuring and potential survival bottlenecks for these fishes in the Gulf of Alaska. Fine-scale diet overlap between juvenile groundfishes and planktivorous juvenile salmon species (pink, chum, and sockeye) ranged from 0 percent to 78 percent and was typically higher than that with piscivorous juvenile salmon (coho and Chinook). The researchers did not find a significant resource bottleneck between the species groups regarding availability of zooplankton. Juvenile groundfishes were directly consumed by juvenile salmon and were less frequently caught at stations where the highest catches of juvenile piscivorous salmon occurred. The study suggested that competition for resources by groundfish and salmon was likely when food resources are low in the Gulf of Alaska.

Further, Daly et al. (2019b) studied diet habits of the five Pacific salmon species caught in the marine waters of the eastern and central regions of the Gulf of Alaska. The central Gulf of Alaska region encompassed the shelf portion of the TMAA. The groundfish study incidentally captured over 52,000 juvenile salmon (most [53 percent] were pink) and 10,000 adult salmon (most [80 percent] were chum) and conducted a diet analysis on over 6,500 juvenile and adult salmon (Daly et al., 2019b). Twice as many juvenile and adult salmon (and five times as many juvenile focal groundfish) were caught in the central Gulf of Alaska than the eastern Gulf of Alaska. Focal groundfish include Pacific cod, walleye pollock, arrowtooth flounder, sablefish (*Anoplopoma fimbria*), and Pacific Ocean perch (*Sebastes alutus*). The study found that Chinook and coho salmon primarily consumed fish, cephalopods, euphausiids (adults), and decapods, whereas sockeye, chum, and pink salmon relied on euphausiids, amphipods, pteropods, and copepods (Daly et al., 2019b). The findings suggest that juvenile, immature, and maturing salmon growth and condition can be influenced by bottom-up forces in the ocean, which may ultimately affect run timing and survival rate.

Forage Fishes

Forage fish species in the Gulf of Alaska, such as age-0 walleye pollock, capelin (*Mallotus villosus*), Pacific herring (*Clupea pallasii*), and mesopelagic fishes (e.g., *Myctophidae*), are ecologically important as both consumers of zooplankton, and as prey for fish, seabirds, and marine mammals (McGowan et al., 2019). Bishop (2018) found that herring move from the Gulf of Alaska into Prince William Sound during the fall and spring, suggesting that fish spawning in the Sound migrate out into the Gulf of Alaska. As part of the Gulf of Alaska Integrated Ecosystem Research Program, an acoustic-trawl survey was conducted in the summer and fall of 2011 and 2013 to quantify variability in species composition, density, and distributions of forage fish over the continental shelf and slope in the central and eastern regions of the Gulf of Alaska (McGowan et al., 2019).

The forage fish community in 2011 was characterized by the absence of age-0 pollock and lower densities of capelin, herring, and mesopelagics compared to observations in 2013 (McGowan et al., 2019). Age-0 pollock were abundant across both regions in summer 2013 but were rarely observed in fall. In contrast, summer observations of herring were rare, while aggregations of herring were observed over the eastern GOA shelf in fall of both years. Seasonal changes in community composition are attributed to the transport of age-0 pollock from offshore waters in summer to nearshore waters in fall, and to immigration of herring to the eastern GOA shelf in fall. Eulachon (*Thaleichthys pacificus*) and Pacific sand lance (*Ammodytes personatus* are important forage fishes in the North Pacific Ocean but are more common outside of the TMAA. Sand lance typically occur in shallow, coastal, and intertidal waters (< 50 m depth) (McGowan et al., 2019). Spatial and temporal variability in community composition and distributions of forage fish species may potentially impact predator foraging in the Gulf of Alaska.

3.6.2.8 Green Sturgeon (*Acipenser medirostris*)

3.6.2.8.1 Status and Management

The Southern DPS of green sturgeon was listed as threatened under the ESA in 2006 (71 FR 17757). Critical habitat was designated for this DPS in 2009, but it does not include the Gulf of Alaska (74 FR 52300). In 2003, NMFS determined that green sturgeon along the West Coast consist of two DPSs: (1) a northern DPS comprising populations in coastal watersheds northward of and including the Eel River ("Northern DPS"); and (2) a southern DPS consisting of coastal and Central Valley populations south of the Eel River, with the only known population occurring in the Sacramento River ("Southern DPS") (71 FR 17757). Only the Southern DPS is ESA-listed. The Northern DPS was found to be "not warranted" for listing and remains a federal Species of Concern.

3.6.2.8.2 Distribution

Green sturgeon are long-lived, slow-growing fish and the most marine-oriented of the sturgeon species. They range along the Pacific coast from Baja California to the Aleutian Islands. Green sturgeon spend the majority of their lives in nearshore oceanic waters, bays, and estuaries. They are anadromous, with adults returning to freshwater to spawn. In marine waters, adults and subadults primarily occur at depths of 40–110 m (Erickson & Hightower, 2007), with most found at depths of 20–80 m (Payne et al., 2015a). They are rarely found deeper than 200 m (Huff et al., 2012). Only a small portion (15 percent) of the TMAA overlaps with shelf areas shallower than the 200 m isobath.

Green sturgeon have been occasionally observed in coastal, nearshore, and estuarine habitats from southeast Alaska through the Gulf of Alaska to the northwest side of Unalaska Island in the Aleutian Chain (Environmental Protection Information Center et al., 2001). Southern DPS fish are confirmed to occur from Graves Harbor, Alaska, to Monterey Bay, California (73 FR 52300). Green sturgeon observed northwest of Graves Harbor have not been identified to DPS. Two tagged Southern DPS green sturgeon were detected at the hydroacoustic monitor in Graves Harbor, indicating that Southern DPS green sturgeon do migrate further north than the 58th parallel, which transects the northern portion of the TMAA.

To evaluate green sturgeon marine migration patterns, researchers tagged 213 subadult and adult sturgeon along the U.S. West Coast (Lindley et al., 2008). Green sturgeon exhibited an annual migration along the continental shelf from U.S. to Canadian waters in the fall and an apparent return migration in the spring. Large numbers of green sturgeon were detected on northwest Vancouver Island, British Columbia, during May to June and October to November. However, only a single fish was detected in southeast Alaska in December, suggesting that use of the Bering Sea and Aleutian archipelago is uncommon for North American green sturgeon.

In 2019, the Northwest Fisheries Science Center conducted a study to characterize the distribution of salmonids within the Northwest Training and Testing area along the coast of Washington state (Smith &

Huff, 2020). The study was conducted in support of the U.S. Navy's 2019 Annual Marine Species Monitoring Report for the Pacific. The study deployed 107 stationary acoustic receivers in a grid pattern along the coast to detect tagged fish. In addition to gaining information on salmonid distribution, the study managed to detect 124 tagged green sturgeon, with sturgeon detected at most of the receiver locations. The study found that green sturgeon were highly distributed along the coastline (between 3 NM and 12 NM offshore) between May and September 2019. Nearly all green sturgeon were found nearshore of the 200 m depth contour when migrating off the coast of Washington, which is consistent with other studies indicating their preference for nearshore habitats (Smith & Huff, 2020).

Historical fisheries records of Alaskan groundfish catches dating back to the 1960s and fisheries observer records from 1986 to 2006 did not contain any records of green sturgeon, and few records have been reported in other databases from these waters (Huff et al., 2012). In 2006, Colway and Stevenson (2007) noted the presence of two unidentified green sturgeon specimens in the Bering Sea and the western Gulf of Alaska. Since then, fishery observers in the Bering Sea have encountered four additional green sturgeon specimens, including three in the past two years (Stevenson & Hunt, 2020). It is unclear whether these fish were part of the Northern or Southern DPS. In contrast, green sturgeon have been regularly captured in groundfish bottom trawls off Oregon and Washington (Erickson & Hightower, 2007).

In marine habitats, green sturgeon regularly occur over flat, sandy substrate (Payne et al., 2015a) but can also be found near complex hard-bottom areas (Huff et al., 2012). An Oregon coastal study found that green sturgeon, on average, spent a longer duration in areas with high seafloor complexity, especially where a greater proportion of the substrate consists of boulders (Hinckley et al., 2019). Sturgeon in this study may have been using complex seafloor habitat because it coincides with the distribution of benthic prey taxa or provides refuge from predators. Huff et al. (2012) found that sturgeon overwintering aggregations are sheltered in rocky, high-relief areas less than 200 m deep and are associated with ephemeral, yet abundant, standing stocks of plankton that support rich benthic communities.

The Gulf of Alaska shelf and continental slope consist of complex bathymetric features, including gulleys and canyons, rocky nearshore habitat, elevated pinnacles, flat muddy areas, and channels with high current flow (Baker et al., 2019). The shelf is dominated by gravel, sand, silt, and mud, punctuated by areas of hard rock. There are numerous banks and reefs with coarse, rocky bottoms, but much of the shelf is covered by glacial silt from the Copper River and the Bering and Malaspina glaciers (Mundy, 2005). Although sand and silt substrate in the TMAA may be used by green sturgeon, it may not provide preferred habitat to support high quality foraging and predator avoidance. Baker et al. (2019) modelled Gulf of Alaska trawlable areas using benthic terrain and oceanographic variables. The researchers found higher rugose substrates along the southern extent of the Kenai Peninsula and the southern coastline of Kodiak Island (within the shelf portion of the TMAA), which may provide more suitable green sturgeon habitat.

