UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Silver Spring, MD 20910

APR 19 2017

Mr. L.M. Foster
Commander, United States Pacific Fleet
250 Makalapa Drive
Pearl Harbor, Hawaii 96860-3131

Refer to NMFS No.: FPR-2015-9118

RE: Endangered Species Act Section 7 biological opinion on U.S. Navy Gulf of Alaska training activities and the National Marine Fisheries Service's proposal to issue regulations and letters of authorization for incidental take of marine mammals pursuant to the Marine Mammal Protection Act

Dear Mr. Foster:

Enclosed is the National Marine Fisheries Service's (NMFS) biological opinion on the effects of the U.S. Navy's Gulf of Alaska training activities for the period April 2017 through April 2022 on endangered and threatened species under NMFS' jurisdiction and critical habitat that has been designated for those species. This consultation also considers the NMFS, Office of Protected Resources, Permits and Conservation Division's issuance of regulations pursuant to the Marine Mammal Protection Act (MMPA), and a letter of authorization for the take of marine mammals incidental to Navy Gulf of Alaska training activities. We have prepared the biological opinion pursuant to section 7(a)(2) of the Endangered Species Act, as amended (ESA; 16 U.S.C. 1536(a)(2)).

Based on our assessment, we concluded that U.S. Navy Gulf of Alaska training activities are likely to adversely affect, but not likely to jeopardize the continued existence of, threatened and endangered species under NMFS' jurisdiction and are not likely to adversely affect, nor destroy or adversely modify, designated critical habitat over the period of the proposed five-year MMPA rule and letter of authorization, and into the reasonably foreseeable future. The enclosed biological opinion supersedes the 2011 and 2013 biological opinions on the U.S. Navy's military readiness activities in the Gulf of Alaska Temporary Maritime Activities Area.

This concludes section 7 consultation on U.S. Navy Gulf of Alaska training activities and NMFS Permits and Conservation Division's actions pursuant to the MMPA. The U.S. Navy is required to reinitiate formal consultation on the proposed action, where it retains discretionary involvement or control over the action and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this formal consultation pursuant to ESA section 7(a)(2); (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this formal consultation pursuant to ESA section 7(a)(2); or (4) a new species is listed or critical habitat designated that may be affected by the action.

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If you have any questions regarding this biological opinion, please contact Cathy Tortorici at (301) 427-8495 or cathy.tortorici@noaa.gov.

Sincerely,

Donna S. Wieting

Donna S. Wieting
Director, Office of Protected Resources

NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 BIOLOGICAL OPINION

Action Agencies:	The United States Navy (Navy) and NOAA's National Marine Fisheries Service, Office of Protected Resources' Permits and Conservation Division
Activity Considered:	(1) The Navy's Gulf of Alaska Training Activities; and (2) the National Marine Fisheries Services' promulgation of regulations pursuant to the Marine Mammal Protection Act for the Navy to "take" marine mammals incidental to Gulf of Alaska activities from April 2017 through April 2022; and (3) the National Marine Fisheries Services' issuance of a Letter of Authorization to the Navy pursuant to regulations under the Marine Mammal Protection Act to "take" marine mammals incidental to Gulf of Alaska activities from April 2017 through April 2022
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service
Approved:	Donna S. Wieting Director, Office of Protected Resources
Date:	APR 19 2017
Public Consultation Tracking System (PCTS) number:	FPR-2015-9118

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1 INTRODUCTION

Section 7 (a)(2) of the ESA requires Federal agencies to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with NOAA's National Marine Fisheries Service (NMFS) or the US Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the USFWS concurs with that conclusion (50 CFR §402.14(b)).

Section 7 (b)(3) of the ESA requires that at the conclusion of consultation, NMFS and/or USFWS provide an opinion stating how the Federal agencies' actions will affect ESA-listed species and their critical habitat under their jurisdiction. If an incidental take is expected, section 7 (b)(4) requires the consulting agency to provide an incidental take statement that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts.

For the actions described in this document, the action agencies are the United States Navy (Navy), which proposes to continue military training activities, and NMFS Office of Protected Resources - Permits and Conservation Division (Permits Division). The Permits Division proposes to promulgate regulations pursuant to the Marine Mammal Protection Act of 1972, as amended (MMPA 16 U.S.C. 1361 et seq.), related to the Navy's training activities in the Gulf of Alaska TMAA that may affect several ESA-listed species. The regulations propose to authorize the issuance of a Letter of Authorization (LOA) that will allow the Navy to "take" marine mammals incidental to its proposed action. The Federal action of issuing an LOA to the Navy is also considered in this biological opinion (opinion). The consulting agency for these proposals is NMFS Office of Protected Resources - Endangered Species Act Interagency Cooperation Division.

The opinion and incidental take statement were prepared by NMFS' Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR §402. This document represents NMFS' opinion on the effects of these actions on endangered and threatened species and critical habitat that has been designated for those species. A complete record of this consultation is on file at NMFS' Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

In April 2011, NMFS issued a programmatic biological opinion on the U.S. Navy's proposed training activities in the Gulf of Alaska Temporary Maritime Activities Area (TMAA) from April 2011 to April 2016 and NMFS Permits and Conservation Division's proposal to promulgate regulations pursuant to the MMPA to authorize the Navy to "take" marine mammals incidental to training in the TMAA from April 2011 to April 2016.

In May 2011, NMFS issued a biological opinion on the Navy's proposed training activities in the Gulf of Alaska TMAA from May 2011 to May 2013 and NMFS Permits Division's issuance of a Letter of Authorization for the Navy to "take" marine mammals incidental to training in the Gulf of Alaska TMAA during that time period.

In May 2013, NMFS issued a biological opinion on the Navy's proposed training activities in the Gulf of Alaska TMAA from May 2013 to May 2016 and NMFS Permits Division's issuance of a Letter of Authorization for the Navy to "take" marine mammals incidental to training in the Gulf of Alaska TMAA during that time period.

This opinion is based on information provided during pre-consultation and in the Navy's 2015 request for ESA consultation package, including draft Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement (SEIS/OEIS), and supplemental information provided throughout the consultation period. This opinion also considers information provided by NMFS' Permits Division, including its request for Section 7 consultation under the ESA, which included the proposed Federal regulations under the MMPA specific to the proposed activities and draft letter of authorization. Also considered were draft or final recovery plans for the endangered or threatened species that are considered in this document, and publications that we identified, gathered, and examined from the public scientific literature, including new information that has become available since the issuance of the previous biological opinions on U.S. Navy Gulf of Alaska training activities.

1.2 Consultation History

- On February 18, 2015, the Navy requested initiation of formal consultation for threatened or endangered marine mammals for the U.S. Navy's Gulf of Alaska training activities from 2016 to 2021. The Navy's ESA determinations for the use of sonar and other active acoustic (non-impulsive) sources for marine mammals were:
    - may affect, likely to adversely affect, North Pacific right whale, humpback whale, blue whale, fin whale, sei whale, sperm whale, and the Western DPS Steller sea lion
    - may affect, not likely to adversely affect, western North Pacific gray whale

The ESA determinations for the use of explosive (impulsive) sources for marine mammals were:

- may affect, not likely to adversely affect, North Pacific right whale, humpback whale, blue whale, fin whale, sei whale, western North Pacific gray whale, sperm whale, and the Western DPS Steller sea lion

In the Navy's request for consultation, the Navy stated that the criteria for re-initiation of formal consultation (as set forth in 50 Code of Federal Regulations (C.F.R.) §402.16) had not been triggered for fish and sea turtles because there was no new information that would change the affected environment or analysis, no new Navy training activities being proposed, and no new or modified ESA status or critical habitat in the TMAA. Therefore NMFS's conclusions for fish and sea turtles (i.e., leatherback, loggerhead, green, and olive ridley sea turtles) as stated in the 2011 GOA biological opinion were incorporated by reference within the request for initiation and biological evaluation. However, NMFS did not quantify the amount or extent of take of fish species in the 2011 (or 2013) GOA biological opinion. Since issuance of that opinion, NMFS has developed interim criteria for assessing effects to fish from explosives and has a better understanding of fish distribution in the action area that would allow quantification of take in this opinion. NMFS therefore reinitiated formal consultation for fish and assesses those species in this opinion.

- On April 8, 2015, NMFS determined that there was sufficient information in the Navy's request to initiate formal consultation. NMFS also proposed to complete the biological opinion on or before March 4, 2016, prior to promulgation of the MMPA rule (incidental take authorization). Since this proposed timeline extended beyond the normal 135 days, NMFS requested the Navy's mutual agreement on the proposed timeline of approximately 330 days.

- On April 9, 2015, NMFS received the Navy's agreement to extend the consultation to 330 days with delivery of the final biological opinion on or before March 4, 2016.

- On February 9, 2016, NMFS provided a draft biological opinion to the Navy.

- On February 26, 2016, NMFS received comments from the Navy on the draft biological opinion.

- In spring 2016, NMFS and Navy agreed to extend consultation based on the dates of the proposed action changing to April 2017 to April 2022 (i.e., changed from 2016 to 2021).

- On January 13, 2017, the Navy informed NMFS via a memo for the record (dated January 12, 2017) that the proposed action would be changed from Alternative 2 to Alternative 1, as described in the 2016 Gulf of Alaska Final Supplemental Environmental Impact Statement.

This change resulted in a reduction in the total anticipated amount of annual training activities.

- On February 23, 2017, NMFS provided Navy with a revised draft biological opinion that reflected the change in the proposed action from Alternative 2 to Alternative 1.

- On March 9, 2017, NMFS received comments from the Navy on the draft biological opinion.

2 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. Interrelated actions are those that are part of a larger action and depend on that action for their justification. Interdependent actions are those that do not have independent use, apart from the action under consideration.

This opinion addresses three interdependent actions: (1) the Navy's military training activities conducted in the Gulf of Alaska TMAA; (2) the regulations proposed by NMFS's Permits Division pursuant to the MMPA governing the Navy's "take" of marine mammals incidental to the Navy's military readiness activities from April 2017 through April 2022; and (3) NMFS Permits Division's proposed issuance of an LOA pursuant to the proposed regulations that will authorize the Navy to "take" marine mammals incidental to military readiness activities in the Gulf of Alaska TMAA through April 2022. This opinion supersedes the biological opinions for the U.S. Navy's Gulf of Alaska training activities issued in April 2011 and May 2013.

The purpose of the military readiness activities the Navy conducts in the GOA TMAA is to achieve and maintain fleet readiness and to meet the Navy's Title 10 mission to maintain, train, and equip combat-ready naval forces capable of winning wars, deterring aggression, and maintaining freedom of the seas. The activities covered are major joint training exercises (often called Northern Edge) in Alaska and off the Alaskan coast that involve the Departments of the Navy, Army, Air Force, and Coast Guard participants coordinated to demonstrate and evaluate the ability of the services to engage in a conflict and carry out plans in response to a threat to national security. The proposed action is to conduct a joint exercise over a maximum of 21 consecutive days on an annual basis in the GOA TMAA. These activities would occur in the time period of April through October. The purpose of the MMPA regulations and the Permits and Conservation Division's LOA is to allow the Navy to "take" marine mammals incidental to military readiness activities in the Gulf of Alaska TMAA conducted through April 2022 in a manner that is consistent with the requirements of the MMPA and implementing regulations.

NMFS recognizes that while Navy training requirements change over time in response to global or geopolitical events and other factors, the general types of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assumed that the training

activities proposed by the Navy during the period of NMFS' proposed incidental take authorization pursuant to the MMPA (April 2017 through April 2022) would continue into the reasonably foreseeable future at levels similar to those assessed in this opinion and described in the Gulf of Alaska Supplemental EIS/OEIS.

The tempo of training within the Gulf of Alaska TMAA is subject to variation within the scope of the activities described in the Navy's Supplemental EIS/OEIS and this opinion. Annual variation in the number of training events and quantities of authorized sonar systems and explosive training could occur based on:

- Frequency of out-of-area training deployments to other Navy range complexes;
- Overseas deployments of ships and aircraft to the western Pacific and Middle East;
- Within-area maintenance and repair work that precludes completing some training within the Gulf of Alaska, and
- Certification and training needs for a given ship, submarine, or aircraft crew (e.g., some units could require a certain amount of one kind of training versus another).

Given the inherent uncertainty and potential variation within the training spectrum due to unforeseen world events, the Navy stated that it cannot predict exact annual system use for the period.

2.1 U.S. Navy Training Activities

The Proposed Action covered in this opinion includes the training activities described in Alternative 1 in the Gulf of Alaska Final Supplemental FSEIS/OEIS. Navy activities within the Gulf of Alaska would only occur between April and October of any given year as described in the Navy's FSEIS/OEIS. The action area is described in detail below in Section 2.6. Typical training activities covered in this opinion are described in more detail within the Gulf of Alaska FSEIS/OEIS.

The Navy categorizes training activities into functional warfare areas called primary mission areas. Most training activities analyzed in this opinion fall into the following primary mission areas:

- Anti-Air Warfare (AAW)
- Anti-Surface Warfare (ASUW)
- Anti-Submarine Warfare (ASW)
- Electronic Combat (EC)
- Naval Special Warfare (NSW)

- Strike Warfare (STW)

The training activities proposed by the Navy are briefly described in and enumerated in Table 1. The table is organized according to primary mission areas and includes the activity name and a short description.