Although Gulf of Alaska trawling and observer data indicate few documented green sturgeon, these fishing activities tend to be performed over flat/sandy habitats to minimize gear damage. Further, green sturgeon don't tend to consume bait as easily as white sturgeon, and are best targeted using on-bottom gill nets (which are not typically deployed in the Gulf of Alaska), which may all contribute to the lack of green sturgeon observations. It is possible that green sturgeon are selectively using more rugose habitat within the TMAA within untrawlable areas so they are not detected by research surveys or as groundfish bycatch. Green sturgeon may also migrate through the Gulf of Alaska to access Alaska Peninsula and

Bering Sea habitats. Since green sturgeon have been documented as far north as Graves Harbor (in the eastern Gulf of Alaska) (73 FR 52300), it is possible that Southern DPS fish could be present in the Gulf of Alaska and the onshelf portion of the TMAA. However, it is more likely that any green sturgeon in the Gulf of Alaska originate from the non-listed Northern DPS fish.

Cold temperatures, perhaps in combination with other factors related to the danger of dispersing far from spawning grounds, may be another reason why green sturgeon are rare visitors north of 54°N latitude (Huff et al., 2012). Although there is a chance that green sturgeon may be seasonally present (fall/winter) in shallower, more rugose portions of the Gulf of Alaska continental shelf (<200 m deep), these areas represent a very small portion of the TMAA (Huff et al., 2020). Thus, the probability that listed Southern DPS green sturgeon would be present in the TMAA is very low, particularly during periods of the year when training activities are proposed. Further investigations are needed to determine presence, distribution, and habitat preferences of Southern DPS fish in the Gulf of Alaska.

3.6.2.9 Essential Fish Habitat

The Magnuson-Stevens Fishery Conservation and Management Act requires that the regional Fishery Management Councils, in cooperation with NMFS, delineate EFH for all federally managed fisheries. The NPFMC has two FMPs in effect for fishes, groundfish and salmon fisheries in the Gulf of Alaska. Information on invertebrate fisheries, such as the scallop fishery in the TMAA, are presented in Section 3.6.2.9 (Essential Fish Habitat). EFH descriptions were presented in the 2011 GOA Final EIS/OEIS and updated in the 2016 GOA Final SEIS/OEIS. This SEIS/OEIS addresses the same activities within the TMAA as was discussed in the previous documents.

The NPFMC and NMFS are required to review the EFH components within each FMP every five years. The 2015 EFH five-year Review was completed in 2017 and implemented in 2018, and hereafter referred to as the “2017 Review.” The next five-year review is scheduled for 2022. Based on the 2017 Review, new habitat and life history information was used to revise the Groundfish and Salmon EFH descriptions and maps in the FMPs. Although the Groundfish and Salmon FMPs have been updated since the 2016 GOA Final SEIS/OEIS was issued, the analyses previously presented remains valid. However, updates to each FMP are summarized below, by species group. Designated EFH for each life stage that occurs within the TMAA are provided in Table 3.6-4 and Table 3.6-5.

Table 3.6-4: Groundfish Species with EFH Designated in the TMAA

Fishery Management Plan	Species	Eggs	Larvae	Early Juvenile	Late Juvenile	Adult
Groundfish	Alaska plaice	X	X	-	X	X
	Arrowtooth flounder	X	X	-	X	X
	Atka mackerel	X	-	-	X	X
	Dover sole	X	X	-	X	X
	Dusky rockfish	X	X	-	X	X
	Flathead sole	X	X	-	X	X
	Northern rockfish	X	X	X	X	X
	Octopus	--	-	-	-	X
	Other rockfish	X	-	-	X	X
	Pacific cod	-	X	X	X	X
	Pacific ocean perch	X	X	X	X	X
	Rex sole	X	X	-	X	X
	Rock sole (Northern/Southern)	X	X	-	X	X
	Rougheye/Blackspotted rockfish	X	X	X	X	X
	Sablefish	-	X	X	X	X
	Sculpins	-	-	-	-	X
	Sharks	-	-	-	-	-
	Shortraker rockfish	X	X	X	X	X
	Skates	X	-	X	X	X
	Squid	-	-		X	X
	Shortspine thornyhead rockfish	-	-	X	X	X
	Walleye pollock	X	X	X	X	X
	Yelloweye rockfish	X	X	X	X	X
	Yellowfin sole	X	X	-	X	X

Sources: (North Pacific Fishery Management Council, 2014, 2019)

Table 3.6-5: Salmon Species with EFH Designated in the TMAA

Fishery Management Plan	Species	Eggs and Larvae	Freshwater Juveniles	Estuarine Juveniles	Marine Juveniles	Marine Immature/ Maturing Adults	Freshwater Adults
Salmon	Chinook	-	-	-	X	X	-
	Chum	-	-	-	X	X	-
	Coho	-	-	-	X	X	-
	Pink	-	-	-	X	X	-
	Sockeye	-	-	-	X	X	-

Source: (North Pacific Fishery Management Council et al., 2018)

3.6.2.9.1 Groundfish Fishery Management Plan

The Fishery Management Plan for Groundfish of the Gulf of Alaska was originally described in the 2011 GOA Final EIS/OEIS, and updates were provided in the 2016 GOA Final SEIS/OEIS. In August 2019, the North Pacific Fishery Management Council (2019) published an updated Groundfish FMP for the Gulf of

Alaska. This Groundfish FMP describes additional amendments that have been implemented since the 2016 GOA Final SEIS/OEIS was prepared:

- Amendment 103 was implemented on September 12, 2016. This amendment allows NMFS to reapportion unused Chinook prohibited species catch within and among specific trawl sectors.
- Amendment 104 was implemented on September 7, 2017, and authorizes NMFS to place electronic monitoring systems for collecting at-sea data on vessels.
- Amendment 105 was implemented on July 5, 2018 and included the following components:
    - Revised EFH description and identification by species; and updated life history, distribution, and habitat association information.
    - Updated the model used to determine fishing effects on EFH and the description of EFH impacts from fishing activities.
    - Updated description of EFH impacts from non-fishing activities and EFH conservation recommendations for non-fishing activities.
- Amendment 106 was implemented on August 6, 2018 and prohibited directed fishing for the squid species complex (squids) by federally permitted groundfish fishermen, moved squid to the Ecosystem Component category, and specified a squid retention limit in the GOA groundfish fisheries.

Since the 2019 FMP was published, NMFS has proposed or implemented two additional amendments to the GOA Groundfish FMP:

- Amendment 107 was implemented on February 20, 2020 (85 FR 9687). This amendment requires that the operator of a federally permitted catcher vessel using hook-and-line, pot, or jig gear in the Gulf of Alaska retain and land all rockfish species caught while fishing for groundfish or Pacific halibut.
- Amendment 108 was implemented on December 20, 2019 (84 FR 70064). This amendment prohibits replaced Amendment 80 catcher/processer vessels from receiving and processing Pacific cod harvested and delivered by catcher vessels directed fishing for Pacific cod in the Gulf of Alaska.
- Amendment 109 was proposed by NMFS on February 28, 2020 (85 FR 11939). This amendment would reduce operational and management inefficiencies in the GOA Pacific cod fisheries by changing seasonal Pacific cod apportionments to allow greater harvest opportunities earlier in the year.
- Amendment 110 was proposed by NMFS on March 23, 2020 (85 FR 16310). This amendment would reclassify sculpins into the non-target ecosystem component category, prohibit directed fisheries for sculpins, and limit the retention and commercial exchange of sculpins.

Since these amendments were proposed or implemented to help facilitate a sustainable groundfish fishery did not significantly change the environmental baseline, the analyses presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remain valid.

Groundfish Species

Groundfish species in the Gulf of Alaska include flatfishes, rockfishes, other roundfishes, skates, sharks, and chimeras. In 2017, as part of an ongoing sampling series, NMFS conducted a bottom trawl survey in the Gulf of Alaska, which overlapped with portions of the TMAA (Von Szalay & Raring, 2018). The survey captured a total of 161 fish and 364 invertebrate species during summer 2017. Species with the highest total catch abundance (by weight) over the entire survey area were: Pacific ocean perch, arrowtooth flounder, walleye pollock, Pacific halibut (*Hippoglossus stenolepis*), flathead sole (*Hippoglossoides elassodon*), giant grenadier (*Coryphaenoides pectoralis*), northern rockfish (*Sebastes polyspinis*), and sablefish.

From 2010 to 2014, the North Pacific Research Board led a large multidisciplinary ecological study that examined the physical and biological mechanisms that determine survival of juvenile groundfishes in the Gulf of Alaska (North Pacific Research Board, 2020). Researchers studied the "gauntlet" faced by groundfishes (specifically walleye pollock, Pacific cod, Pacific Ocean perch, sablefish, and arrowtooth flounder) during their first year of life as these fish are transported from offshore areas where they are spawned to nearshore nursery areas. The studies were structured around two GOA study regions (eastern and western) primarily on the continental shelf, ranging from Chatham Strait to Kodiak Island, with the dividing boundary at Prince William Sound. The western portion of the study area encompasses the continental shelf of the TMAA, and the eastern study area is mostly shoreward of the TMAA. The continental shelf within the TMAA is broad, with high demersal fish biomass but low species diversity (North Pacific Research Board, 2020). The following summarizes some of the pertinent findings from the Gulf of Alaska Integrated Ecosystem Research Program.

Siddon et al. (2019) found that Pacific cod and walleye pollock larvae were more abundant over the shelf and slope in the western Gulf of Alaska, reflecting preferred habitat for spawning adults and settling juvenile fish. Conversely, sablefish larvae were more abundant near areas of deeper water in the eastern Gulf of Alaska, such as Yakutat Canyon. Rockfish larvae (predominantly persistent organic pollutants in spring) were ubiquitous across the region. Larval Pacific ocean perch were associated with the slope, troughs, and canyons intersecting the slope, and the outer shelf. Arrowtooth flounder larval abundances varied by region and year. These larvae were collected primarily along the slope and near canyons and troughs. Deep-water features such as troughs and canyons that bisect the shelf appear to be "hot spots" for rockfish, sablefish, and arrowtooth flounder larvae originating from slope or basin spawning habitat.

Goldstein et al. (2019) found that the large number of groundfish species spawning in late winter and spring leads to high spring diversity and distinct eastern versus western larval fish assemblages. Most larval assemblages were found on the shelf in the west portion of the TMAA during spring, with fewer assemblages detected in summer. Assemblage patterns were largely driven by regional spring spawning events and only minimally influenced by temperature, salinity, and bottom depth over the shelf. For most groundfish species, the emergence of larvae is timed to coincide with the spring phytoplankton bloom, with early-phenology species associated with deep water and slope habitats and late-phenology species occupying primarily coastal and shelf habitats. Exceptions are the deep-water larvae of rex sole (*Glyptocephalus zachirus*), dover sole (*Solea solea*), and rockfishes that are most abundant in spring through summer. Species that are most abundant in winter (e.g., arrowtooth flounder and Pacific halibut) are spawned in deep water and the primary larval habitat is over the slope. These fishes appear to have adapted to the temporal and spatial complexity of the Gulf of Alaska ecosystem (Doyle et al., 2019).

For more specific information on groundfish status, distribution, and ecology in the Gulf of Alaska, please refer to the following publications:

- Arrowtooth flounder (Debenham et al., 2019; Doyle et al., 2018; Stockhausen et al., 2019)
- Sablefish (Gibson et al., 2019)
- Pacific cod (Hinckley et al., 2019)

- Walleye pollock (Parada et al., 2015)
- Pacific Ocean perch (National Oceanic and Atmospheric Administration, 2020a)
- All groundfish (Pirtle et al., 2017)

Information on groundfish harvest and harvest updates are presented in Section 3.11.1.1.2 (Commercial and Recreational Fishing). Although Pacific halibut are an important component to fisheries within the Study Area, this species is not managed under the groundfish fishery management plan.

3.6.2.9.2 Salmon Fishery Management Plan

The FMP for the Salmon Fisheries in the EEZ off Alaska was originally described in the 2011 GOA Final EIS/OEIS, and updates were provided in the 2016 GOA Final SEIS/OEIS. In October 2018, the North Pacific Fishery Management Council (2018b) published an updated Salmon FMP. This Salmon FMP describes one additional amendment that has been implemented since the 2016 GOA Final SEIS/OEIS was prepared:

- Amendment 13 was approved on July 5, 2018 (83 FR 31340). Based on the salmon five-year EFH review, Amendment 13 updated the description of EFH for all five species of Pacific salmon, replaced the maps of marine EFH for all five species of Pacific salmon, and updated the analysis of fishing and non-fishing impacts on salmon habitat in areas that are considered salmon EFH. The updated EFH descriptions reduced the area of designated EFH for Pacific salmon by 71.3 percent on average (Echave et al., 2012).

Salmon EFH is still currently designated within the Gulf of Alaska, including the TMAA, but does not extend out to the limits of the U.S. EEZ (Echave et al., 2012). Juvenile salmon EFH generally consist of the water over the continental shelf within the Bering Sea extending north to the Chukchi Sea, and over the continental shelf throughout the Gulf of Alaska and within the inside waters of the Alexander Archipelago. Within the TMAA, the continental shelf ranges from 150 to 200 m. Immature and mature Pacific salmon EFH include nearshore and oceanic waters, often extending well beyond the shelf break, with fewer areas within the inside waters of the Alexander Archipelago and Prince William Sound (National Marine Fisheries Service, 2018b).

Since this amendment just refined the geographic scope of EFH for Pacific salmon in marine waters off Alaska and did not propose any new restrictions on the habitat or the species, the analyses presented in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS remains valid.

3.6.3 Environmental Consequences

As presented in Section 1.3 (Proposed Action), there are no changes to the current Proposed Action from that presented in the 2016 GOA Final SEIS/OEIS. This SEIS/OEIS analyzes the impacts on fish under two alternatives, the No Action Alternative and Alternative 1 (the Proposed Action).

This section presents changes since the 2016 GOA Final SEIS/OEIS and evaluates how and to what degree the activities described in Proposed Action could impact fish in the TMAA. The stressors analyzed for impacts on fish in the TMAA in included the following:

- Acoustic Stressors (sonar and other transducers, vessel noise, aircraft noise, weapons noise)
- Explosive Stressors

3.6.3.1 Acoustic Stressors

The analysis of effects to fishes follows the concepts outlined in Section 3.0.4.3 (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 3.6.3.1.1 (Background). This is followed by an analysis of estimated impacts on fishes due to sonar and other transducers. Additional explanations of the acoustic terms and sound energy concepts used in this section are found in Appendix B (Acoustic and Explosive Concepts).

The Navy will rely on the previous 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS analysis of vessel, aircraft, 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 3.6.3.1.2 (Impacts from Sonar and Other Transducers) of this SEIS/OEIS supplants the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS for fishes, and may change estimated impacts for some species since the 2016 GOA Final SEIS/OEIS. In addition, this analysis includes the consideration of ESA-listed green sturgeon not previously analyzed.

3.6.3.1.1 Background

Effects of human-generated sound on fishes have been examined and summarized in numerous publications (de Jong et al., 2020; 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 & Hawkins, 2019; Popper, 2003, 2008; Popper et al., 2016; Popper & Hastings, 2009b; Popper & Hawkins, 2018; Popper et al., 2014). The potential impacts from Navy activities are based on the analysis of available literature related to each type of effect. Where applicable, interim criteria and 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 B (Acoustic and Explosive Concepts).

Although air guns and pile driving are not used during GOA training activities, the analysis of some explosive impacts (Section 3.6.3.2, Explosive Stressors) will in part rely on data from fishes exposed to impulsive sources where appropriate. Therefore, background information on impulsive sources are provided below.

3.6.3.1.1.1 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 Sources) and lacks the amplitude and energy to cause any direct injury. Section 3.0.4.3 (Conceptual Framework for Assessing

Effects from Acoustic and Explosive Activities) provides additional information on injury and the framework used to analyze this potential impact.

Injury due to 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 as part of this proposed action, 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 (0.1–6 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., 2013a; Casper et al., 2012b; Casper et al., 2013b; Halvorsen et al., 2012a; Halvorsen et al., 2011, 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. saxatilis*) 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 measured 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 reported 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 fulyescens*), 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, special tissues 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) 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., 2013a; Casper et al., 2012b; Casper et al., 2013b; Halvorsen et al., 2012a; Halvorsen et al., 2011, 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 Guideline* 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 morhura*), saithe (*Pollachius virens*), and spotted wolffish

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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 where impulsive sound has near-instantaneous rise to a high peak pressure. For more detail, see Appendix B (Acoustic and Explosive Concepts).

(*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 centimeters, respectively, which lies within the range of frequencies used during sound exposures and therefore may explain some of the noted mortalities.

Past research has demonstrated that fish species, size, and depth influences the resonant frequency of the swim bladder (Løvik & Hovem, 1979; McCartney & Stubbs, 1971). For example, 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); higher frequencies, greater than 1 kHz, could produce swim bladder resonance in smaller fishes. At resonance, the swim bladder may absorb much of the acoustic energy in the impinging sound wave. It was hypothesized that the resulting oscillations may cause mortality, or harm the auditory organs or the swim bladder (Jørgensen et al., 2005; Kvadsheim & Sevaldsen, 2005). However, damage to the swim bladder and to tissues surrounding the swim bladder was not observed in fishes exposed to multiple sonar pulses from approximately 165–195 dB re 1 µPa at their presumed swim bladder resonant frequency (Jørgensen et al., 2005). Fishes may be more susceptible to injury from swim bladder resonance when exposed to continuous signals within the resonant frequency range; although, based on the above studies, injury or mortality from swim bladder resonance under real-world conditions is unlikely.

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,* also known as steelhead), 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.

3.6.3.1.1.2 Hearing Loss

Researchers have examined the effects on hearing in fishes from sonar-like signals, tones, and different impulsive noise sources. Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on hearing loss and the framework used to analyze this potential impact.