Table 1. Representative Training Activities Occurring in the Action Area

Activity Name	Activity Description
Anti-Air Warfare (AAW)	
Air Combat Maneuver (ACM)	Aircrews engage in flight maneuvers designed to gain a tactical advantage during combat.
Air Defense Exercise	Train surface and air assets in coordination and tactics for defense of the strike group or other Naval Forces from airborne threats.
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	Surface ship crews defend against threat missiles and aircraft with missiles.
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A])	Surface ship crews defend against threat aircraft or missiles with guns.
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	Aircrews defend against threat aircraft with missiles.
Anti-Surface Warfare (ASUW)	
Visit, Board, Search, and Seizure	Teams of personnel are deployed from ships at sea into small zodiac boats to board and inspect ships and vessels suspected of carrying contraband.
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	Fixed-wing aircrews simulate firing precision-guided missiles using captive air training missiles against surface targets. There is no firing of explosive missiles.
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Fixed-wing aircrews deliver bombs against surface targets.
Gunnery Exercise (Surface-to-Surface) (GUNEX [S-S])	Ship and small boat crews engage surface targets with ship's small-, medium-, and large-caliber guns. Some of the small- and medium-caliber gunnery exercises analyzed include those conducted by the U.S. Coast Guard.
Maritime Interdiction	A coordinated defensive preplanned attack against multiple sea-borne and air targets using airborne and surface assets.
Sea Surface Control	Airborne assets investigate surface contacts of interest and attempt to identify, via onboard sensors or cameras, the type, course, speed, name, and other pertinent data about the ship of interest.

Activity Name	Activity Description
Anti-Submarine Warfare (ASW)	
Tracking Exercise – Helicopter (TRACKEX – Helo)	Helicopter crews search for, detect, and track submarines.
Tracking Exercise – Maritime Patrol Aircraft (TRACKEX – MPA)	Maritime patrol aircraft crews employ sonobuoys to search for, detect, and track submarines.
Tracking Exercise – Maritime Patrol Aircraft (Multi-static Active Coherent [MAC]) (TRACKEX MPA MAC)	Maritime patrol aircraft crews search for, detect, and track submarines using MAC sonobuoys.
Tracking Exercise – Surface (TRACKEX – Surface)	Surface ship crews search for, detect, and track submarines.
Tracking Exercise – Submarine (TRACKEX – Sub)	Submarine crews search for, detect, and track submarines and surface ships.
Electronic Combat (EC)	
EC Exercises	Aircraft fly threat profiles against ships so that the ship's crews are trained to detect electronic signatures of various threat aircraft and counter the jamming of the ship's own electronic equipment by the simulated threat.
Chaff Exercises	Ships, fixed-winged aircraft, and helicopters deploy chaff to disrupt threat targeting and missile guidance radars and to defend against an attack.
Counter Targeting Exercises	A coordinated, defensive activity utilizing surface and air assets, that attempts to use jamming and chaff to show a false force presentation to inbound surface-to-surface platforms.
Naval Special Warfare (NSW)	
Special Warfare Operations	Training involves specialized tactics, techniques, and procedures, employed in training events that could include insertion/extraction activities using parachutes, rubber boats, helicopters, and other equipment.
Strike Warfare (STW)	
Air-to-Ground Bombing Exercise	Fixed-winged strike fighter aircraft deliver bombs and rockets against land targets.
Personnel Recovery	Train aircrews to locate, protect, and evacuate downed aviation crew members.
Other Training Activities/ Support Operations	
Deck Landing Qualifications	Trains helicopter crews to land on ships underway at sea.

2.2 Training Activity Levels

The Navy uses a variety of sensors, platforms, weapons, and other devices, including those used to ensure the safety of Sailors and Marines, to meet its mission. Training with these systems may introduce acoustic (sound) energy into the environment. This section summarizes sonar systems, ordnance, munitions, targets, and other systems used by the Navy and proposed activity levels

(Table 2). As stated earlier, the Navy's proposed activity includes a maximum of 21 days of activity each year between April and October. Additional details on these activities may be found in Chapter 3 of the Navy's Gulf of Alaska FSEIS/OEIS.

Table 2. Gulf of Alaska Training Activity Levels

Range Activity	2011 Gulf of Alaska Final EIS/OEIS Alternatives (Assessed in the Environmental Baseline)				Changes to the Action in the 2011 Gulf of Alaska Final EIS/OEIS					
	Platform	System or Ordnance	Location	Alternative 1	Platform	System or Ordnance	Location	Alternative 1	Number of events (yearly) or Number of Sonar hours/items (yearly)[5]	Required re-analysis utilizing NAEMO
ANTI-AIR WARFARE (AAW)										
Aircraft Combat Maneuvers	EA-6B, EA-18G, FA-18, F-16, F-15, F-22, E-2	None	TMAA, Air Force SUA[1]	300 sorties	No Change					No
Air Defense Exercise	FA-18, F-16, F-15, F-22, EA-6B, EA-18G, E-2, P-3C, P-8 MMA, CVN, CG, DDG	None	TMAA	4 events	No Change					No
Surface-to-Air Missile Exercise	CVN, CG, DDG	Sea Sparrow Missile, Standard Missile 1, or RAM *Targets:* BQM-74E	TMAA	3 events	No Change					No
Surface-to-Air Gunnery Exercise	CG, DDG, AOE	5-inch/54BLP, 20 mm CIWS, 7.62 mm. *Targets:* Towed TDU-34	TMAA	3 events	No Change					No
Air-to-Air Missile Exercise	FA-18, F-16, F-15, F-22, E-2, EA-6B, EA-18G	AIM-7, AIM-9, AIM-120 *Targets:* TALD or LUU-2B/B	TMAA, Air Force SUA[1]	3 events	No Change					No
ANTI-SURFACE WARFARE (ASUW)										

Visit, Board, Search, and Seizure	MH-60S, RHIB, NSW Personnel	None	TMAA	12 events	No Change				No
Air-to-Surface Missile Exercise	MH-60R/S, FA-18, F-16, F-15, F-22, EA-6B, EA-18G	None	TMAA	2 events	No Change				No
Air-to-Surface Bombing Exercise	FA-18, F-16, F-15, F-22	MK-82 (live), MK-83 (live), MK-84 (live), BDU-45 (inert), MK-58 marine marker	TMAA	18 events	No Change				Yes
Air-to-Surface Gunnery Exercise	MH-60R/S	GAU-16 (0.50 cal) or M-60 (7.62 mm) machine gun *Targets:* HSMST, Trimaran, SPAR, Surface Target Balloon	TMAA	7 events	No Change				No
Surface-to-Surface Gunnery Exercise	CVN, CG, DDG, AOE	5 inch/54 BLP, 20 mm CIWS, 25 mm, 7.62 mm, 57 mm, .50 cal *Targets:* HSMST, Trimaran, SPAR, Surface Target Balloon	TMAA	6 events	No Change				Yes
Maritime Interdiction	All	None	TMAA	14 events	No Change				No
Sea Surface Control	FA-18, EA-6B, EA-18G, E-2, P-3C, P-8 MMA, CG, DDG	None	TMAA	6 events	No Change				No
ANTI-SUBMARINE WARFARE (ASW)									
ASW Tracking Exercise – Helicopter	MH-60R	*Targets*: SSN, MK-39 EMATT Sonobuoys: AN/AQS-22, SSQ-36 BT, SSQ-53	TMAA	22 events	No Change	Same; however, removed SSQ-62 DICASS as	No Change	210 dips (increase of 18 dips due to modeling changes)	Yes

		DIFAR (passive), SSQ-62 DICASS (active), SSQ-77 VLAD Other: MK-58 marine marker				all MF5 bin buoys are now accounted for in ASW Tracking – MPA			
ASW Tracking Exercise – Maritime Patrol Aircraft (MPA)	P-3C, P-8 MMA	*Targets*: SSN, MK-39 EMATT Sonobuoys: SSQ-36 BT, SSQ-53 DIFAR (passive), SSQ-62 DICASS (active), SSQ-77 VLAD Other: MK-58 marine marker	TMAA	13 events	No Change			252 DICASS buoys (decrease of 14 buoys due to modeling changes)	Yes
ASW Tracking Exercise – Extended Echo Ranging (EER) (includes IEER & MAC)	P-3C, P-8 MMA	SSQ-110A EER/IEER, SSQ-125 MAC, SSQ-77 VLAD	TMAA	2 events	No Change	Same; however, removed all SSQ-110A EER/IEER.	No Change	80 MAC buoys were modeled	Yes
ASW Tracking Exercise – Surface Ship	DDG	SQS-53C, SQS-56 MFA sonar *Targets*: SSN, MK-39 EMATT	TMAA	2 events	No Change	Same; however, removed all SQS-56 MFA sonar hours and added them to SQS-53 hours total. Added SQL-25 NIXIE as none were modeled in previous EIS/OEIS.	No Change	619 hours MF1 + MF11 bins (decrease of 2 hours, previously 578 hours of MF1 and 52 hours of MF2, ASW3), NIXIE = 546 hours (NIXIE was not modeled in previous EIS/OEIS)	Yes
ASW Tracking	SSBN, SSGN	*Targets*: MK-39 EMATT	TMAA	2 events	SSN	No Change		48 hours of MF3 (same as before), 24 hours of HF1 (same as before)	Yes

Exercise – Submarine								
ELECTRONIC COMBAT (EC)								
EC Exercises	EA-6B, EA-18G, E-2, P-3, EP-3, CVN, CG, DDG	None	TMAA, Air Force SUA[1]	5 events	No Change			No
Chaff Exercises	EA-6B, EA-18G, P-3, EP-3, FA-18, CVN, CG, DDG, AOE	Chaff	TMAA, Air Force SUA[1]	2 events	No Change			No
Counter Targeting Exercises	EA-6B, EA-18G, P-3, EP-3, FA-18, CVN, CG, DDG, AOE	None	TMAA	4 events	No Change			No
NAVAL SPECIAL WARFARE (NSW)								
Special Warfare Operations	C-130, MH-60S, SDV, RHIB, NSW Personnel	None	TMAA, Air Force SUA[1] Army Training Lands[1]	10 events	No Change			No
STRIKE WARFARE (STW)								
Air-to-Ground Bombing Exercise	FA-18, F-16, F-15, F-22, EA-6B, EA-18G, E-2	MK-82/83/84 (live/inert), BDU-45 (inert), CATM-88C (not released)	Air Force SUA[1], Army Training Lands[1]	150 sorties	No Change			No
Personnel Recovery	CVN, CG, DDG, AOE, E-2, MH-60S, RHIB, NSW Personnel	None	Air Force SUA[1], Army Training Lands[1]	4 events	No Change			No
SUPPORT OPERATIONS								
Deck Landing Qualifications	Helicopters (Air Force, Army, Coast Guard – various)	None	TMAA	6 events	No Change			No

[1] Activities within and upon these areas are covered under separate NEPA analysis.

[2] A sortie is defined as a single activity by one aircraft (i.e., one complete flight from takeoff to landing).

[4] SSN, as a firing platform, was included in original activity description but left off of original table.

[5] ASW is depicted in hours to be consistent with the new modeling technique. Although ASW is modeled as a scenario (multi-day) vice individual events, the hours per event have been provided for clarity.

Notes: AIM = Air Intercept Missile; ASW = Anti-submarine Warfare; BDU = Bomb Dummy Unit; BQM = Aerial Target Drone Designation; cal = caliber; CATM = Combat Arms and Training Maintenance; CG = Cruiser; CVN = Aircraft Carrier, Nuclear; CIWS = Close-in Weapons System; DDG = Destroyer; DICASS = Directional Command Activated Sonobuoy System; DIFAR = Directional Frequency and Ranging; EIS/OEIS = Environmental Impact Statement/Overseas Environmental Impact Statement; EMATT = Expendable Mobile ASW Training Target; EPA = Environmental Protection Agency; Gulf of Alaska = Gulf of Alaska; HARM = High Speed Anti-radiation Missile; HSMST = High Speed Maneuverable Surface Target; IEER = Improved Extended Echo Ranging; MAC = Military Operations in Urban Terrain Assault Course; MFA = Mid-frequency Active; mm = millimeters; MMA = Multi-mission Maritime Aircraft; MPA = Maritime Patrol Aircraft; n/a = not applicable; NAEMO = Navy Acoustic Effects Model; Navy = United States Department of the Navy; NEPA = National Environmental Policy Act; RAM = Rolling Airframe Missile; RHIB = Rigid Hull Inflatable Boat; SDV = Sea, Air, Land Delivery Vehicle; SSBN = Ship, Submersible, Ballistic, Nuclear (submarine); SSGN = Guided Missile Submarine; SSN = Nuclear-Powered Fast Attack Submarine; SUA = Special Use Airspace; TALD = Tactical Air-Launched Decoy; TDU = Target Drone Unit; TMAA = Temporary Maritime Activities Area

2.2.1 Sonar and Other Active Acoustic Sources

Modern sonar technology includes a variety of sonar sensors and processing systems. In concept, the simplest active sonar emits sound waves, or "pings," sent out in multiple directions. The sound waves then reflect off of the target object in multiple directions. The sonar source calculates the time it takes for the reflected sound waves to return; this calculation determines the distance to the target object. More sophisticated active sonar systems emit a ping and then rapidly scan or listen to the sound waves in a specific area. This provides both distance to the target and directional information. Even more advanced sonar systems use multiple receivers to listen to echoes from several directions simultaneously and provide efficient detection of both direction and distance.

It should be noted that active sonar is rarely used continuously throughout the listed activities. In general, when sonar is in use, the sonar "pings" occur at intervals, referred to as a duty cycle, and the signals themselves are very short in duration. For example, sonar that emits a 1-second ping every 10 seconds has a 10 percent duty cycle. The Navy utilizes sonar systems and other acoustic sensors in support of a variety of mission requirements.

Major training exercises conducted in the GOA TMAA can last several weeks, and during those exercises there may be periods of continuous sonar use. Not every major training exercise has anti-submarine warfare events where sonar is used. However, even the longest periods of "continuous" active sonar use rarely last longer than 12 hours, and active sonar use is not truly continuous because a sonar system is actively transmitting a small portion of the time (once per minute for approximately 10 seconds). For Navy active sonar use, a period of concentrated, near continuous anti-submarine warfare sonar use means that sound energy is being put in the water up to approximately two percent of the time. Sonar sound is not transmitting when trying to listen for returns of a detection of a submarine or contact of something else in the water column. Vessels equipped with the most powerful sonar systems would also generally be moving at speeds of 10 to 15 knots.

Primary uses include the detection of and defense against submarines (ASW), safe navigation and effective communications, use of unmanned undersea vehicles, and oceanographic surveys.