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 exposure (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. 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. For example, if 5 dB of PTS occurs at a certain frequency, then a sound at that same frequency would need to be 5 dB louder for the animal to detect it. 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., 2014; Popper et al., 2005; Smith et al., 2006). 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 (five 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 20–25 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 (*Pargus auratus*), a species that has a swim bladder that is not involved in hearing, were exposed to multiple air gun shots for up to one and one-half 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 (*Lutjanus 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., 2014; Popper et al., 2005; 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- or other high-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., Breitzler et al., 2020; Scholik & Yan, 2002a; Smith et al., 2006; Smith et al., 2004a). Smith et al. (2006; 2004a) 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 levels (Smith et al., 2004a). 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, 2002b). 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 species 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 (e.g., 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 fish's 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 sound pressure levels of 130 dB and 140 dB re 1 µPa recovered within three days, whereas it took up to 14 days for fish exposed to the highest sound pressure level (150 dB re 1 µPa) 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 significant hair cell loss, whereas lower exposure levels did not. Similar to the other effects measured in this study, hair cell loss attributed to the highest exposure level 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 (lacking evidence of PTS).

Butler et al. (2020) presented playbacks of pure tones ranging from 100 to 2,000 Hz to African cichlids (*Astatotilapia burtoni*), a freshwater species with a swim bladder involved in hearing, stationed in a small aquarium to investigate the effects on hearing. Playbacks were presented at a sound pressure level of 140 dB re 1 µPa for three hours. After review of the playback, the authors noted that the sound source was more broadband than intended and therefore may not be analogous to other tonal sources (such as sonar), but rather could be more comparable to vessel noise playbacks or an example of elevated background levels. Observed threshold shifts were only significantly different than controls in lower frequencies (200 and 300 Hz), which corresponds to the species' best range of sensitivity. Recovery of hearing thresholds was not measured during this study.

When reviewing results from the above 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 subjected 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 noise produced by a vessel passing by in their natural environment, even in areas with high levels of vessel movement, would only be exposed for short durations (e.g., seconds or minutes) and therefore relatively low sound exposure levels as vessels pass by. As evidence suggests that fish can recover from hearing loss (both threshold sensitivity and actual physical damage) even after long duration exposures in a confined space, it also indicates similar results to lower level and shorter duration exposures. Therefore, overall effects would not likely rise to the level of impact demonstrated in the summarized laboratory studies.

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 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.

3.6.3.1.1.3 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 result in a reduction in communication and listening space, effectively limiting the distance over which an animal can communicate and detect biologically relevant sounds (Pine et al., 2020). Human-generated continuous sounds (e.g., some sonar, vessel noise, and vibratory

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

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 3.0.4.3 (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 three freshwater fishes: the goldfish and the lined Raphael catfish (*Platydoras costatus*), fishes with notable hearing specializations for sound pressure detection; and the pumpkinseed sunfish (*Lepomis gibbosus*), a freshwater fish without notable specializations. 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., 2009; Ramcharitar et al., 2006; Ramcharitar et al., 2001; Stanley et al., 2017), including vocalizations made during reproductive phases or sounds emitted from a reef for navigating larvae (de Jong et al., 2020; 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 it could 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 as a result 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 decrease with increasing distance (Popper et al., 2014).

3.6.3.1.1.4 Physiological Stress

Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on physiological stress and the framework used to analyze this potential impact. 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. Increases in background sound have been shown to cause stress in humans and animals, which also includes the measurement of biochemical responses by fishes to acoustic stress (e.g., Goetz et al., 2015; Madaro et al., 2015; Remage-Healey et al., 2006; Smith et al., 2004b; Wysocki et al., 2007; Wysocki et al., 2006). However, results from these studies 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 includes 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 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).

Mills et al. (2020) observed the hormonal effects of motorboat noise on orange-fin anemonefish (*Amphiprion chrysopterus*) over short-term (30 minutes) and longer-term (48 hours) periods. Cortisol levels did not differ significantly between the periods for either sex. Testosterone levels were significantly higher in males exposed to motorboat-noise playback and 11-ketotestosterone (11-KT) levels were significantly higher in males during the short-term experiment and in both sexes during the longer-term experiment.

Kusku et al. (2020) measured respiratory changes as secondary indicators of stress in Nile tilapia (*Oreochromis niloticus*) to determine potential effects of long-term exposure to underwater sound playback, including shipping noise. Fish exposed to noise showed as much as a two-fold increase in respiratory indicators (opercular beat rate and pectoral wing rate) after 10 minutes of sound exposure as compared to controls and pre-exposure rates. Over the next 120 days of continuous sound exposure, respiratory indicators declined steadily and returned to baseline. The authors conclude that the data support habituation of fish to chronic noise exposure.

However, not all species show these reactions. Smith et al. (2004b) 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. In addition, although there was a difference of 10 dB in overall background level and boat activity between test sites, reef fish, *Halichoeres bivittatus,* showed similar levels of whole-body cortisol (Staaterman et al., 2020). This suggests that boat noise, in this context, was not as stressful as handling of the fish for this particular experiment and contradicts previous conclusions that follow similar study designs.

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.

3.6.3.1.1.5 Behavioral Reactions

Section 3.0.4.3 (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 to 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 either do not respond, learn to tolerate or habituate to repeated exposures, or learn to tolerate noise that does not seem threatening (e.g., Bruintjes et al., 2016; Currie et al., 2020; Hubert 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 vary depending on the type of exposure and sound source present. Changes in sound intensity may be more important to a fish’s 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 TMAA. 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 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 (*Pagrus auratus*) 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 6 hours), resulting in single pulse sound exposure levels of 125 to 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 (*Clupea 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) (Slotte et al., 2004).

Bruce et al. (2018) investigated the potential behavioral effects of sharks exposed to a seismic survey. In the first part of the study, researchers attached acoustic and accelerometer tags to swell sharks (*Cephaloscyllium laticeps)*, gummy sharks (*Mustelus antarcticus)*, and tiger flathead (*Neoplatycephalus richardsoni*) in order to monitor their behavior during seismic surveys. Although tagging was successful and provided a large sample size for two out of the three species, most tagged individuals moved out of range of the experimental site where autonomous acoustic receivers were placed or sporadically returned to the monitoring site throughout the duration of the survey. This made it difficult to correlate displacement from the area with the actual survey. In the second part of the study, modeled predicted catch rates within the experimental site were compared to actual catch per unit effort data collected from local fisheries. Of the nine species analyzed, only three of them showed reductions in catch rates following the seismic survey. Contrary to past findings and assumptions, catch rates for six species actually increased after the survey. Although these findings are interesting and, in some ways, may contradict prior conclusions, there are some improvements that should be made to similar studies in the future to better understand the true effects of seismic surveys on fish behavior and catch rates.

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 often-times 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. (2014; 2012) 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 $\mu Pa^2 \cdot s$. 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-–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 marine fishes can detect sonars above 1 kHz (see Section 3.6.2.1.3, 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 fish ahead of a ship that showed avoidance reactions did so at ranges of 50 to 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 depending 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 sound source (e.g., type of vessel or motor vs. playback of broadband sounds), 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 observed responses include increased group cohesion; increased distractions or evidence of modified attention; changes in vertical distribution in the water column, swim speeds, distance traveled, and feeding efficacy such as reduced foraging/hunting attempts and increased mistakes (i.e., lowered discrimination between food and non-food items) (e.g., Bracciali et al., 2012; De Robertis & Handegard, 2013; Gendron et al., 2020; Handegard et al., 2015; Jimenez et al., 2020; Mauro et al., 2020; McCormick et al., 2019; Nedelec et al., 2017a; Nedelec et al., 2015; Neo et al., 2015; Payne et al., 2015b; 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). Furthermore, both playbacks and actual noise from nearby boats have resulted in alterations in reproductive and nesting behaviors, such as changes in visual displays; signaling and aggression towards potential mates, competitors, and conspecifics; diminished territorial interactions; and reduced parental care behaviors such as egg fanning and vigilance (Butler & Maruska, 2020; McCloskey et al., 2020). In addition to physiological stress responses discussed in Section 3.6.3.1.1.4, Mills et al. (2020) observed the behavioral effects of motorboat noise on orange-fin anemonefish (*Amphiprion chrysopterus*) over short-term (30 minutes) and longer-term (48 hours) periods. Significant behavioral effects included increased hiding, reduction in distance from anemone, and increased aggressive behavior toward heterospecifics over both time periods.

Behavioral responses may also be dependent on the type of vessel to which a fish is exposed. 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 response, 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). Furthermore, juvenile Ambon damselfish showed a reduction in learned anti-predator behaviors likely as a result of distraction which could lead to an increased risk of survival (Ferrari et al., 2018). 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 displayed 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) as well as no effect of boat noise or presence on round goby (*Neogobius melanostomus)* calling behaviors (Higgs & Humphrey, 2019). Mensinger et al. (2018) found that Australian snapper (*Pagrus auratus*) 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.

To investigate potential avoidance on a larger scale, Ivanova et al. (2020) tagged Arctic cod and recorded movement and behavior during exposure to noise produced by cargo and cruise ship traffic. Overall, cod increased their horizontal movement outside of their estimated home range when vessels were either present or moving, compared to periods where vessels were absent indicating periods of potential avoidance. In addition, changes in feeding, travel, and search behaviors were observed when comparing each sound condition. The authors note that future studies should continue to investigate whether these observed effects are prolonged or how quickly fish may return to their home range and baseline behaviors.

Most fish species should be able to detect vessel noise due to its low-frequency content and their hearing capabilities (see Section 3.6.2.1.3, Hearing and Vocalization). The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) suggests that fishes have a high to moderate 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).

3.6.3.1.1.6 Long-Term Consequences

Section 3.0.4.3 (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 and 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.

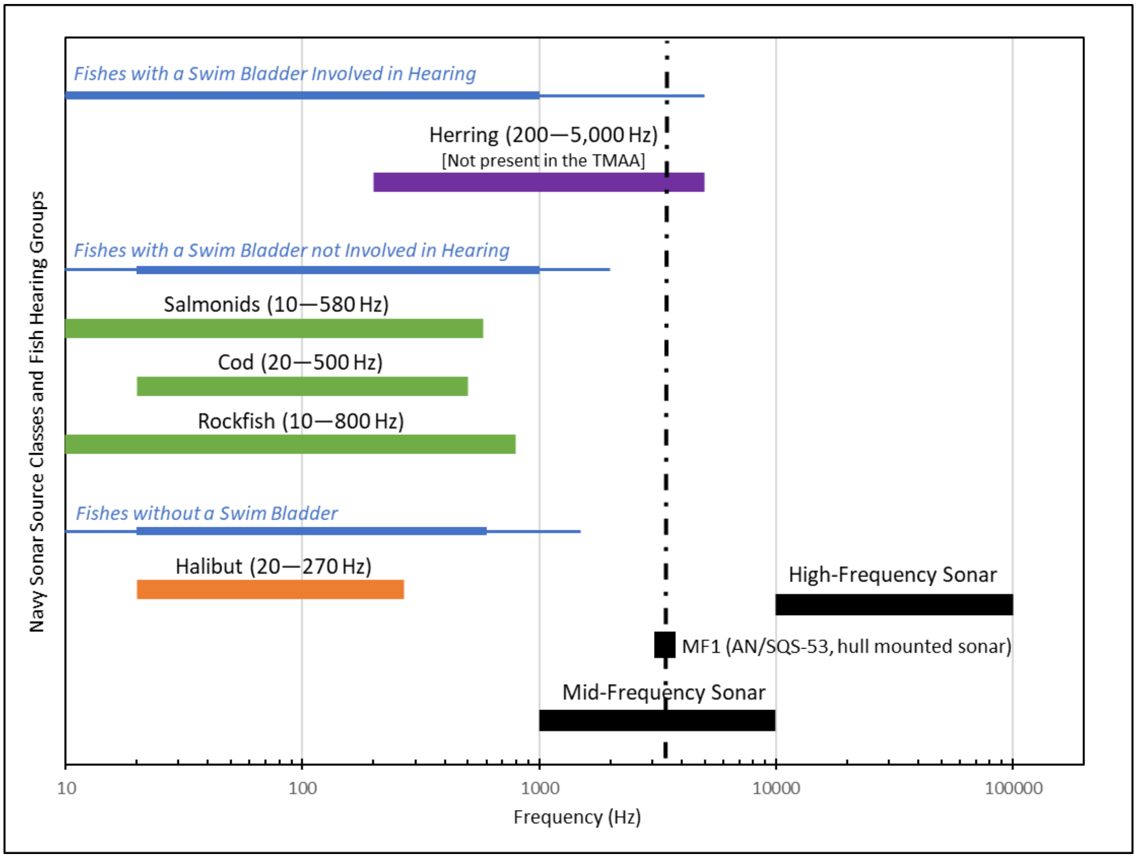
3.6.3.1.2 Impacts from Sonar and Other Transducers

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

As described under Section 3.6.3.1.1.1 (Injury), direct injury from sonar and other transducers is highly unlikely because injury has not been documented in fish exposed to sonar (Halvorsen et al., 2013; Halvorsen et al., 2012c; 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 3.6.2.1.3 (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 with, or in close proximity to, the inner ear).

Figure 3.6-2 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 potential overall range of frequency detection for each hearing group.



Notes: Thin blue lines represent the estimated minimum and maximum range of frequency detection for the hearing group. All hearing groups are assumed to detect frequencies down to 10 Hz regardless of available data. Thicker portions of each blue line represent the estimated minimum and maximum range of best sensitivity for that group. Thick colored lines (purple, green, orange) below each hearing group represent example hearing data for specific species. Not all fishes within a hearing group would be able to detect all frequencies. For example, flatfish such as halibut can only detect frequencies up to 270 Hz, although other fishes in the same hearing group can detect much higher frequencies (e.g., bull sharks [not present in the TMAA] can detect up to 1,500 Hz, the upper limit of the hearing group). Each sonar source class that occurs in the TMAA is represented graphically by the horizontal black bars. Not all sources within each class would operate at all the displayed frequencies and may not overlap all fish hearing groups as demonstrated by the dotted black line. Hz = hertz, MF1 = 3,500 Hz.

Sources: (Casper & Mann, 2006; Chapman & Hawkins, 1973; Chapman & Sand, 1974; Hawkins & Johnstone, 1978; Mann et al., 2005; Popper, 2008; Popper et al., 2007; Tavolga & Wodinsky, 1963)

Figure 3.6-2: Fish Hearing Group and Navy Sonar Bin Frequency Ranges

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 most sensitive to lower frequencies, well below 1 kHz, the bull shark (a species not known to occur in the TMAA) has been tested and can detect frequencies up to 1.5 kHz (Kritzler & Wood, 1961; Myrberg, 2001), representing 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 3.6-2 is not a composite audiogram but rather displays the basic overlap in potential frequency content for each hearing group with Navy defined sonar classes (i.e., mid- and high-frequency) as discussed under Section 3.0.4.1.1 (Sonar and Other Transducers).

Systems within the low-frequency sonar class present the greatest potential for overlap with fish hearing, although these sonars are not used as part of the proposed action. Some mid-frequency sonars and other transducers may also overlap some species' hearing ranges, but to a much lesser extent than low-frequency sonars. For example, the only hearing groups that have the potential 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 marine fishes would not hear, or be affected by, most 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, although none of these species (subfamily Alosinae [menhaden, shad]) are known to be present in the TMAA.

The most probable impacts from exposure to sonar and other transducers are TTS (for more detail see Section 3.6.3.1.1.2, Hearing Loss), masking (for more detail see Section 3.6.3.1.1.3, Masking), physiological stress (for more detail see Section 3.6.3.1.1.4, Physiological Stress), and behavioral reactions (for more detail see Section 3.6.3.1.1.5, Behavioral Reactions). Analysis of these effects are provided below.

3.6.3.1.2.1 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 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 TMAA 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 3.6-6. 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., 2013; Halvorsen et al., 2012c; Kane et al., 2010; Popper et al., 2007). General research findings from these studies can be reviewed in Section 3.6.3.1.1.2 (Hearing Loss).

Table 3.6-6: Sound Exposure Criteria for TTS from Mid-Frequency Sonar

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

Criteria for high- and very-high-frequency sonar were not presented 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 in the subfamily Alosinae could potentially be affected. As stated previously, these fish species are not present in the TMAA.

3.6.3.1.2.2 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 3.6-7 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 (Acoustic Sources).

Table 3.6-7: Ranges to Temporary Threshold Shift from Three Representative Sonar Bins

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

Notes: (1) 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.
(2) MF = mid-frequency, NR = no criteria are available and therefore no range to effects are estimated.

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

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

3.6.3.1.2.4 Impacts from Sonar and Other Transducers Under Alternative 1

The Navy's Proposed Action is consistent with the Proposed Action presented in the 2011 GOA Final EIS/OEIS (U.S. Department of the Navy, 2011a), *Record of Decision for Final Environmental Impact Statement/Overseas Environmental Impact Statement for the Gulf of Alaska Navy Training Activities* (U.S. Department of the Navy, 2011b), the 2016 GOA Final SEIS/OEIS (U.S. Department of the Navy, 2016), and *Record of Decision for the Gulf of Alaska Final Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement* (U.S. Department of the Navy, 2017). The Proposed Action, described in detail in Chapter 2 (Description of Proposed Action and Alternatives), entails the military continuing training activities previously conducted and as described in the 2016 GOA Final SEIS/OEIS, for which a Record of Decision was issued. Though the types of activities and level of events in the Proposed Action are the same as in the previous documents (Alternative 1 in both the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS), there have been minor changes in the platforms and systems used as part of those activities.

Sonar and other transducers proposed for use are typically transient and temporary because activities that involve sonar and other transducers take place at different locations and many platforms are generally moving throughout the TMAA. In addition, the Proposed Action would occur over a maximum time period of up to 21 consecutive days during the months of April–October, further limiting the total potential time sonar and other transducers may impact fishes within the TMAA. General categories and characteristics of sonar systems and the number of hours these sonars would be operated during training under Alternative 1 are described in Section 3.0.4.1.1 (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 (Navy Activities Descriptions). The proposed use of sonar for training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Although the existing conditions have not changed appreciably, and no new Navy training activities are proposed in the TMAA in this SEIS/OEIS, a detailed re-analysis of Alternative 1 with respect to fishes is provided here to supplant previous analyses based on available new literature, adjusted sound exposure criteria, and new acoustic effects modeling.