All sounds, including sonar, are categorized by frequency. For this analysis, active sonar is categorized into four frequency ranges: low-frequency, mid-frequency, high-frequency, and very high-frequency.

- Low-frequency active sonar emits sounds at frequencies less than 1 kilohertz (kHz). Low-frequency active sonar is useful for detecting objects at great distances because low-frequency sounds do not dissipate as rapidly as higher frequency sounds.
- Mid-frequency active sonar emits sound at frequencies from 1 to 10 kHz. Mid-frequency active sonar is the Navy's primary tool for detecting and identifying submarines. Active sonar in this frequency range provides a valuable combination of range and target accuracy.

- High-frequency active sonar emits sound at frequencies greater than 10 kHz, up to 100 kHz. High-frequency sounds dissipate rapidly and have a small effective range; however, they provide higher resolution of objects and are useful at detecting and identifying smaller objects such as sea mines.
- Very high-frequency sources are those that operate above 100 kHz but below 200 kHz. These sources dissipate rapidly and have a small effective range, and may be used for such purposes as bottom mapping.

2.2.2 Ordnance/Munitions

Most ordnance and munitions used during training activities fall into three basic categories: projectiles, missiles, and bombs. Explosive ordnance can be further defined by net explosive weight (NEW), which is the actual weight in pounds of the explosive substance without the packaging, casings, bullets, etc. Net explosive weight is a measure of the strength of bombs and other explosives. For example, a 2,000-pound (lb.) (907.2-kilogram [kg]) bomb may have anywhere from 600 to 1,000 lb. (272.2 to 453.8 kg) of NEW.

Projectiles are fired during gunnery exercises from a variety of weapons, from pistols and rifles to large-caliber turret-mounted guns on the decks of military ships. Projectiles can be either explosive munitions (e.g., certain cannon shells) or non-explosive practice munitions (e.g., rifle/pistol bullets). Explosive rounds can be fused to either explode on impact or in the air (i.e., just prior to impact). Projectiles are broken down into three basic categories: small caliber (up to approximately 0.5 inch [in.]), medium caliber (greater than 0.5 in., up to approximately 2.24 in. in diameter), and large caliber (up to 5 in.).

Bombs are unpowered munitions dropped from aircraft on land and water targets. Bombs are in two categories: general-purpose bombs and subscale practice bombs. Similar to missiles, bombs are further classified according to the NEW of the bomb.

Signal Underwater Sound (SUS) Sonobuoys are also proposed for use in Naval at-sea training activities but do not fit into one of the above categories. These are mini sound-source seeker sonobuoys that use explosive charges as the active sound source instead of electrically produced sounds.

2.2.3 Other Systems and Expended Materials

Navy training activities may use other systems that are expended into the marine environment, such as torpedo accessories, parachutes, and targets, as a direct result of using these items for their intended purpose. In addition to these items, some accessory materials—related to the carriage or expenditure of these items—may also be released into the environment. These materials, referred to as military expended materials (MEM), are not recovered, and are analyzed as potential stressors on ESA-listed species. For detailed information on MEM used in the action

area, refer to Chapter 2 (Description of Proposed Action and Alternatives) of the 2011 Gulf of Alaska Final EIS/OEIS.

MEM analyzed in this opinion include, but are not limited to, the following systems and devices:

- **Sonobuoys.** Passive and active sonobuoys scuttle following completion of the training activity. The decelerator/parachute, which separates from the sonobuoy after water entry, may remain at the surface for a period of time, but is designed to eventually sink to the sea floor. Sonobuoys are not recovered in the open ocean environment.
- **Torpedo Launch Accessories.** Non-explosive torpedoes could be used in the action area. Non-explosive torpedoes are typically recovered for reuse and to evaluate performance. Expended materials such as decelerators/parachutes used with air-dropped torpedoes, guidance wires used with some submarine-launched torpedoes, and ballast weights used to recover non-explosive torpedoes would be expended. In addition to the materials described for non-explosive torpedoes, torpedo fragments would be expended into the environment when using explosive torpedoes.
- **Projectiles and Bombs.** Non-explosive projectiles and bombs and fragments from explosive projectiles and bombs would be expended during some training activities. These items consist primarily of lead (most small-caliber projectiles) or steel (medium- and large-caliber projectiles and all bombs).
- **Countermeasures.** Countermeasures (acoustic, chaff, flares) are expended during some training activities. Towed acoustic countermeasures are not expended.
- **Targets.** Some targets are designed to be expended; other targets, such as aerial drones and remote-controlled boats, are recovered for re-use. Targets struck with ordnance will release target fragments. There may be cases when these targets are not be recoverable and would be expended.

2.2.4 Classification of Non-impulsive and Impulsive Sources Analyzed

In this opinion, underwater sound is described as one of two types: impulsive and non-impulsive. Explosions and similar percussive events are sources of impulsive sounds. Sonar and other active acoustic systems are categorized as non-impulsive sound sources. In order to better organize and facilitate the analysis of approximately 300 individual sources of underwater acoustic sound or explosive energy, a series of source classifications, or source bins, was developed. The use of source classification bins provides the following benefits:

- Enables new acoustic sources and explosives sources that share the classification parameters of a bin to be covered under existing authorizations,
- Simplifies the collection of source utilization data and the reporting requirements anticipated under the ESA,

- Ensures a conservative approach to all impacts estimates, because all sources within a given bin are modeled using the parameters of the "loudest" source (i.e., the source with the lowest frequency, highest source level, longest duty cycle, or largest NEW within that bin),
- Allows analysis to be conducted in a more efficient manner (i.e., by bin instead of individual sound source), without any compromise of analytical results, and
- Provides a framework to support the reallocation of source usage (i.e., hours or number of explosives) between different source bins, as long as the total number of exposure estimates remains within the overall analyzed and authorized limits. Building in this flexibility supports evolving training requirements, which are linked to potentially unpredictable real world events.

Impulsive sources (i.e., explosives) are grouped into bins based on the NEW of the explosive device. Non-impulsive sources (e.g., sonar) are grouped into bins based on the sound source frequency, source level, and the application in which the source would be used.

The following factors further describe the considerations associated with the development of non-impulsive source bins:

- Frequency of the non-impulsive source:
- Low-frequency sources operate below 1 kHz
- Mid-frequency sources operate at and above 1 kHz, up to and including 10 kHz
- High-frequency sources operate above 10 kHz, up to and including 100 kHz
- Very high-frequency sources operate above 100 kHz but below 200 kHz

Source level of the non-impulsive source (i.e., the sound pressure level [SPL] produced at a distance of 1 m from a sound source):

- Greater than 160 decibels (dB), but less than 180 dB
- Equal to 180 dB and up to 200 dB
- Greater than 200 dB

Application in which the source would be used:

- How a sensor is employed supports how the sensor's acoustic emissions are analyzed

- Factors considered include pulse length (time source is on); beam pattern (whether sound is emitted as a narrow, focused beam or, as with most explosives, in all directions); and duty cycle (how often or how many times a transmission occurs in a given time period during an event)

There are non-impulsive sources of low source level, narrow beam width, downward directed transmission, short pulse lengths, frequencies beyond known hearing ranges of marine mammals, or some combination of these factors that are not anticipated to result in takes of protected species and therefore were not modeled. These sources are qualitatively analyzed because they have met the following criteria:

- Acoustic sources with frequencies greater than 200 kHz
- Sources with source levels less than 160 dB

Further discussion of these sources is included in section 6.2.2 of this opinion.

2.2.5 Source Classes Analyzed for Training Activities

Table 3 and Table 4 show the impulsive sources (e.g., explosives) and non-impulsive sources (e.g., sonar) associated with Navy training activities in the TMAA.

Table 3. Impulsive Training Sources Classes Analyzed

Source Class	Representative Munitions	Net Explosive Weight[1] (lb.)
E5	5 in. projectiles	> 5–10
E9	500 lb. bomb	> 100–250
E10	1,000 lb. bomb	> 250–500
E12	2,000 lb. bomb	> 650–1,000

[1] Net Explosive Weight refers to the amount of explosives; the actual weight of a munition may be larger due to other components such as the casing for a bomb, missile, projectile, or device.

Notes: in. = inches, lb. = pounds

Table 4. Non-impulsive Training Sources Classes Analyzed

Source Class Category	Source Class	Description of Representative Source
Mid-Frequency (MF): Tactical and non-tactical sources that produce mid-frequency (1–10 kHz) signals	MF1	Hull-mounted surface ship sonar (e.g., AN/SQS-53C and AN/SQS-61)
	MF3	Hull-mounted submarine sonar (e.g., AN/BQQ-10)
	MF4	Helicopter-deployed dipping sonar (e.g., AN/AQS-22 and AN/AQS-13)
	MF5	Active acoustic sonobuoys (e.g., DICASS)
	MF6	Active underwater sound signal devices (e.g., MK-84)
	MF11	Hull-mounted surface ship sonar with an active duty cycle greater than 80%
High-Frequency (HF): Tactical and non-tactical sources that produce high-frequency (greater than 10 kHz but less than 180 kHz) signals	HF1	Hull-mounted submarine sonar (e.g., AN/BQQ-10)
	HF6	Active sources (equal to 180 dB and up to 200 dB) not otherwise binned
Anti-Submarine Warfare (ASW): Tactical sources such as active sonobuoys and acoustic countermeasures systems used during the conduct of ASW training activities	ASW2	Mid-frequency Multistatic Active Coherent sonobuoy (e.g., AN/SSQ-125)
	ASW3	Mid-frequency towed active acoustic countermeasure systems (e.g., AN/SLQ-25)
	ASW4	Mid-frequency expendable active acoustic device countermeasures (e.g., MK-3)

Notes: dB = decibels, DICASS = Directional Command Activated Sonobuoy System, kHz = kilohertz

2.3 Navy Mitigation Measures to Minimize or Avoid Exposure to Stressors

Mitigation measures proposed for the action area are presented in Table 6 and discussed in detail in Chapter 5 (Standard Operating Procedures, Mitigation, and Monitoring) of the Gulf of Alaska Draft and Final Supplemental EIS/ OEIS. Mitigation measures can be grouped into three categories: Mitigation Zone Procedural Measures, Lookout Procedural Measures, and Area and Activity Specific Mitigation Measures. For purposes of activities in the TMAA, all vessels under the Navy's operational control will comply with the mitigation measures described below.

2.3.1 Mitigation Zone Procedural Measures

A mitigation zone is designed solely for the purpose of reducing potential impacts on marine mammals and sea turtles from training activities. Mitigation zones are measured as the radius from a source. Unique to each activity category, each radius represents a distance that the Navy will visually observe to help reduce injury to marine species. Visual detections of applicable marine species will be communicated immediately to the appropriate watch station for information dissemination and appropriate action. If the presence of marine mammals is detected acoustically, Lookouts posted in aircraft and on surface vessels will increase the vigilance of their visual surveillance (see section 2.3.2 for additional information on Lookout Procedural Measures). As a reference, aerial surveys are typically made by flying at 1,500 feet (460 meters) altitude or lower at the slowest safe speed.

Many of the proposed activities have mitigation measures that have been implemented since 2011, as required by previous environmental documents or consultations. Most of the 2011 Gulf of Alaska Final EIS/OEIS mitigation zones for activities that involve the use of impulsive and non-impulsive sources were originally designed to reduce the potential for onset of Temporary Threshold Shifts (TTS). For this consultation, the Navy updated the acoustic propagation modeling to incorporate updated hearing threshold metrics (i.e., upper and lower frequency limits), updated density data for marine mammals, and factors such as an animal's likely presence at various depths. An explanation of the acoustic propagation modeling process can be found in the Determination of Acoustic Effects on Marine Mammals for the Gulf of Alaska Navy Training Activities Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement acoustic primer technical report (Navy 2014a). Additionally, since publication of the proposed MMPA rule, the Navy re-evaluated the range to effects in consideration of the acoustic thresholds in NMFS' new Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing, which resulted in larger ranges for some explosive sources (for more information on NMFS' and the Navy's consideration of the new Technical Guidance, see section 3.1.4).

As a result of the updates described above, in some cases the ranges to onset of TTS are much larger than those estimated by previous Phase I models. Because lookouts are not able to effectively observe marine mammals at those farther ranges with the naked eye, or even with Big Eye Binoculars, as well as the unacceptable operational impacts associated with mitigating these large areas, the Navy is unable to mitigate for onset of TTS for every activity (see Chapter 5 of 2016 GOA SEIS/OEIS for further discussion). In this analysis, the Navy developed each recommended mitigation zone to avoid or reduce the potential for onset of the lowest level of injury (i.e., Permanent Threshold Shift [PTS] since TTS is not considered an injury), out to the predicted maximum range. In some cases where the ranges to effects are smaller than previous models estimated, the mitigation zones were adjusted accordingly to provide consistency across the measures. Mitigating to the predicted maximum range to PTS consequently also mitigates to the predicted maximum range to onset mortality (1 percent mortality), onset slight lung injury, and onset slight gastrointestinal tract injury, since the maximum range to effects for these criteria are shorter than for PTS. Furthermore, in many cases, the predicted maximum range to PTS also consequently covers the predicted average range to TTS. Table 5 summarizes the predicted average range to TTS, average range to PTS, maximum range to PTS, and recommended mitigation zone for each activity category, based on the Navy's acoustic propagation modeling results and updated by consideration of NMFS' new Technical Guidance. The predicted ranges are based on local environmental conditions and are unique to the TMAA.

The activity-specific mitigation zones are based on the longest range for all the functional hearing groups. The mitigation zone for the majority of activities is driven by either the high-frequency cetaceans or the sea turtle functional hearing groups. Therefore, the mitigation zones are even more protective for the remaining function hearing groups (i.e., low-frequency

cetaceans, mid-frequency cetaceans, and pinnipeds), and likely cover a larger portion of the potential range to onset of TTS. In some instances, the Navy recommends mitigation zones that are larger or smaller than the predicted maximum range to PTS based on the effectiveness and operational assessments. The recommended mitigation zones and their associated assessments are provided throughout the remainder of this section. The recommended measures are either currently implemented, are modifications of current measures, or are new measures.