All marine fishes detect low-frequency sound. However, low-frequency sources would not be used as part of this proposed action and therefore are not analyzed further. As shown in Figure 3.6-2, the majority of marine fish species present within the TMAA are not expected to detect sounds in the mid-frequency range above a few kHz. The fish species that are known to detect mid-frequencies up to a few kHz (i.e., those with swim bladders, including some sciaenids [drum], most clupeids [herring], and potentially deep-water fish such as myctophids [lanternfish]) do not have their best sensitivities in the range of the operational sonars. Thus, these species 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 are more susceptible to hearing loss due to exposure to mid-frequency sonars; however, the maximum estimated range to TTS for fish within this hearing group is equal to or less than 10 m for the most powerful sonar bin. Fishes within this hearing group would have to be very close to the source and the source levels would have to be relatively high in order to experience TTS. Most marine species lack these hearing specializations and therefore would be unable to detect sound greater than approximately 1 kHz and likewise would not be susceptible to TTS from these sound sources.

Most mid-frequency active sonars used in the TMAA 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 resulting from 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 masking effects would be temporary and infrequent as a vessel operating mid-frequency sonar transits an area.

Fishes that are able to detect sonar and other transducers above a few kHz within near (tens of meters) to far (thousands of meters) distances of the source would be more likely to experience: mild physiological stress or behavioral reactions such as startle or avoidance responses, although risk would be low even close to the source; or no reaction. 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, impacts, if any, would be localized and infrequent, only lasting a few seconds or minutes. As such, mid-frequency sonar use is unlikely to impact individuals or impacts would likely be insignificant. Based on the low level and short duration of potential exposure to sonar and other transducers and the limited number of days the Proposed Action would occur in a given year (21 consecutive days), long-term consequences for fish populations are not expected.

Various ESA-listed populations of salmonids (Chinook salmon, coho salmon, chum salmon, sockeye salmon, and steelhead) migrate north to mature in the Gulf of Alaska and may occur in the TMAA. As discussed in Table 3.6-2, juvenile salmonids predominantly occur in coastal waters on the continental shelf and along the slope with the exception of juvenile chum and steelhead salmon which could occur in portions of the TMAA farther offshore. Immature and maturing adult salmonids may occur throughout the TMAA (near and offshore) with seasonal and interannual variability depending on the species and population of interest. In addition, the Southern DPS of green sturgeon (not previously analyzed), although rare, has the potential to occur in the TMAA. If green sturgeon are present within the TMAA, it is more likely that they would occur in coastal areas on the shelf rather than in the open ocean. Sonar and other transducers would be used throughout the TMAA and therefore may overlap areas where any of these ESA-listed species could occur.

As discussed previously in Section 3.6.2.1.3 (Hearing and Vocalization) and as shown in Figure 3.6-2, all ESA-listed salmonids and green sturgeon are capable of detecting sound produced by some mid-frequency sonars and other transducers. Specifically, ESA-listed salmonids and green sturgeon have a swim bladder not involved in hearing and may be able to detect some mid-frequency sources operating below 2 kHz, but they are not particularly sensitive to these frequencies. In addition, there are only a few sources utilized within the TMAA that would potentially overlap frequencies ESA-listed fishes could detect, limiting the overall impact from exposure. Furthermore, 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. Physiological and behavioral reactions would be expected to be brief (seconds to minutes) and infrequent based on the low probability of co-occurrence between training activities and these species. Therefore, impacts from sonar and other transducers would be minor and insignificant for all ESA-listed species.

Although ESA-listed salmonids and green sturgeon have designated critical habitat, none of the designated critical habitat occurs within the TMAA; therefore, critical habitat for these species would not be impacted.

As described above, there is new information that applies to the analysis of impacts of sonar and other transducers on fishes. Though the types of activities and level of events in the Proposed Action are the same as in the previous documents (Alternative 1 in both the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS), there have been changes in the platforms and systems used as part of those activities. However, this new information does not substantively change the affected environment, which forms the environmental baseline of the analysis in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Additionally, no new Navy training activities are being proposed in this SEIS/OEIS that would affect fishes in the TMAA. Therefore, conclusions for fishes made for Alternative 1 that were analyzed in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS remain unchanged in this SEIS/OEIS. For a summary of effects of the action alternative on fishes under both the National Environmental Policy Act (NEPA) and EO 12114, please refer to Table 3.6-11 in the 2011 GOA Final EIS/OEIS.

In addition, new evidence suggests that ESA-listed green sturgeon may be present in the TMAA where they were not previously anticipated to occur. Therefore, conclusions based on the present analysis are also made for green sturgeon.

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

3.6.3.1.3 Impacts from Vessel Noise

3.6.3.1.3.1 Impacts from Vessel Noise Under the No Action Alternative

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

3.6.3.1.3.2 Impacts from Vessel Noise Under Alternative 1

Training activities within the TMAA involve maneuvers by various types of surface ships, boats, and submarines (collectively referred to as vessels). Fishes may be exposed to noise from vessel movement throughout the TMAA. A detailed description of the acoustic characteristics and typical sound levels of vessel noise are in Section 3.0.4.1 (Acoustic Sources). Proposed training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS.

Although ESA-listed salmonids and green sturgeon have designated critical habitat, none of the designated critical habitat occurs within the TMAA; therefore, critical habitat for these species would not be impacted.

Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Because the existing baseline conditions have not changed appreciably, and no new Navy training activities are proposed in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to fishes is not warranted.

However, new evidence suggests that ESA-listed green sturgeon may be present in the TMAA, where they were not previously anticipated to occur. Therefore, conclusions based on the previous analysis are also made for green sturgeon as the potential impacts from vessel noise would not differ between all previously analyzed ESA-listed fishes.

Pursuant to the ESA, vessel noise produced during training activities, as described under Alternative 1, may affect ESA-listed salmonids and green sturgeon. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.

3.6.3.1.4 Impacts from Aircraft Noise

3.6.3.1.4.1 Impacts from Aircraft Noise Under the No Action Alternative

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

3.6.3.1.4.2 Impacts from Aircraft Noise Under Alternative 1

Training activities within the TMAA involve maneuvers by various types of fixed, rotary-wing, and tilt-rotor aircraft (collectively referred to as aircraft). Most aircraft noise would be concentrated around airbases and fixed ranges within the range complex, especially in the waters immediately surrounding aircraft carriers at sea during takeoff and landing. Other aircraft overflights include commercial air traffic in addition to U.S. Navy aircraft. Fishes may be exposed to noise from aircraft overflights. A detailed description of the acoustic characteristics and typical sound levels of aircraft overflights are in Section 3.0.4 (Stressors-Based Analysis). Proposed training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS.

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, hearing loss is not considered further 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., vessel noise).

Although ESA-listed salmonids and the green sturgeon have designated critical habitat, none of the designated critical habitat occurs within the TMAA; therefore, critical habitat for these species would not be impacted.

Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Because the existing conditions have not changed appreciably, and no new Navy training activities are being proposed for use in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to fishes is not warranted.

However, new evidence suggests that ESA-listed green sturgeon may be present in the TMAA, where they were not previously anticipated to occur. Therefore, conclusions based on the previous analysis are also made for green sturgeon as the potential impacts from aircraft noise would not differ between all previously analyzed ESA-listed fishes.

Pursuant to the ESA, aircraft noise produced during training activities, as described under Alternative 1, may affect ESA-listed salmonids and green sturgeon. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.

3.6.3.1.5 Impacts from Weapon Noise

3.6.3.1.5.1 Impacts from Weapons Noise Under the No Action Alternative

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

3.6.3.1.5.2 Impacts from Weapon Noise Under Alternative 1

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 (Acoustic Sources). 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 impulsive sounds. Proposed training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) and would be operated within the same location as analyzed under the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS.

Although ESA-listed salmonids and the green sturgeon have designated critical habitat, none of the designated critical habitat occurs within the TMAA; therefore, critical habitat for these species would not be impacted.

Alternative 1 for this SEIS/OEIS remains consistent with the description of Alternative 1 in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Because the existing conditions have not changed appreciably, and no new Navy training activities are being proposed for use in the TMAA in this SEIS/OEIS, a detailed re-analysis of the alternatives with respect to fishes is not warranted.

However, new evidence suggests that ESA-listed green sturgeon may be present in the TMAA, where they were not previously anticipated to occur. Therefore, conclusions based on the previous analysis are also made for green sturgeon as the potential impacts from weapon noise would not differ between all previously analyzed ESA-listed fishes.

Pursuant to the ESA, weapon noise produced during training activities, as described under Alternative 1, may affect ESA-listed salmonids and green sturgeon. The Navy will consult with NMFS as required by Section 7(a)(2).

3.6.3.2 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 in part 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 B (Acoustic and Explosive Concepts).

This section begins with a summary of relevant data regarding explosive impacts on fishes in Section 3.6.3.2.1 (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 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), and this section follows that framework.

Due to available new literature, adjusted sound exposure criteria, and new acoustic effects modeling, the analysis provided in Section 3.6.3.2.2 (Impacts from Explosives) of this FSEIS supplants the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS for fishes, and may change estimated impacts for some species since the 2016 GOA Final SEIS/OEIS. In addition, this analysis includes the consideration of ESA-listed green sturgeon not previously analyzed.

3.6.3.2.1 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 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.

3.6.3.2.1.1 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 (Dahl et al., 2020; Keevin & Hempen, 1997; Wright, 1982; Yelverton & Richmond, 1981; Yelverton et al., 1975). At the same distance from the source, larger fishes are generally less susceptible to death or injury, elongated forms that are round in cross-section are less at risk than deep-bodied forms, and fishes oriented sideways to the blast suffer the greatest impact (Edds-Walton & 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 one (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. As the largest air-filled organ in the body of most fishes with one, the swim bladder is the primary site of damage from explosives (Dahl et al., 2020; 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 cartilaginous 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 kilograms [2–3 pounds]) 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 40–70 psi. However, near the bottom (all water depths were less than 100 feet [ft.]) fish exposed to pressures over twice as high exhibited no sign of injury. Yelverton et al. (1975) similarly found that peak pressure was not correlated to injury susceptibility; instead, injury susceptibility of swim bladder fish at shallow depths (10 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.

Dahl et al. (2020) reported the effects of underwater explosions on one species of Clupeiform fish, Pacific sardines (*Sardinops sagax*), with a physostomous swim bladder (an open swim bladder with direct connection to the gut via the pneumatic duct). Fish were stationed at various distances prior to each explosion, in addition to a control group that was not exposed. Necropsies following explosions observed significant injuries, including fat hematoma, kidney rupture, swim bladder rupture, and reproductive blood vessel rupture. While most significant injuries were consistently present at close range (<50 m), there were inconsistent findings at the 50–125 m range, suggesting possible acoustic refraction effects, including waveform paths that were bottom reflected, surface reflected, or a combination of both. Ranges at which injuries were observed within the present study are similar to those estimated by the Navy's Acoustic Effects Model for fishes with a swim bladder for detonations modeled in Southern California (where the study took place, for ranges see U.S. Department of the Navy, 2018b). The Navy continues to fund similar projects, including survival studies and those examining other types of fish (such as physoclists, species with a closed swim bladder), as they are crucial to consider before extrapolating findings to other fish species.

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. In general, fish at greater depths and near the surface are predicted to be less likely to be injured because geometries of the exposures would limit the amplitude of swim bladder oscillations. In contrast, detonations at or near the surface (i.e., similar to most Navy activities that utilize bombs and missiles) would result in energy loss at the water-air interface, resulting in lower overall ranges to effect than those predicted here.

In contrast to fish with swim bladders, fishes without swim bladders have been shown to be more resilient to explosives (Gaspin, 1975; Gaspin et al., 1976; Goertner et al., 1994). For example, some small (average 116 millimeter length; approximately 1 ounce) hogchokers (*Trinectes maculatus*) exposed less than 5 ft. from a 10 pound 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.

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.

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.

3.6.3.2.1.2 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., 2014; Popper et al., 2005; 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 3.6.3.1.1.2 (Hearing Loss).

3.6.3.2.1.3 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 3.0.4.3 (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 3.6.3.2.1.3 (Masking). Potential masking from explosives is likely to be similar to masking studied for other impulsive sounds such as air guns.

3.6.3.2.1.4 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 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities) provides additional information on physiological stress and the framework used to analyze this potential impact.

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 this section. 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.

3.6.3.2.1.5 Behavioral Reactions

As discussed in Section 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), any stimuli in the environment can cause a behavioral response in fishes, including sound and energy produced by explosions. Alterations in natural behavior patterns due to exposure to explosions have not been studied as thoroughly, but reactions are likely to be similar to reactions studied for other impulsive sounds such as those produced by air guns (e.g., startle response, changes in swim speed and depth). 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 this section.

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).

3.6.3.2.1.6 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 3.0.4.3 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). Physical effects from explosive sources that could lead to a reduction in the population growth rate include mortality or injury, which could remove animals from the reproductive pool, and 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.

3.6.3.2.2 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 activities, synthesizing the background information presented above. The proposed use of explosives for training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details), with one exception. Consistent with the previous analyses for Alternative 1, the sinking exercise activity will not be part of the proposed action for this SEIS/OEIS, and therefore the explosive use associated with that activity is no longer part of this proposed action.

As discussed above, sound and energy from underwater 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. TTS can also impair an animal's abilities, although the individual may recover quickly with little significant effect.

3.6.3.2.2.1 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 activities. Inputs to the quantitative analysis included sound propagation modeling in the Navy's Acoustic Effects Model to the sound exposure criteria and thresholds presented below. Density data for fish species within the TMAA 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 below in Table 3.6-8. 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 3.6.3.2.1.1 (Injury), this threshold likely over-estimates 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., 2012a; Halvorsen et al., 2011, 2012b) were used as a proxy for the analysis in the Atlantic Fleet and Hawaii-Southern California Training and Testing Draft EIS/OEISs (U.S. Department of the Navy, 2018a, 2018b). Upon re-evaluation during consultation with NMFS, the Navy determined that pile driving data was not appropriate and over conservative for use in the analysis of explosive effects on fishes. The Navy recommended a different peak pressure threshold derived from explosive literature be utilized in the analysis. Consequently, this threshold was later used and published in the Navy's Final EIS/OEISs (U.S. Department of the Navy, 2018a, 2018b) and is recommended for future analyses until better information can be obtained to inform explosive sound exposure criteria. Although NMFS agreed to evaluate the use of the Navy's proposed threshold in future consultations (National Marine Fisheries Service, 2018a), NMFS does not currently have a formal criteria established for explosive thresholds effects on fishes. Therefore, the Navy's injury criteria have been revised as follows.

Thresholds for the onset of injury from exposure to an explosion 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).

Table 3.6-8: Sound Exposure Criteria for Mortality and Injury from Explosives for All Fishes

Onset of Mortality	***Onset of Injury***
SPL_{peak}	***SPL_{peak}***
229	220

Note: SPLpeak = Peak sound pressure level.

It is important to note that some of the available literature is not peer-reviewed and there may be some caveats to consider when reviewing the data (e.g., issues with controls, limited details on injuries observed) 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 3.6.3.2.1.1 (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 3.6-8 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.

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 3.6.3.2.1.1 (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 3.6-9. 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 3.6.3.2.1.2 (Hearing Loss).

Table 3.6-9: Sound Exposure Criteria for Hearing Loss from Explosives

Fish Hearing Group	***TTS (SEL_{cum})***
Fishes with a swim bladder not involved in hearing	> 186
Fishes with a swim bladder involved in hearing	186

Notes: TTS = Temporary Threshold Shift, SEL_{cum} = Cumulative sound exposure level (decibel referenced to 1 micropascal squared seconds [dB re 1 μPa^2-s]), > indicates that the given effect would occur above the reported threshold.

As discussed below Section 3.6.3.1.1.2 (Hearing Loss, Hearing Loss due to Impulsive Sound Sources), 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 3.6-9.

3.6.3.2.2.2 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 3.6-10 and Table 3.6-11 and the Navy Acoustic Effects Model. All detonations conducted during Navy activities would occur at or near the surface. Specifically, as discussed in Section 3.0.4.2.1 (Explosions in Water), detonations occurring in air at a height of 33 ft. (10 m) or less above the water surface, and detonations occurring directly on the water surface, were modeled to detonate at a depth of 0.3 ft. (0.1 m) below the water surface since there is currently no means to model impacts from in-air detonations. 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. In addition, not all fish present within these ranges would be predicted to receive the associated effect as there are portions of the water column within these ranges that would not exceed the threshold (see Figure 3.0-1 for details). Ranges may vary greatly depending on factors such as the location, water depth, and season of the event.

Table 3.6-10 provides range to mortality and injury for all fishes. Only one table (Table 3.6-11) is provided for range to TTS for fishes with a swim bladder. Ranges to TTS for fishes with a swim bladder not involved in hearing and those without a swim bladder would be shorter than those reported because this effect has not been observed in fishes within those hearing groups.

Table 3.6-10: Range to Mortality and Injury for All Fishes from Explosives

Bin[1]	*Range to Effects (meters)*	
	Onset of Mortality ***229 SPL$_{peak}$***	***Onset of Injury*** ***220 SPL$_{peak}$***
E5	175 (170–180)	445 (440–450)
E9	500 (500–500)	1,025 (1,025–1,025)
E10	638 (625–650)	1,400 (1,275–1,525)
E12	800 (800–800)	1,775 (1,775–1,775)

[1] Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650–1,000)
Notes: (1) NEW = net explosive weight, SPL$_{peak}$ = Peak sound pressure level. (2) Range to effects represent modeled predictions in different areas and seasons within the Action Area.
Each cell contains the estimated average, minimum and maximum range to the specified effect. Minimal variability in modeled environmental parameters (e.g., low variation in sound speed profile between simulations, deep water/lack of bottom reflections) can lead to low variation between the estimated average, minimum, and maximum ranges for a given bin and threshold, or cases where these ranges are identical.