For some activities specified throughout the remainder of this section, Lookouts are required to observe for concentrations of detached floating vegetation (kelp paddies), which are indicators of potential marine mammal and sea turtle presence within the mitigation zone. Those specified activities will not commence if floating vegetation is observed within the mitigation zone prior to the initial start of the activity. If floating vegetation is observed prior to the initial start of the activity, the activity will be relocated to an area where no floating vegetation is observed. Training will not cease as a result of indicators (i.e., floating vegetation) entering the mitigation zone after activities have commenced. This measure is intended only for floating vegetation detached from the seafloor.

Table 5. Predicted Range to Effects and Recommended Mitigation Zones

Activity Category	Representative Source (Bin)[1]	Predicted (Longest) Average Range to TTS	Predicted (Longest) Average Range to PTS	Predicted Maximum Range to PTS	Recommended Mitigation Zone
Non-Impulse Sound					
Hull-Mounted Mid-Frequency Active Sonar	SQS-53 ASW hull-mounted sonar (MF1)	3,821 yd. (3.5 km) for one ping	100 yd. (91 m) for one ping	Not Applicable	6 dB power down at 1,000 yd. (914 m); 4 dB power down at 500 yd. (457 m); and shutdown at 200 yd. (183 m)
Non-Impulse Sound (continued)					
High-Frequency and Non-Hull Mounted Mid-Frequency Active Sonar	AQS-22 ASW dipping sonar (MF4)	230 yd. (210 m) for one ping	20 yd. (18 m) for one ping	Not applicable	200 yd. (183 m)
Explosive and Impulse Sound					
Signal Underwater Sound (SUS) buoys using 0.6–2.5 lb. NEW	Explosive sonobuoy (E3)	290 yd. (265 m)	113 yd. (103 m)	309 yd. (283 m)	350 yd. (320 m)
Gunnery Exercises – Small- and Medium-Caliber (Surface Target)	40 mm projectile (E2)	190 yd. (174 m)	83 yd. (76 m)	182 yd. (167 m)	200 yd. (183 m)
Gunnery Exercises – Large-Caliber (Surface Target)[3]	5 in. projectiles (E5 at the surface)	771 yd. (705 m)	327 yd. (299 m)	327 yd. (299 m)	600 yd. (549 m)

Activity Category	Representative Source (Bin)[1]	Predicted (Longest) Average Range to TTS	Predicted (Longest) Average Range to PTS	Predicted Maximum Range to PTS	Recommended Mitigation Zone
Bombing Exercises[4]	MK-84 2,000 lb. bomb (E12)	5,430 yd. (4.97 km)	1,772 yd. (1.62 km)	1,851 yd. (1.69 km)	2,500 yd. (2.3 km)[2]

[1] This table does not provide an inclusive list of all sources in a given bin; bins presented here represent the source bin with the largest range to effects within the given activity category.

[2] Recommended mitigation zones are larger than the modeled injury zones to account for multiple types of sources or charges being used.

[3] The representative source bin E5 has different range to effects depending on the depth of activity occurrence (at the surface or at various depths). Bin E5 TTS value reflects correct GOA-specific value for average TTS (Table 3.8-18 of the GOA FSEIS/OEIS). PTS re-assessed using NOAA's August 2016 revised explosive acoustic criteria applicable to the most sensitive functional hearing group. PTS value for bin E5 was lower than previously modeled range, so TTS not re-calculated and TTS value from previous model shown as conservative (over predictive) value. Lower weight bins re-assessed similarly did not result in any values larger than existing values shown.

[4] Bin E12 PTS and TTS re-assessed using NOAA's August 2016 revised explosive acoustic criteria applicable to the most sensitive functional hearing group.

Notes: ASW = Anti-submarine Warfare, dB = decibels, km = Kilometers, lb. = Pounds, m = Meters, mm = millimeters, NEW = Net Explosive Weight, PTS = Permanent Threshold Shift, TTS = Temporary Threshold Shift, yd. = yards

2.3.2 Lookout Procedural Measures

The use of Lookouts is a critical component of the U.S. Navy procedural measures and implementation of mitigation zones (detailed above). Lookouts are highly qualified and experienced observers of the marine environment. Their duties require that they report all objects sighted in the water (e.g., trash, a periscope, marine mammals, sea turtles) to the Officer of the Deck and all disturbances (e.g., surface disturbance, discoloration) that may be indicative of a threat to the vessel and its crew. There are personnel standing watch on station at all times (day and night) when a ship or surfaced submarine is moving through the water. Ships have personnel assigned to stand watch at all times while underway. Watch personnel may perform watch duties in conjunction with lookout responsibilities that extend beyond looking at the water or air (such as supervision of other personnel).

The Navy will have two types of Lookouts for the purposes of conducting visual observations: those positioned on ships; and those positioned in aircraft, or on small boats. Lookouts positioned on ships will diligently observe the air and surface of the water. They will have multiple observation objectives, which include but are not limited to detecting the presence of biological resources and recreational or fishing boats, observing the mitigation zones described in Table 6, and monitoring for vessel and personnel safety concerns.

Due to manning and space restrictions on aircraft, small boats, and some Navy ships, Lookouts for these platforms may be supplemented by the aircraft crew or pilot, boat crew, or range site personnel. Lookouts positioned in minimally manned platforms may be responsible for tasks in addition to observing the air or surface of the water (e.g., navigation of a helicopter or small boat). However, all Lookouts will, considering personnel safety, practicality of implementation,

and impact on the effectiveness of the activity, comply with the observation objectives described above for Lookouts positioned on ships.

The procedural measures implemented primarily consist of having Lookouts during specific training activities. For example, during bombing exercises, the Navy will have one Lookout positioned in the aircraft conducting the exercise, and trained lookouts in any surface vessels involved. The Lookout in the aircraft visually survey the target and buffer zone for marine mammals prior to and during the exercise. The survey of the impact area is made by flying at 1,500 feet or lower, if safe to do so and at the slowest safe speed. Release of ordnance through cloud cover is prohibited as aircraft must be able to observe ordnance impact areas. Additional details for each specific activity type is detailed in Section 5.3.1 (Lookout Procedural Measures) of the Gulf of Alaska Final Supplemental EIS/OEIS. The number of lookouts corresponds to the number of lookouts present during each activity. Lookouts may be present on aircraft or vessels participating in the activity.

Table 6. U.S. Navy Mitigation Measures

Activity Category or Mitigation Area	Proposed Lookout Procedural Measure	Proposed Mitigation Zone and Protection Focus
Specialized Training		
Marine Species Awareness Training (Modules 1 through 4)	Applicable personnel will complete the U.S. Navy Marine Species Awareness Training prior to standing watch or serving as a Lookout.	The mitigation zones observed by Lookouts are specified for each Mitigation Zone Procedural Measure below.
Acoustic Stressors – Sonar and Other Active Acoustic Sources		
Low-Frequency and Hull-Mounted Mid-Frequency Active Sonar during Anti-Submarine Warfare	2 Lookouts (general) 1 Lookout (minimally manned, moored, or anchored)	1,000 yd. (914 m) and 500 yd. (457 m) power downs and 200 yd. (183 m) shutdown for cetaceans and sea turtles (excludes bow-riding dolphins)
High-Frequency and Non-Hull Mounted Mid-Frequency Active Sonar	2 Lookouts (general) 1 Lookout (minimally manned, moored, or anchored)	200 yd. (183 m) for marine mammals and concentrations of floating vegetation.
Explosive and Impulsive Sound		
Explosive Signal Underwater Sound buoys using 0.6–2.5 lb. NEW	1 Lookout	350 yd. (320 m) for marine mammals, sea turtles, and concentrations of floating vegetation.
Gunnery Exercises – Small- and Medium-Caliber Using a Surface Target	1 Lookout	200 yd. (183 m) for marine mammals, sea turtles, and concentrations of floating vegetation.
Gunnery Exercises – Large-Caliber Explosive Rounds using a Surface Target	1 Lookout	600 yd. (549 m) for marine mammals, sea turtles, and concentrations of floating vegetation.
Explosive and Non-Explosive Bombing Exercises	1 Lookout	Explosive: 2,500 yd. (2.3 km) for marine mammals, sea turtles, and concentrations of floating vegetation. Non-Explosive: 1,000 yd. (914 m) for marine mammals, sea turtles, and concentrations of floating vegetation.
Activity Category or Mitigation Area	**Proposed Lookout Procedural Measure**	**Proposed Mitigation Zone and Protection Focus**
Explosive and Impulsive Sound (continued)		
Weapons Firing Noise During Gunnery Exercises – Large-Caliber	1 Lookout	70 yd. (60 m) within 30 degrees on either side of the gun target line on the firing side for marine mammals, sea turtles, and concentrations of floating vegetation.

Activity Category or Mitigation Area	Proposed Lookout Procedural Measure	Proposed Mitigation Zone and Protection Focus
Physical disturbance		
Vessel Movements	1 Lookout	500 yd. (457 m) for whales. 200 yd. (183 m) for all other marine mammals (except bow riding dolphins).
Towed In-Water Device Use	1 Lookout	250 yd. (229 m) for marine mammals

Notes: EIS = Environmental Impact Statement, km = kilometer, lb.= pound, m = meter, NEW = net explosive weight, nm = nautical mile, OEIS = Overseas Environmental Impact Statement, TMAA = Temporary Maritime Activities Area, yd.= yard

2.3.3 Area and Activity Specific Mitigation Measures in the TMAA

In addition to the procedural mitigation measures explained in Sections 2.3.1 and 2.3.1, the following subsections present the area and activity specific mitigation measures that the Navy will operate under in the Gulf of Alaska.

2.3.3.1 *Portlock Bank*

The use of explosives will not occur in the Portlock Bank area (Figure 1). Portlock Bank is located offshore to the east of Kodiak Island and partially overlaps with the far western portion of the TMAA. This prevents any training in the Portlock Bank area involving use of improved extended echo ranging sonobuoys, explosive signal underwater sound buoys, gunnery expercise using explosive projectiles, and explosive bombing exercises.

2.3.3.2 *North Pacific Right Whale Cautionary Area*

The Navy will not use surface ship hull mounted mid-frequency sonar or explosives during training within the portion of the NMFS-identified North Pacific right whale feeding area overlapping the TMAA in the June to September timeframe (Figure 1). The Navy reserves the right to use surface ship hull mounted mid-frequency sonar or explosives in this area in the event of national security needs. Approval from the Commander, U.S. Third Fleet is required prior to using surface ship hull mouted mid-frequency sonar or explosives in this area.

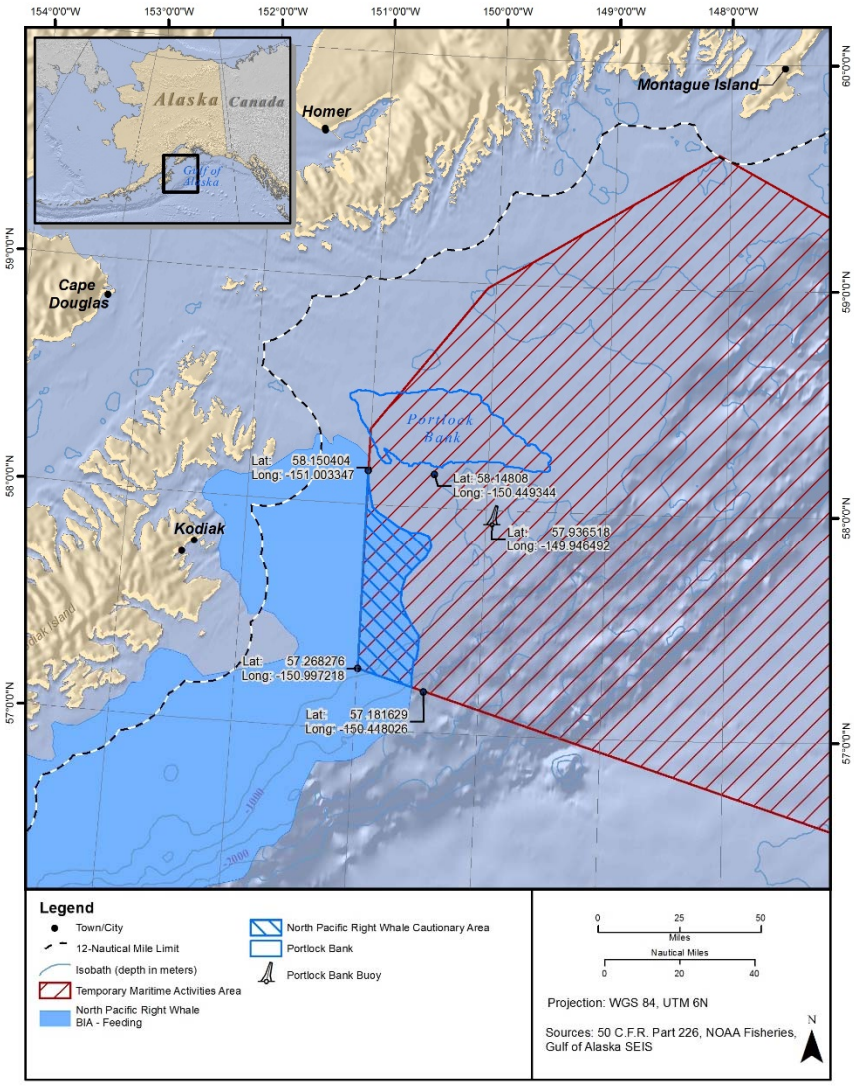


Figure 1. Depiction of Portlock Bank and the North Pacific right whale feeding area overlapping the Gulf of Alaska TMAA.

2.4 Approach to Monitoring and Reporting

The Navy is committed to avoiding and reducing impacts of the proposed action through mitigation. The Navy will undertake monitoring efforts to track compliance with take authorizations, help evaluate the effectiveness of implemented mitigation measures, and gain a better understanding of the effects of the proposed action on marine resources. Taken together, mitigation and monitoring comprise the Navy's integrated approach for reducing environmental impacts from the proposed action. The Navy's overall monitoring approach will seek to leverage and build on existing research efforts whenever possible.