Table 3.6-11: Range to TTS for Fishes with a Swim Bladder from Explosives

Bin[1]	*Cluster Size*	*Range to Effects (meters)*
		TTS
		SEL$_{cum}$
E5	1	155 (150–160)
	7	365 (360–370)
E9	1	450 (440–460)
E10	1	563 (550–575)
E12	1	711 (700–750)

[1] Bin (net explosive weight, lb.): E5 (> 5–10), E9 (> 100–250), E10 (> 250–500), E12 (> 650– 1,000)
Notes: (1) NEW = net explosive weight, SELcum = Cumulative sound exposure level, TTS = Temporary Threshold Shift. (2) Range to effects represent modeled predictions in different areas and seasons within the Action Area.
Each cell contains the estimated average, minimum and maximum range to the specified effect. Minimal variability in modeled environmental parameters (e.g., low variation in sound speed profile between simulations, deep water/lack of bottom reflections) can lead to low variation between the estimated average, minimum, and maximum ranges for a given bin and threshold.

3.6.3.2.2.3 Impacts from Explosives Under the No Action Alternative

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

3.6.3.2.2.4 Impacts from Explosives Under Alternative 1

The Navy's Proposed Action is consistent with the Proposed Action presented in the 2011 GOA Final EIS/OEIS (U.S. Department of the Navy, 2011a), *Record of Decision for Final Environmental Impact Statement/Overseas Environmental Impact Statement for the Gulf of Alaska Navy Training Activities* (U.S. Department of the Navy, 2011b), the 2016 GOA Final SEIS/OEIS (U.S. Department of the Navy, 2016), and *Record of Decision for the Gulf of Alaska Final Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement* (U.S. Department of the Navy, 2017). The Proposed Action, described in detail in Chapter 2 (Description of Proposed Action and Alternatives), entails the military continuing training activities previously conducted and as described in the 2016 GOA Final SEIS/OEIS, for which a Record of Decision was issued. Though the types of activities and level of events in the Proposed Action are the same as in the previous documents (Alternative 1 in both the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS), there have been minor changes in the platforms and systems used as part of those activities.

Training activities under Alternative 1 would use surface or near-surface detonations and explosive ordnance. The use of explosives would occur throughout the TMAA, the same location as analyzed under the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS, and are typically dispersed in space and time. The number and type (i.e., source bin) of explosives that would be used during training under Alternative 1 are described in Section 3.0.4.2 (Explosive Stressors). Activities using explosives would be conducted as described in Chapter 2 (Description of Proposed Action and Alternatives) and Appendix A (Navy Activities Descriptions). The proposed use of explosives for training activities would be almost identical to what is currently conducted (see Table 2.6-1 for details) with one exception. Consistent with the previous analyses for Alternative 1, the sinking exercise activity will not be part of the proposed action for this SEIS/OEIS. Although the existing conditions have not changed appreciably, and no new Navy training activities are proposed in the TMAA in this SEIS/OEIS, a detailed re-analysis of Alternative 1 with respect to fishes is provided here to supplant previous analyses based on available new literature, adjusted sound exposure criteria, and new acoustic effects modeling.

To avoid or reduce potential impacts from explosive training activities on marine species, and as described in Chapter 5 (Mitigation), the Navy will implement mitigation to not use in-water explosives June through September in the North Pacific Right Whale Mitigation Area and April through October in the Portlock Bank Mitigation Area. Mitigation will help the Navy avoid or reduce potential impacts on fish and fishery resources in these important habitat areas in the TMAA.

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 3.6-10 and Table 3.6-11. 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). 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. However, the Proposed Action would only occur over a maximum time period of up to 21 consecutive days during the months of April–October, further limiting the total potential time explosives may impact fishes throughout the TMAA.

If an individual fish were repeatedly exposed to sound and energy from underwater 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 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.

Various ESA-listed populations of salmonids (Chinook salmon, coho salmon, chum salmon, sockeye salmon, and steelhead) migrate north to mature in the Gulf of Alaska and may occur in the TMAA. As discussed in Table 3.6-2, juvenile salmonids predominantly occur in coastal waters on the continental shelf and along the slope with the exception of juvenile chum salmon and steelhead which could occur in portions of the TMAA farther offshore. Immature and maturing adult salmonids may occur throughout the TMAA (both near and offshore) with seasonal and interannual variability depending on the species and population of interest. In addition, the Southern DPS of green sturgeon (not previously analyzed), although rare, has the potential to occur in the TMAA. If green sturgeon are present within the TMAA, it is more likely that they would occur in coastal areas on the shelf rather than in the open ocean. Each of these ESUs could only be exposed to explosive activities during the time they are present in the TMAA and during the same 21 consecutive days in which the Proposed Action would occur. Generally, surface-oriented fishes and those that occur in the top tens of meters of the water column, such as some ESA-listed salmonids, have a higher potential of being exposed to and affected by detonations at or above the water’s surface. There would be little to no potential for species such as sturgeon, which typically occur at greater depths or on the seafloor, for exposure to sound and energy produced by detonations at or above the surface.

As discussed previously in Section 3.6.2.1.3 (Hearing and Vocalization), all ESA-listed fish species that occur in the TMAA are capable of detecting sound produced by explosives. Impacts on ESA-listed fishes, if they occur, would be similar to impacts on fishes in general. 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 (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.

Although ESA-listed salmonids and the green sturgeon have designated critical habitat, none of the designated critical habitat occurs within the TMAA; therefore, critical habitat for these species would not be impacted.

As described above, there is new information that applies to the analysis of impacts of explosives on fishes. Though the types of activities and level of events in the Proposed Action are the same as in the previous documents (Alternative 1 in both the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS), there have been changes in the platforms and systems used as part of those activities. However, this new information does not substantively change the affected environment, which forms the environmental baseline of the analysis in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS. Additionally, no new Navy training activities are being proposed in this SEIS/OEIS that would affect fishes in the TMAA. Therefore, conclusions for fishes made for Alternative 1 analyzed in the 2011 GOA Final EIS/OEIS and the 2016 GOA Final SEIS/OEIS remain unchanged in this SEIS/OEIS. For a summary of effects of the action alternative on fishes under both the NEPA and Executive Order (EO) 12114, please refer to Table 3.6-11 in the 2011 GOA Final EIS/OEIS.

In addition, new evidence suggests that ESA-listed green sturgeon may be present in the TMAA where they were not previously anticipated to occur. Therefore, conclusions based on the present analysis are also made for green sturgeon.

Pursuant to the ESA, the use of explosives during training activities, as described under Alternative 1, may affect ESA-listed salmonids and green sturgeon. The Navy will consult with NMFS as required by Section 7(a)(2) of the ESA.

3.6.4 Summary of Stressor Assessment (Combined Impacts of All Stressors) on Fish

As described above, there is new information on fish stock assessment reports, tagging studies, and fish hearing since the 2016 GOA Final SEIS/OEIS was prepared. However, this new information does not significantly change the affected environment, which forms the environmental baseline of the fish analysis in the 2011 GOA Final EIS/OEIS and 2016 GOA Final SEIS/OEIS. Additionally, no new Navy training activities are being proposed in this SEIS/OEIS that would affect fishes in the TMAA. Therefore, conclusions for impacts on fish species made for the stressors that were not re-analyzed in this SEIS/OEIS remain unchanged from the conclusions under Alternative 1 analyzed in the 2011 GOA Final EIS/OEIS, and training activities do not compromise productivity of fishes or impact their habitats. For a summary of effects due to other stressors not reanalyzed in this SEIS/OEIS for Alternative 1 on fishes under both NEPA and EO 12114, please refer to Table 3.6-11 in the 2011 GOA Final EIS/OEIS.

Essential Fish Habitat Determinations

According to 50 Code of Federal Regulations Section 600.920(a), a supplemental consultation for EFH is required for renewals, reviews, or substantial revisions of actions if these actions may adversely affect EFH. There are no changes to Navy activities or designated EFH in the TMAA that are substantial in nature and that may adversely affect EFH previously analyzed. The Navy is corresponding with the NMFS regarding new species and life stages for scallop and groundfish species and other EFH to ensure concurrence. The analysis previously captured in Appendix C (Regulatory Consultations) of the 2011 GOA Final EIS/OEIS (U.S. Department of the Navy, 2011a) remains unchanged.

Endangered Species Act

As part of the SEIS/OEIS, the Navy is consulting under section 7 of the ESA with NMFS for the ESA-listed fishes and will continue to rely on the prior analysis from the 2011 GOA Final EIS/OEIS, as reviewed and amended by this SEIS/OEIS, and Biological Evaluation, as they remain valid. Specifically, there has not been an exceedance of incidental take for listed fishes under the current Biological Opinion; there is no new information that reveals new effects to listed fish species or critical habitat for listed fishes that were not previously considered; and Navy training activities in the TMAA are not being substantially modified in a manner that would affect listed fish species or their critical habitat that was not previously considered. Two new salmonid ESUs are being considered for listing under the ESA, and NMFS is proposing to issue new species status reviews in the near future. However, the proposed project would not affect any future listed salmonids differently than those already evaluated in the 2016 GOA Final SEIS/OEIS or Biological Evaluation. Due to new information on ESA-listed salmonid occurrence and the presence of the southern DPS of green sturgeon, this SEIS/OEIS and Biological Assessment have expanded the number of potentially-occurring ESA-listed fishes addressed from the 2016 GOA Final SEIS/OEIS.

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