Consistent with the cooperating agency agreement with NMFS, mitigation and monitoring measures presented in this opinion focus on the requirements for protection and management of

marine resources. A well-designed monitoring program can provide important feedback for validating assumptions made in analyses and allow for adaptive management of marine resources. Since monitoring will be required for compliance with the Final Rule issued for the proposed action under the MMPA, details of the monitoring program will be developed in coordination with NMFS through the regulatory process.

2.5 NMFS' Promulgation of Regulations Pursuant to the Marine Mammal Protection Act

Under the MMPA, the Navy may obtain authorization to "take" marine mammals only if the "take" occurs incidental to training activities within the Gulf of Alaska TMAA. In order to authorize incidental take under the MMPA, NMFS must determine that the incidental taking of marine mammals will have a negligible impact on the species or stock(s) and will not have an unmitigable adverse impact on the availability of the species or stock(s) for subsistence uses (where relevant). NMFS has defined negligible impact in 50 CFR 216.103 as "an impact resulting from the specified activity that cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species or stock through effects on annual rates of recruitment or survival."

NMFS Permits Division determined that the Navy's proposed action (summarized above) would result in the take of ESA-listed species and that such take would be in the form of exposure to sound or pressure waves in the water. The specific activity and geographic region where take may occur, the dates when take may occur, and permissible method of taking that are set by the proposed regulations are all consistent with the Navy's action described previously in this opinion so they will not be repeated here.

2.5.1 Taking and Importing Marine Mammals; U.S. Navy's Gulf of Alaska TMAA

The take of ESA-listed species by harassment incidental to the Navy's training activities in the Gulf of Alaska TMAA authorized pursuant to NMFS Permit Division's proposed MMPA rule is presented in the following sections.

2.5.1.1 *§ 218.150 Specified activity and specified geographical region.*

(a) Regulations in this subpart apply only to the U.S. Navy for the taking of marine mammals that occurs in the area outlined in paragraph (b) of this section and that occurs incidental to the activities described in paragraph (c) of this section.

(b) The taking of marine mammals by the Navy is only authorized if it occurs within the GOA TMAA Study Area, which is bounded by a hexagon with the following six corners: 57°30′°N. lat., 141°30′°W. long.; 59°36′°N. lat., 148°10′°W. long.; 58°57′°N. lat., 150°04′°W. long.; 58°20′°N. lat., 151°00′°W. long.; 57°16′°N. lat., 151°00′°W. long.; and 55°30′°N. lat., 142°00′°W. long.

(c) The taking of marine mammals by the Navy is only authorized if it occurs incidental to the following activities:

(1) Sonar and other Active Sources Used During Training:

(i) Mid-frequency (MF) Source Classes:

(A) MF1 – an average of 271 hours per year.

(B) MF3 – an average of 24 hours per year.

(C) MF4 – an average of 26 hours per year.

(D) MF5 – an average of 126 items per year.

(E) MF6 – an average of 11 items per year.

(F) MF11 – an average of 39 hours per year.

(ii) High-frequency (HF) Source Classes:

(A) HF1 – an average of 12 hours per year.

(B) HF6 – an average of 40 items per year.

(iii) Anti-Submarine Warfare (ASW) Source Classes:

(A) ASW2 – an average of 40 hours per year.

(B) ASW3 – an average of 273 hours per year.

(C) ASW4 – an average 6 items per year.

(iv) Torpedoes (TORP):

TORP2 – an average of 0 items per year.

[Reserved]

(2) Impulsive Source Detonations During Training:

(i) Explosive Classes:

(A) E5 (>5 to 10 pound (lb) net explosive weight (NEW)) – an average of 56 detonations per year.

(B) E9 (>100 to 250 lb NEW) – an average of 64 detonations per year.

(C) E10 (>250 to 500 lb NEW) – an average of 6 detonations per year.

(D) E12 (>650 to 1,000 lb NEW) – an average of 2 detonations per year.

(ii) [Reserved]

2.5.1.2 ***§ 218.151 Effective dates and definitions.***

(a) Regulations in this subpart are effective [insert date of filing in the FEDERAL REGISTER], through [insert date 5 years after date of filing in the FEDERAL REGISTER].

(b) The following definitions are utilized in these regulations:

(1) Uncommon Stranding Event (USE) – A stranding event that takes place during a Major Training Exercise (MTE) and involves any one of the following:

(i) Two or more individuals of <u>any</u> cetacean species (*i.e.,* could be two different species, but not including mother/calf pairs, unless of species of concern listed in next bullet) found dead or live on shore within a three- to four-day period and within 10 miles of one another.

(ii) A single individual or mother/calf pair of any of the following marine mammals of concern: beaked whale of any species, North Pacific right whale, humpback whale, sperm whale, blue whale, fin whale, sei whale, Cook Inlet beluga whale, Northern fur seal, and Steller sea lion.

(iii) A group of two or more cetaceans of any species exhibiting indicators of distress.

(2) [Reserved]

2.5.1.3 ***§ 218.152 Permissible methods of taking.***

(a) Under letter of authorization (LOA) issued pursuant to §§ 216.106 and 218.157 of this chapter, the holder of the LOA may incidentally, but not intentionally, take marine mammals within the area described in § 218.150, provided the activity is in compliance with all terms, conditions, and requirements of these regulations and the LOA.

(b) The activities identified in § 218.150(c) must be conducted in a manner that minimizes, to the greatest extent practicable, any adverse impacts on marine mammal species or stocks and their habitat.

(c) The incidental take of marine mammals under the activities identified in § 218.150(c) is limited to the following species, by the identified method of take and the indicated number of times:

(1) Level B Harassment for all Training Activities:

(i) Mysticetes:

(A) Blue whale (Balaenoptera musculus), Eastern North Pacific – 235 (an average of 47 per year).

(B) Fin whale (Balaenoptera physalus), Northeast Pacific – 6,455 (an average of 1,291 per year).

(C) Humpback whale (Megaptera novaeangliae), Central North Pacific – 305 (an average of 61 per year).

(D) Humpback whale (Megaptera novaeangliae), Western North Pacific – 5 (an average of 1 per year).

(E) Humpback whale (Megaptera novaeangliae), CA/OR/WA – 35 (an average of 7 per year).

(F) Minke whale (Balaenoptera acutorostrata), Alaska – 215 (an average of 43 per year).

(G) North Pacific right whale (Eubalaena japonica), Eastern North Pacific – 15 (an average of 3 per year).

(H) Sei whale (Balaenoptera borealis), Eastern North Pacific – 30 (an average of 6 per year).

(ii) Odontocetes:

(A) Baird's beaked whale (Berardius bairdii), Alaska – 1,000 (an average of 200 per year).

(B) Cuvier's beaked whale (Ziphius cavirostris), Alaska – 6,355 (an average of 1,271 per year).

(C) Dall's porpoise (Phocoenoidea dalli), Alaska – 41,350 (an average of 8, 270 per year).

(D) Harbor porpoise (Phocoena phocoena), GOA – 13,710 (an average of 2,742 per year).

(E) Harbor porpoise (Phocoena phocoena), Southeast Alaska – 4,815 (an average of 963 per year).

(F) Killer whale (Orcinus orca), Alaska Resident – 1,405 (an average of 281 per year).

(G) Killer whale (Orcinus orca), Eastern North Pacific Offshore – 130 (an average of 26 per year).

(H) Killer whale (Orcinus orca), GOA, Aleutian Island, and Bearing Sea Transient – 360 (an average of 72 per year).

(I) Pacific white-sided dolphin (Lagenorhynchus obliquidens), North Pacific – 4,905 (an average of 981 per year).

(J) Stejneger's beaked whale (Mesoplodon stejnegeri), Alaska – 2,880 (an average of 576 per year).

(K) Sperm whale (Physeter macrocephalus), North Pacific – 490 (an average of 98 per year).

(iii) Pinnipeds:

(A) California sea lion (Zalophus californianus), U.S. – 10 (an average of 2 per year).

(B) Steller sea lion (Eumetopias jubatus), Eastern U.S. – 1,675 (an average of 335 per year).

(C) Steller sea lion (Eumetopias jubatus), Western U.S. – 1,430 (an average of 286 per year).

(D) Harbor seal (Phoca vitulina), South Kodiak – 5 (an average of 1 per year).

(E) Harbor seal (Phoca vitulina), Prince William Sound – 5 (an average of 1 per year).

(F) Northern elephant seal (Mirounga angustirostris), California Breeding – 610 (an average of 122 per year).

(G) Northern fur seal (Callorhinus ursinus), Eastern Pacific – 3,565 (an average of 713 per year).

(2) Level A Harassment for all Training Activities:

(i) Odontocetes:

Dall's porpoise (Phocoenoidea dalli), Alaska – 12 (an average of 4 per year).

[Reserved]

(ii) [Reserved]

2.5.1.4 ***§ 218.153 Prohibitions.***

Notwithstanding takings contemplated in § 218.152 and authorized by an LOA issued under §§ 216.106 and 218.157 of this chapter, no person in connection with the activities described in § 218.150 may:

(a) Take any marine mammal not specified in § 218.152(c);

(b) Take any marine mammal specified in § 218.152(c) other than by incidental take as specified in § 218.152(c);

(c) Take a marine mammal specified in § 218.152(c) if such taking results in more than a negligible impact on the species or stocks of such marine mammal; or

(d) Violate, or fail to comply with, the terms, conditions, and requirements of these regulations or an LOA issued under §§ 216.106 and 218.157 of this chapter.

2.5.1.5 ***§ 218.154 Mitigation.***

(a) After review of best available science, the following mitigation was determined to result in the least practicable adverse effect on marine mammal species or stocks. When conducting training activities, as identified in § 218.150, the mitigation measures contained in the LOA issued under §§ 216.106 and 218.157 of this chapter must be implemented. These mitigation measures include, but are not limited to:

(1) *Lookouts*. The Navy shall have two types of lookouts for the purposes of conducting visual observations: those positioned on ships; and those positioned ashore, in aircraft, or on boats. The following are protective measures concerning the use of lookouts.

(i) Lookouts positioned on surface ships shall be dedicated solely to diligent observation of the air and surface of the water. Their observation objectives shall include, but are not limited to, detecting the presence of biological resources and recreational or fishing boats, observing mitigation zones, and monitoring for vessel and personnel safety concerns.

(ii) Due to manning and space restrictions on aircraft, small boats, and some Navy ships, lookouts for these platforms may be supplemented by the aircraft crew or pilot, boat crew, range site personnel, or shore-side personnel. Lookouts positioned in minimally manned platforms may be responsible for tasks in addition to observing the air or surface of the water (*e.g.*,

navigation of a helicopter or small boat). However, all lookouts shall, considering personnel safety, practicality of implementation, and impact on the effectiveness of the activity, comply with the observation objectives described above for lookouts positioned on ships.

(iii) All personnel standing watch on the bridge, Commanding Officers, Executive Officers, maritime patrol aircraft aircrews, anti-submarine warfare helicopter crews, civilian equivalents, and lookouts shall successfully complete the United States Navy Marine Species Awareness Training prior to standing watch or serving as a lookout.

(iv) Lookout measures for non-impulsive sound:

(A) With the exception of vessels less than 65 ft (20 m) in length, ships using hull-mounted mid-frequency active sonar sources associated with anti-submarine warfare activities at sea shall have two Lookouts at the forward position of the vessel.

(B) While using hull-mounted mid-frequency active sonar sources associated with anti-submarine warfare activities at sea, vessels less than 65 ft (20 m) in length shall have one lookout at the forward position of the vessel due to space and manning restrictions.

(C) During non-hull mounted mid-frequency active sonar training activities, Navy aircraft participating in exercises at sea shall conduct and maintain, when operationally feasible and safe, surveillance for marine species of concern as long as it does not violate safety constraints or interfere with the accomplishment of primary operational duties. Helicopters shall observe/survey the vicinity of an anti-submarine warfare training event for 10 minutes before the first deployment of active (dipping) sonar in the water.

(D) Ships or aircraft conducting non-hull-mounted mid-frequency active sonar, such as helicopter dipping sonar systems, shall maintain one lookout.

(E) Ships conducting high-frequency active sonar shall maintain one lookout.

(v) Lookout measures for explosives and impulsive sound:

(A) Aircraft conducting explosive signal underwater sound buoy activities using >0.5–2.5 lb. net explosive weight (NEW) shall have one lookout.

(B) Surface vessels or aircraft conducting small-, medium-, or large-caliber gunnery exercises against a surface target shall have one Lookout. From the intended firing position, trained Lookouts shall survey the mitigation zone for marine mammals prior to commencement and during the exercise as long as practicable. Towing vessels, if applicable, shall also maintain one Lookout. If a marine mammal is sighted in the vicinity of the exercise, the tow vessel shall immediately notify the firing vessel in order to secure gunnery firing until the area is clear.

(C) Aircraft conducting explosive bombing exercises shall have one lookout and any surface vessels involved shall have trained Lookouts. If surface vessels are involved, Lookouts shall survey for floating kelp and marine mammals. Aircraft shall visually survey the target and buffer zone for marine mammals prior to and during the exercise. The survey of the impact area shall be made by flying at 1,500 ft. (460 m) or lower, if safe to do so, and at the slowest safe speed. Release of ordnance through cloud cover is prohibited: aircraft must be able to actually

see ordnance impact areas. Survey aircraft should employ most effective search tactics and capabilities.

(D) When aircraft are conducting missile exercises against a surface target, the Navy shall have one Lookout positioned in an aircraft. Aircraft shall visually survey the target area for marine mammals. Visual inspection of the target area shall be made by flying at 1,500 ft. (457 m) or lower, if safe to do so, and at the slowest safe speed. Firing or range clearance aircraft must be able to actually see ordnance impact areas.

(E) Ships conducting explosive and non-explosive gunnery exercises shall have one Lookout on the ship. This may be the same lookout described in paragraph (B) above for surface vessels conducting small-, medium-, or large-caliber gunnery exercises when that activity is conducted from a ship against a surface target.

(vi) Lookout measures for physical strike and disturbance:

While underway, surface ships shall have at least one Lookout with binoculars, and surfaced submarines shall have at least one Lookout with binoculars. Lookouts already posted for safety of navigation and man-overboard precautions may be used to fill this requirement. As part of their regular duties, Lookouts will watch for and report to the Officer of the Deck the presence of marine mammals.

[Reserved]

(vii) Lookout measures for non-explosive practice munitions:

(A) Gunnery exercises using non-explosive practice munitions (*e.g.*, small-, medium-, and large-caliber) using a surface target shall have one Lookout.

(B) During non-explosive bombing exercises one Lookout shall be positioned in an aircraft and trained lookouts shall be positioned in any surface vessels involved.

(C) When aircraft are conducting non-explosive missile exercises (including exercises using rockets) against a surface target, the Navy shall have one Lookout positioned in an aircraft.

(2) Mitigation Zones – The following are protective measures concerning the implementation of mitigation zones.

(i) Mitigation zones shall be measured as the radius from a source and represent a distance to be monitored.

(ii) Visual detections of marine mammals or sea turtles within a mitigation zone shall be communicated immediately to a watch station for information dissemination and appropriate action.

(iii) Mitigation zones for non-impulsive sound:

(A) The Navy shall ensure that hull-mounted mid-frequency active sonar transmission levels are limited to at least 6 dB below normal operating levels if any detected marine mammals or sea turtles are within 1,000 yd. (914 m) of the sonar dome (the bow).

(B) The Navy shall ensure that hull-mounted mid-frequency active sonar transmissions are limited to at least 10 dB below the equipment's normal operating level if any detected marine mammals or sea turtles are within 500 yd. (457 m) of the sonar dome.

(C) The Navy shall ensure that hull-mounted mid-frequency active sonar transmissions are ceased if any detected cetaceans or sea turtles are within 200 yd. (183 m) and pinnipeds are within 100 yd. (90 m) of the sonar dome. Transmissions shall not resume until the marine mammal has been observed exiting the mitigation zone, is thought to have exited the mitigation zone based on its course and speed, has not been detected for 30 minutes, the vessel has transited more than 2,000 yd. (1830 m) beyond the location of the last detection, or the ship concludes that dolphins are deliberately closing in on the ship to ride the ship's bow wave (and there are no other marine mammal sightings within the mitigation zone). Active transmission may resume when dolphins are bow riding because they are out of the main transmission axis of the active sonar while in the shallow-wave area of the ship bow.

(D) The Navy shall ensure that high-frequency and non-hull-mounted mid-frequency active sonar transmission levels are ceased if any detected cetaceans are within 200 yd. (183 m) and pinnipeds are within 100 yd. (90 m) of the source. Transmissions shall not resume until the marine mammal has been observed exiting the mitigation zone, is thought to have exited the mitigation zone based on its course and speed, the mitigation zone has been clear from any additional sightings for a period of 10 minutes for an aircraft-deployed source, the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a vessel-deployed source, the vessel or aircraft has repositioned itself more than 400 yd. (370 m) away from the location of the last sighting, or the vessel concludes that dolphins are deliberately closing in to ride the vessel's bow wave (and there are no other marine mammal sightings within the mitigation zone).

(iv) Mitigation zones for explosive and impulsive sound:

(A) A mitigation zone with a radius of 350 yd. (320 m) shall be established for explosive signal underwater sonobuoys using >0.5 to 2.5 lb NEW. Explosive signal underwater sonobuoys shall not be deployed if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone (around the intended deployment location). Explosive signal underwater sonobuoy deployment shall cease if a marine mammal is sighted within the mitigation zone. Detonations shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, or the mitigation zone has been clear from any additional sightings for a period of 10 minutes. Passive acoustic monitoring shall also be conducted with Navy assets, such as sonobuoys, already participating in the activity. These assets would only detect vocalizing marine mammals within the frequency bands monitored by Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot provide locations of these animals. Passive acoustic detections would be reported to Lookouts posted in aircraft in order to increase vigilance of their visual surveillance.

(B) A mitigation zone with a radius of 200 yd. (183 m) shall be established for small- and medium-caliber gunnery exercises with a surface target. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing shall cease if a marine mammal is sighted within the mitigation zone. Firing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, the mitigation zone has been clear from any additional sightings for a period of 10 minutes for a firing aircraft, the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a firing ship, or the intended target location has been repositioned more than 400 yd. (370 m) away from the location of the last sighting.

(C) A mitigation zone with a radius of 600 yd. (549 m) shall be established for large-caliber gunnery exercises with a surface target. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing shall cease if a marine mammal is sighted within the mitigation zone. Firing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, or the mitigation zone has been clear from any additional sightings for a period of 30 minutes.

(D) A mitigation zone with a radius of 2,500 yd. (2.3 km) around the intended impact location for explosive bombs and 1000 yd. (920 m) for non-explosive bombs shall be established for bombing exercises. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Bombing shall cease if a marine mammal is sighted within the mitigation zone. Bombing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, or the mitigation zone has been clear from any additional sightings for a period of 10 minutes.

(E) A mitigation zone of 70 yd. (64 m) shall be established for all explosive large-caliber gunnery exercises conducted from a ship. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing shall cease if a marine mammal is sighted within the mitigation zone. Firing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, the mitigation zone has been clear from any additional sightings for a period of 30 minutes, or the vessel has repositioned itself more than 140 yd. (128 m) away from the location of the last sighting.

(v) Mitigation zones for vessels and in-water devices:

(A) Vessels shall avoid approaching marine mammals head on and shall maneuver to keep at least 500 yd. (457 m) away from observed whales and 200 yd (183 m) away from all other marine mammals (except bow riding dolphins), providing it is safe to do so. These requirements shall not apply if a vessel's safety is threatened and to the extent that vessels are restricted in their ability to maneuver. Restricted maneuverability includes, but is not

limited to, situations when vessels are engaged in dredging, submerged activities, launching and recovering aircraft or landing craft, minesweeping activities, replenishment while underway and towing activities that severely restrict a vessel's ability to deviate course.

(B) A mitigation zone of 250 yd. (229 m) shall be established for all towed in-water devices, providing it is safe to do so.

(vi) Mitigation zones for non-explosive practice munitions:

(A) A mitigation zone of 200 yd. (183 m) shall be established for small-, medium-, and large-caliber gunnery exercises using a surface target. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing shall cease if a marine mammal is sighted within the mitigation zone. Firing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, the mitigation zone has been clear from any additional sightings for a period of 10 minutes for a firing aircraft, the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a firing ship, or the intended target location has been repositioned more than 400 yd. (370 m) away from the location of the last sighting.

(B) A mitigation zone of 1,000 yd. (920 m) shall be established for bombing exercises. Bombing shall cease if a marine mammal is sighted within the mitigation zone. The exercise shall not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Bombing shall recommence if any one of the following conditions is met: the animal is observed exiting the mitigation zone, the animal is thought to have exited the mitigation zone based on its course and speed, or the mitigation zone has been clear from any additional sightings for a period of 10 minutes.

(3) *Cautionary Areas* – The following are additional measures the Navy shall comply with when conducting training activities in the GOA TMAA Study Area:

(i) The Navy shall avoid training activities using hull-mounted surface ship active sonar and explosive detonations within the North Pacific Right Whale Cautionary Area, defined as the portion of the NMFS-identified biologically important feeding area for North Pacific right whale overlapping the GOA TMAA, except when required by national security needs.

(ii) In the event of national security needs, the Navy shall seek approval in advance from the Commander, U.S. Third Fleet, prior to conducting training activities using hull-mounted active sonar or explosive detonations within the Cautionary Area.

(4) *Stranding response plan*.

(i) The Navy shall abide by the letter of the "Stranding Response Plan for the Gulf of Alaska Temporary Maritime Activities Area," to include the following measures:

(A) *Shutdown procedures*. When an Uncommon Stranding Event (USE – defined in § 218.151) occurs during an MTE in the Study Area, the Navy shall implement the procedures described below:

(*1*) The Navy shall implement a shutdown when advised by a NMFS Office of Protected Resources Headquarters Senior Official designated in the GOA TMAA Study Area Stranding Communication Protocol that a USE involving live animals has been identified and that at least one live animal is located in the water. NMFS and the Navy shall maintain a dialogue, as needed, regarding the identification of the USE and the potential need to implement shutdown procedures.

(*2*) Any shutdown in a given area shall remain in effect in that area until NMFS advises the Navy that the subject(s) of the USE at that area die or are euthanized, or that all live animals involved in the USE at that area have left the area (either of their own volition or herded).

(*3*) If the Navy finds an injured or dead animal floating at sea during an MTE, the Navy shall notify NMFS immediately or as soon as operational security considerations allow. The Navy shall provide NMFS with species or description of the animal(s), the condition of the animal(s), including carcass condition if the animal(s) is/are dead, location, time of first discovery, observed behavior (if alive), and photo or video (if available). Based on the information provided, NFMS shall determine if, and advise the Navy whether a modified shutdown is appropriate on a case-by-case basis.

(*4*) In the event, following a USE, that qualified individuals are attempting to herd animals back out to the open ocean and animals are not willing to leave, or animals are seen repeatedly heading for the open ocean but turning back to shore, NMFS and the Navy shall coordinate (including an investigation of other potential anthropogenic stressors in the area) to determine if the proximity of mid-frequency active sonar training activities or explosive detonations, though farther than 14 nautical miles from the distressed animal(s), is likely contributing to the animals' refusal to return to the open water. If so, NMFS and the Navy shall further coordinate to determine what measures are necessary to improve the probability that the animals will return to open water and implement those measures as appropriate.

(B) Within 72 hours of NMFS notifying the Navy of the presence of a USE, the Navy shall provide available information to NMFS (per the GOA TMAA Study Area Communication Protocol) regarding the location, number and types of acoustic/explosive sources, direction and speed of units using mid-frequency active sonar, and marine mammal sightings information associated with training activities occurring within 80 nautical miles (148 km) and 72 hours prior to the USE event. Information not initially available regarding the 80-nautical miles (148-km), 72-hour period prior to the event shall be provided as soon as it becomes available. The Navy shall provide NMFS investigative teams with additional relevant unclassified information as requested, if available.

(ii) [Reserved]

(b) [Reserved]

2.5.1.6 ***§ 218.155 Requirements for monitoring and reporting.***

(a) The Holder of the Authorization must notify NMFS immediately (or as soon as operational security considerations allow) if the specified activity identified in § 218.150 is thought to have

resulted in the mortality or injury of any marine mammals, or in any take of marine mammals not identified in § 218.152(c).

(b) The Holder of the LOA must conduct all monitoring and required reporting under the LOA, including abiding by the GOA TMAA monitoring plan.

(c) *General notification of injured or dead marine mammals*. Navy personnel shall ensure that NMFS (regional stranding coordinator) is notified immediately (or as soon as operational security considerations allow) if an injured or dead marine mammal is found by Navy personnel during or shortly after, and in the vicinity of, a Navy training activity utilizing mid- or high-frequency active sonar, or underwater explosive detonations. The Navy shall provide NMFS with species or description of the animal(s), the condition of the animal(s) (including carcass condition if the animal is dead), location, time of first discovery, observed behaviors (if alive), and photo or video (if available). In the event that an injured, stranded, or dead marine mammal is found by the Navy that is not in the vicinity of, or during or shortly after, MFAS, HFAS, or underwater explosive detonations, the Navy shall report the same information as listed above as soon as operationally feasible and clearance procedures allow.

(d) *General notification of ship strike*. In the event of a ship strike by any Navy vessel, at any time or place, the Navy shall do the following:

(1) Immediately report to NMFS the species identification (if known), location (lat/long) of the animal (or the strike if the animal has disappeared), and whether the animal is alive or dead (or unknown), and the time of the strike.

(2) Report to NMFS as soon as operationally feasible the size and length of animal, an estimate of the injury status (ex., dead, injured but alive, injured and moving, unknown, etc.), vessel class/type and operational status.

(3) Report to NMFS the vessel length, speed, and heading as soon as feasible.

(4) Provide NMFS a photo or video, if equipment is available.

(5) Within 2 weeks of the strike, provide NMFS with a detailed description of the specific actions of the vessel in the 30-minute timeframe immediately preceding the strike, during the event, and immediately after the strike (*e.g.*, the speed and changes in speed, the direction and changes in direction, other maneuvers, sonar use, etc., if not classified); a narrative description of marine mammal sightings during the event and immediately after, and any information as to sightings prior to the strike, if available; and use established Navy shipboard procedures to make a camera available to attempt to capture photographs following a ship strike.

(e) *Communication plan*. The Navy and NMFS shall develop a communication plan that will include all of the communication protocols (phone trees, etc.) and associated contact information required for NMFS and the Navy to carry out the necessary expeditious communication required in the event of a stranding or ship strike, including information described in the proposed notification measures above.

(f) *Annual GOA TMAA monitoring report*. The Navy shall submit an annual report of the GOA TMAA monitoring describing the implementation and results from the previous calendar year. Data collection methods shall be standardized across range complexes and study areas to allow for comparison in different geographic locations. The report shall be submitted either 90 days after the calendar year, or 90 days after the conclusion of the monitoring year to be determined by the adaptive management process. The GOA TMAA Monitoring Report may be provided to NMFS within a larger report that includes the required Monitoring Plan reports from multiple range complexes and study areas (the multi-Range Complex Annual Monitoring Report). Such a report would describe progress of knowledge made with respect to monitoring plan study questions across all Navy ranges associated with the Integrated Comprehensive Monitoring Program. Similar study questions shall be treated together so that progress on each topic shall be summarized across all Navy ranges. The report need not include analyses and content that does not provide direct assessment of cumulative progress on the monitoring plan study questions.

(g) *Annual GOA TMAA exercise reports*. Each year, the Navy shall submit a preliminary report detailing the status of authorized sound sources within 21 days after the anniversary of the date of issuance of the LOA. Each year, the Navy shall submit a detailed report within 3 months after the anniversary of the date of issuance of the LOA. The annual report shall contain information on Major Training Exercises (MTEs) and a summary of all sound sources used, as described in paragraph (g)(3) of this section. The analysis in the detailed report shall be based on the accumulation of data from the current year's report and data collected from previous the report. The detailed reports shall contain information identified in paragraphs (g)(1) through (4) of this section.

(1) MFAS/HFAS Major Training Exercises - This section shall contain the following information for Major Training Exercises conducted in the GOA TMAA:

(i) Exercise Information (for each MTE):

(A) Exercise designator.

(B) Date that exercise began and ended.

(C) Location.

(D) Number and types of active sources used in the exercise.

(E) Number and types of passive acoustic sources used in exercise.

(F) Number and types of vessels, aircraft, etc., participating in exercise.

(G) Total hours of observation by lookouts.

(H) Total hours of all active sonar source operation.

(I) Total hours of each active sonar source bin.

(J) Wave height (high, low, and average during exercise).

(ii) Individual marine mammal sighting information for each sighting in each exercise when mitigation occurred:

(A) Date/Time/Location of sighting.

(B) Species (if not possible, indication of whale/dolphin/pinniped).

(C) Number of individuals.

(D) Initial Detection Sensor.

(E) Indication of specific type of platform observation made from (including, for example, what type of surface vessel or testing platform).

(F) Length of time observers maintained visual contact with marine mammal.

(G) Sea state.

(H) Visibility.

(I) Sound source in use at the time of sighting.

(J) Indication of whether animal is <200 yd, 200 to 500 yd, 500 to 1,000 yd, 1,000 to 2,000 yd, or >2,000 yd from sonar source.

(K) *Mitigation implementation*. Whether operation of sonar sensor was delayed, or sonar was powered or shut down, and how long the delay was.

(L) If source in use is hull-mounted, true bearing of animal from ship, true direction of ship's travel, and estimation of animal's motion relative to ship (opening, closing, parallel).

(M) *Observed behavior*. Lookouts shall report, in plain language and without trying to categorize in any way, the observed behavior of the animals (such as animal closing to bow ride, paralleling course/speed, floating on surface and not swimming, etc.) and if any calves present.

(iii) An evaluation (based on data gathered during all of the MTEs) of the effectiveness of mitigation measures designed to minimize the received level to which marine mammals may be exposed. This evaluation shall identify the specific observations that support any conclusions the Navy reaches about the effectiveness of the mitigation.

(2) Summary of sources used.

(i) This section shall include the following information summarized from the authorized sound sources used in all training events:

(A) Total annual hours or quantity (per the LOA) of each bin of sonar or other non-impulsive source;

(B) Total annual number of each type of explosive exercises and total annual expended/detonated rounds (missiles, bombs, sonobuoys, etc.) for each explosive bin.

(4) *Geographic information presentation*. The reports shall present an annual (and seasonal, where practical) depiction of training exercises and testing bin usage geographically across the Study Area.

(g) *MTE Prior Notification*. The Navy shall submit to NMFS (contact as specified in the LOA and Stranding Plan) an electronic notice of pending MTEs 72 hours prior to the start of the MTE indicating:

(i) Location of the exercise.

(ii) Beginning and end dates of the exercise.

(iii) Type of exercise.

(h) *Five-year close-out exercise report*. This report shall be included as part of the 2021 annual exercise report. This report shall provide the annual totals for each sound source bin with a comparison to the annual allowance and the 5-year total for each sound source bin with a comparison to the 5-year allowance. Additionally, if there were any changes to the sound source allowance, this report shall include a discussion of why the change was made and include the analysis to support how the change did or did not result in a change in the SEIS and final rule determinations. The report shall be submitted 3 months after the expiration of this subpart. NMFS shall submit comments on the draft close-out report, if any, within 3 months of receipt. The report shall be considered final after the Navy has addressed NMFS' comments, or 3 months after the submittal of the draft if NMFS does not provide comments.

2.5.1.7 *§ 218.156 Applications for letters of authorization (LOA).*

To incidentally take marine mammals pursuant to the regulations in this subpart, the U.S. citizen (as defined by § 216.106 of this chapter) conducting the activity identified in § 218.150(c) (the U.S. Navy) must apply for and obtain either an initial LOA in accordance with § 218.157 or a renewal under § 218.158.

2.5.1.8 *§ 218.157 Letters of authorization (LOA).*

(a) An LOA, unless suspended or revoked, shall be valid for a period of time not to exceed the period of validity of this subpart.

(b) Each LOA shall set forth:

(1) Permissible methods of incidental taking;

(2) Means of effecting the least practicable adverse impact on the species, its habitat, and on the availability of the species for subsistence uses (*i.e.*, mitigation); and

(3) Requirements for mitigation, monitoring and reporting.

(c) Issuance and renewal of the LOA shall be based on a determination that the total number of marine mammals taken by the activity as a whole shall have no more than a negligible impact on the affected species or stock of marine mammal(s).

2.5.1.9 *§ 218.158 Renewals and modifications of letters of authorization (LOA) and adaptive*

management.

(a) A letter of authorization issued under §§ 216.106 and 218.157 of this chapter for the activity identified in § 218.150(c) shall be renewed or modified upon request of the applicant, provided that:

(1) The proposed specified activity and mitigation, monitoring, and reporting measures, as well as the anticipated impacts, are the same as those described and analyzed for these regulations (excluding changes made pursuant to the adaptive management provision of this chapter), and;

(2) NMFS determines that the mitigation, monitoring, and reporting measures required by the previous LOA under these regulations were implemented.

(b) For LOA modification or renewal requests by the applicant that include changes to the activity or the mitigation, monitoring, or reporting (excluding changes made pursuant to the adaptive management provision of this chapter) that do not change the findings made for the regulations or result in no more than a minor change in the total estimated number of takes (or distribution by species or years), NMFS may publish a notice of proposed LOA in the **Federal Register**, including the associated analysis illustrating the change, and solicit public comment before issuing the LOA.

(c) An LOA issued under § 216.106 and § 218.157 of this chapter for the activity identified in § 218.154 may be modified by NMFS under the following circumstances:

(1) *Adaptive management*. NMFS may modify and augment the existing mitigation, monitoring, or reporting measures (after consulting with the Navy regarding the practicability of the modifications) if doing so creates a reasonable likelihood of more effectively accomplishing the goals of the mitigation and monitoring.

(i) Possible sources of data that could contribute to the decision to modify the mitigation, monitoring, and reporting measures in an LOA:

(A) Results from Navy's monitoring from the previous year(s);

(B) Results from other marine mammal and/or sound research or studies; or

(C) Any information that reveals marine mammals may have been taken in a manner, extent, or number not authorized by these regulations or subsequent LOA.

(ii) If, through adaptive management, the modifications to the mitigation, monitoring, or reporting measures are substantial, NMFS would publish a notice of proposed LOA in the **Federal Register** and solicit public comment.

(2) *Emergencies*. If NMFS determines that an emergency exists that poses a significant risk to the well-being of the species or stocks of marine mammals specified in § 218.152(c), an LOA may be modified without prior notification and an opportunity for public comment. Notification would be published in the **Federal Register** within 30 days of the action.

2.6 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 CFR 402.02).

The action area (Figure 2) for this opinion is the Gulf of Alaska Temporary Maritime Activities Area (TMAA), in addition to any areas outside the TMAA in which acoustic energy associated with the activities is propagated. The TMAA is a temporary area that is established in conjunction with the Federal Aviation Administration during the April to October timeframe for one exercise period of up to 21 days. The TMAA is a surface, undersea space, and airspace maneuver area within the Gulf of Alaska for ships, submarines, and aircraft to conduct required training activities. The TMAA overlies a majority of Warning Area (W)-612 located over Blying Sound, towards the northwestern quadrant of the TMAA. No Navy training activities analyzed in this opinion occur in the area of W-612 that is outside of the TMAA. The TMAA is a polygon that roughly resembles a rectangle oriented from northwest to southeast, approximately 300 nautical miles (nm) in length by 150 nm in width, located south of Prince William Sound and east of Kodiak Island. The TMAA's northern boundary is located approximately 24 nm (44 km) south of the shoreline of the Kenai Peninsula, which is the largest proximate landmass. The only other shoreline close to the TMAA is Montague Island, which is located 12 nm (24 km) north of the TMAA. The approximate middle of the TMAA is located 140 nm offshore.

As described in the 2016 Final Gulf of Alaska SEIS/OEIS, the Navy rarely, if ever, operates near the corners or edge of the TMAA (Navy 2016a). To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as Visit, Board, Search, and Seizure training events that are without sonar or explosives (Navy 2016a).

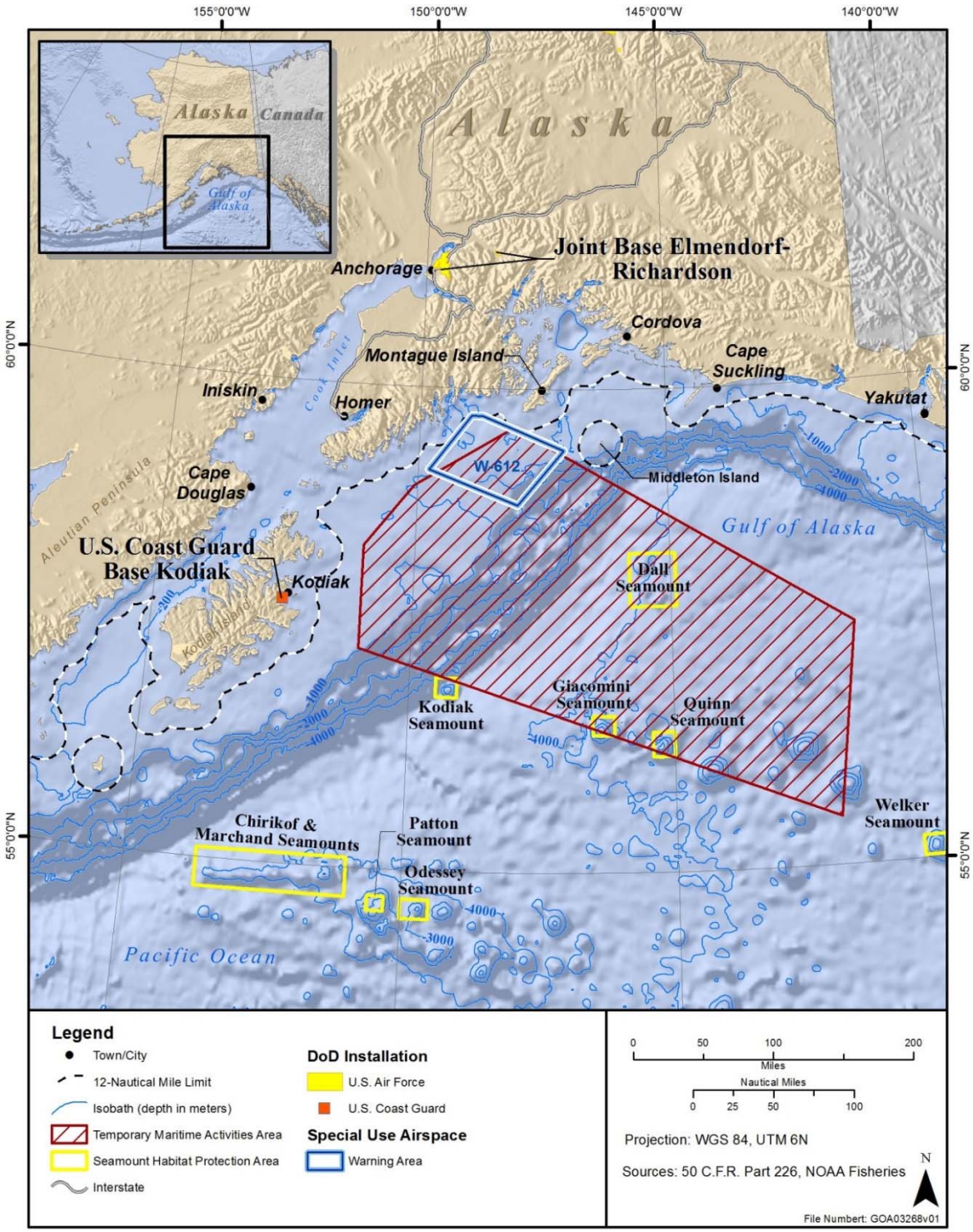


Figure 2. U.S. Navy Gulf of Alaska training activities occur in the Temporary Maritime Activities Area. The action area for this consultation includes the Temporary Maritime Activities Area, and any areas outside this area in which acoustic energy associated with the activities is propagated.

2.7 Interrelated and Interdependent Actions

Interrelated actions are those that are part of a larger action and depend on that action for their justification. *Interdependent* actions are those that do not have independent use, apart from the action under consideration. NMFS determined that there are no interrelated or interdependent actions outside the scope of Navy training activities and NMFS' issuance of incidental take authorizations pursuant to the MMPA as analyzed in this opinion.

3 OVERVIEW OF NMFS' ASSESSMENT FRAMEWORK

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

"To jeopardize the continued existence of an ESA-listed species" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02). The jeopardy analysis considers both survival and recovery of the species.

Section 7 assessment involves the following steps:

1) We identify the proposed action and those aspects (or stressors) of the proposed action that are likely to have direct or indirect effects on the physical, chemical, and biotic environment within the action area, including the spatial and temporal extent of those stressors.

2) We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time.

3) We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, impacts of state or private actions that are contemporaneous with the consultation in process.

4) We identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. This is our exposure analysis.

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

6) We assess the consequences of these responses to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. This is our risk analysis.

7) The adverse modification analysis considers the impacts of the proposed action on the critical habitat features and conservation value of designated critical habitat. Under NMFS's regulations, the destruction or adverse modification of critical habitat "means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features" (81 FR 7214).

8) We describe any cumulative effects of the proposed action in the action area.

   Cumulative effects, as defined in our implementing regulations (50 CFR §402.02), are the effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area. Future Federal actions that are unrelated to the proposed action are not considered because they require separate section 7 consultation.

9) We integrate and synthesize the above factors by considering the effects of the action to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

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

   b) Reduce the conservation value of designated or proposed critical habitat. These assessments are made in full consideration of the status of the species and critical habitat.

10) We state our conclusions regarding jeopardy and the destruction or adverse modification of critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, we must identify a reasonable and prudent alternative (RPA) to the action. The RPA must not be likely to jeopardize the continued existence of ESA-listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

3.1 Evidence Available for the Consultation

To conduct these analyses, we considered all lines of evidence available through published and unpublished sources that represent evidence of adverse consequences or the absence of such

consequences. A considerable body of scientific information on anthropogenic sounds and their effect on marine mammals and other marine life has become available. NMFS' status reviews for listed species also provide information on the status of the species including their resiliency, population trends, and specific threats to recovery that contributes to our *Status of Listed Resources*, *Environmental Baseline*, and Risk Analyses.

To comply with our obligation to use the best scientific and commercial data available, we conducted electronic literature searches throughout the consultation, including within NMFS Office of Protected Resources' electronic library (using *EndNote* ® software). We examined the literature that was cited in the submittal documents and any articles we collected through our electronic searches. The Navy provided NMFS with a Draft and Final Supplemental EIS/OEIS on training activities that are proposed in the action area, along with a Biological Evaluation (BE). We also evaluated the Navy's annual and comprehensive monitoring reports required by the existing MMPA rule and LOAs and the previous biological opinion to assess the effectiveness of mitigation and actual take incidental to training activity levels where feasible. In addition, we engage regularly with the Navy to discuss new science and technical issues as part of the ongoing adaptive management program for Navy training.

Considering the information that was available, this consultation and our opinion includes uncertainty about the basic hearing capabilities of some marine mammals, sea turtles, and fishes; how these taxa use sounds as environmental cues; how they perceive acoustic features of their environment; the importance of sound to the normal behavioral and social ecology of species; the mechanisms by which human-generated sounds affect the behavior and physiology (including the non-auditory physiology) of exposed individuals; and the circumstances that are likely to produce outcomes that have adverse consequences for individuals and populations of exposed species.

3.1.1 The Navy's Exposure Estimates

To estimate exposure of marine species to acoustic sounds, the Navy uses acoustic modeling and marine species density information developed by the Navy in cooperation with NMFS. A subsequent review on behalf of NMFS by the Center for Independent Experts analyzed the various approaches the Navy used for acoustic effects analyses, leading to the refinement of the previous methodologies for determining acoustic effects. The result was the development of a standard Navy model for acoustic effects, the Navy Acoustic Effects Model (NAEMO).

This opinion analyzes the environmental consequences based on marine mammal density data, and acoustic modeling methodology that employs acoustic criteria, and new scientific information as summarized below. The criteria used for predicting acoustic impacts to marine mammals are described in section 6.3.14 of this opinion.

3.1.1.1 *The Navy Acoustic Effects Model (NAEMO)*

Since 1997, the Navy has modeled the potential acoustic effects on marine mammals and sea turtles from Navy training activities. Various models used "area density" approaches in which acoustic footprints were computed and then multiplied by animal densities to calculate effects. As a result of a review conducted by the Center for Independent Experts, the Navy refined its process. The current model—the Navy Acoustic Effects Model (NAEMO)—is the model used by the Navy to estimate the potential acoustic effects of proposed Navy training activities on marine mammals and sea turtles. We have verified methodology and data used in NAEMO and accept the modeling conclusions on exposure of marine species. A full description of NAEMO can be accessed in the Naval Undersea Warfare Center Newport Technical Report 12,196, December 1, 2015 (NUWC 2015).

3.1.1.1.1 Overview

NAEMO is comprised of seven modules: Scenario Builder, Environment Builder, Acoustic Builder, Marine Species Distribution Builder, Scenario Simulator, Post Processor, and Report Generator. Scenario Builder defines where an activity would occur, the duration of the activity, a description of the activity, and what platforms would be participating. Once a platform is identified, all the sound sources typically associated with that platform are displayed, thus providing standardization and repeatability when different analysts are entering data. Individual sources can be turned on or off according to the requirements of the scenario. Platforms are either stationary or can be moved through the action area in either a defined track or random straight-line movement.

Environment Builder extracts all of the oceanographic and environmental data required for a scenario simulation. When an area is selected, information on bathymetry, sound speed profiles, wind speeds, and bottom properties are extracted from an array of points across the region.

Acoustic Builder generates acoustic propagation data. It reads the Scenario Builder file, allows the user to define analysis points for propagation software, and creates the propagation model inputs. Depending on the source characteristics, the propagation models utilized are Comprehensive Acoustic Simulation System/Gaussian Ray Bundle , Range-Dependent Acoustic Model , or Reflection and Refraction Multilayered Ocean/Ocean Bottoms with Shear Wave Effects.

Marine Species Distribution Builder allows the user to distribute marine species within the modeling environment in accordance with the bathymetry and relevant descriptive data. Marine species density data reviewed in consultation with NMFS, which include seasonal information when available, are obtained from the Navy Marine Species Density Database (NMSDD); the sizes of cells and density of marine species within each cell vary by species and location. Final

data input into the NMSDD prior to modeling is reviewed and coordinated with NMFS as the best species-specific scientific information available.

Scenario Simulator executes the simulation and records the sound received by each marine mammal and sea turtle in the area. It incorporates the scenario definition, sound propagation data, and marine species distribution data and ultimately provides raw data output for each simulation. Most scenarios are run in 4- to 12-hour segments based on representative training activities. Some scenarios are evaluated by platform and single locations, while others are evaluated in multiple locations within a single range complex or training range. Within each scenario, multiple ship track iterations are run to provide a set of raw data results.

Post Processor provides the computation of estimated effects that exceed defined threshold criteria (described in section 6.3.14 of this opinion) from each of the raw data files produced by Scenario Simulator. The post-processed computations determine harassment and mortality as defined by the MMPA for military readiness activities. It also tabulates and graphs the output data for review. As described further in section 6.3.14, the Navy uses a behavioral response function to quantify the number of behavioral responses that could qualify as Level B behavioral harassment under the MMPA. As the statutory definition is currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated using the behavioral response functions do not differentiate between the different types of potential reactions nor the significance of those potential reactions. These estimates also do not provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals.

Report Generator assembles a series of simulation results from multiple post-processing runs and produces a combined result. Multipliers can be applied to each scenario to compute the effects of conducting them multiple times. Results can also be exported via Microsoft Excel files for further analysis and reporting.

Modeled effects from NAEMO were used to support the Navy's analyses in the Gulf of Alaska FSEIS/OEIS, mitigation strategies, Biological Evaluations, and MMPA incidental take authorization applications. We have verified methodology and data used in NAEMO and accept the modeling conclusions on exposure of marine species. A full description of NAEMO can be accessed in the Naval Undersea Warfare Center Newport Technical Report 12,196, December 1, 2015 (NUWC 2015). The following paragraphs provide an overview of the NAEMO process and its more critical data inputs.

The NAEMO improves upon previous modeling efforts in several ways. First, unlike earlier methods that modeled acoustic sources individually, the NAEMO has the capability to run all sources within a scenario simultaneously, providing a more realistic depiction of the potential

effects of an activity. Second, previous models calculated sound received levels within set volumes of water and spread animals uniformly across the volumes; in the NAEMO, animats (virtual animals) are distributed non-uniformly based on higher resolution species-specific density, depth distribution, and group size, and animats serve as dosimeters, recording energy received at their location in the water column. Third, a fully three-dimensional environment is used for calculating sound propagation and animat exposure in the NAEMO, rather than a two-dimensional environment where the worst case sound pressure level across the water column is always encountered. Additionally, NAEMO expands upon previous modelling efforts by incorporating Type II frequency weighting functions, incorporating a behavioral response function, and developing estimates from a new density function (NMSDD). Finally, current efforts incorporate site-specific bathymetry, sound speed profiles, wind speed, and bottom properties into the propagation modeling process rather than the flat-bottomed provinces used during earlier modeling (NUWC 2015).

Using data from the NMSDD, the NAEMO derives an abundance (total number of individuals (i.e., animats)) for the modeled area. The NAEMO then distributes the animats into an area bounded by the maximum distance acoustic energy propagates out to a threshold value (energy footprint). For example, for non-impulsive sources, animats that could receive sound pressure levels greater than or equal to 120 dB are distributed. Animats are distributed based on density differences across the area, the group (pod) size, and known depth distributions (dive profiles) (see Marine Species Modeling Team (2013) for a discussion of animal dive profiles in detail). Animats change depths every 4 minutes but do not otherwise mimic actual animal behaviors, such as avoidance or attraction to a stimulus (horizontal movement), or foraging, social, or traveling behaviors.

Schecklman et al. (2011) argue static distributions underestimate acoustic exposure compared to models with three-dimensionally moving animals. Their static method is different from the NAEMO in several ways. First, they distribute the entire population at a depth typical for that species and those animats remain static at that position throughout the entire simulation. In the NAEMO, animats are placed horizontally dependent on non-uniform density information, and then move vertically over time based on species-specific diving behavior. Second, the static method calculates acoustic received level for designated volumes of the ocean and then sums the animats that occur within that volume, rather than using the animats themselves as dosimeters, as in the NAEMO. Third, Schecklman et al. (2011) ran 50 iterations of the moving distribution to arrive at an average number of exposures, but because they rely on uniform horizontal density (and static depth density) only a single iteration of the static distribution is realized. In addition to moving the animats vertically, the NAEMO overpopulates the animats over a non-uniform density and then resamples the population a number of times to arrive at an average number of exposures. Tests comparing fully moving animats and horizontally static animats with vertical mobility were compared during development of the NAEMO. For vertical position updates occurring more frequently than every 5 minutes, the number of estimated exposures was similar

between the NAEMO and the fully moving distribution; however, computational time was much longer for the fully moving distribution.

The NAEMO calculates the likely propagation for various levels of energy (sound or pressure) resulting from each non-impulse or impulse source used during a training or training event. These calculations account for bathymetric relief and bottom types (e.g., reflective), estimated sound speeds, and sea surface roughness. Platforms (such as a ship using one or more sound sources) are modeled moving across an area representative of what would normally occur during a training or training scenario. The model uses typical platform speeds and event durations. Moving source platforms either travel along a predefined track or move in straight lines from a random initial starting point, reflecting at the edges of a predefined boundary. Static sound sources are stationary in a fixed location for the duration of a scenario. Modeling locations were chosen based on historical data where activities have been ongoing and to include environmental variation within the TMAA.

The NAEMO records the energy received by each animat within the energy footprint of the event and calculates the number of animats having received levels of energy exposures that fall within defined impact thresholds.

Predicted effects on the animats are tallied and the most severe effect (e.g., PTS over TTS) predicted for a given animat is assumed. Each scenario, or each 24-hour period for scenarios lasting greater than 24 hours, is independent of all others. Therefore, the same individual could be impacted during each independent scenario or 24-hour period. In few instances, although the activities themselves all occur within the Gulf of Alaska TMAA, sound may propagate beyond the boundary of the TMAA. Any exposures occurring outside the boundary of the TMAA are counted as if they occurred within the TMAA boundary or within the action area for this opinion.

3.1.1.1.2 Model Assumptions

There are limitations to the data used in the NAEMO, and the results must be interpreted within this context. While the most accurate data and input assumptions have been used, when there is a lack of definitive data to support an aspect of the modeling, assumptions that overestimate exposures have been chosen:

Animats are modeled as being underwater, stationary, and facing the source and therefore always predicted to receive the maximum sound level (i.e., no porpoising or pinnipeds' heads above water). Some odontocetes have been shown to have directional hearing, with best hearing sensitivity facing a sound source and higher hearing thresholds for sounds propagating towards the rear or side of an animal (Mooney et al. 2008) (Popov and Supin 2009) (Kastelein et al. 2009).

- Animats do not move horizontally (but change their position vertically within the water column), which may overestimate physiological effects such as hearing loss, especially for slow moving or stationary sound sources in the model.

- Animats are stationary horizontally and therefore do not avoid the sound source, unlike in the wild where animals would most often avoid exposures at higher sound levels, especially those exposures that may result in PTS.

- Animats are assumed to receive the full impulse of the initial positive pressure wave of an explosion, although the impulse-based thresholds (onset mortality and onset slight lung injury) assume an impulse delivery time adjusted for animal size and depth. Therefore, these impacts are overestimated at farther distances and increased depths.

- Multiple exposures within any 24-hour period are considered one continuous exposure for the purposes of calculating the temporary or permanent hearing loss, because there are not sufficient data to estimate a hearing recovery function for the time between exposures.

- Mitigation measures implemented during training activities were not considered in the model. In reality, sound-producing activities would be reduced, stopped, or delayed if marine mammals are detected within the mitigation zones.

3.1.2 Post Processing - Avoidance Behavior and Mitigation Measures as Applied to Sonar and Active Acoustic Sources

Because of the model limitations and simplifications described above, initial predicted model results must be further analyzed, considering such factors as likely avoidance by marine mammals and the likelihood of successfully implementing mitigation measures. This analysis uses a number of factors in addition to the acoustic model results to more accurately estimate the acoustic effects to marine mammals.

The Navy assessed the effect of animal avoidance behavior and implementation of mitigation by considering the following:

- Best available science on species' behavior,

- Number of platforms (i.e., aircraft, vessels) used during specific activities,

- Ability to detect specific species, and

- Ability to observe the mitigation zone around different platforms during different activities.

3.1.2.1 *Animal avoidance behavior*

As described in the Gulf of Alaska FSEIS/OEIS, the NAEMO model accounts for an animat's position vertically in the water column by taking into account species-specific dive profiles. However, it does not account for an animat's horizontal movement, so the model assumes that an animal would remain stationary and tolerate repeated intense sound exposures at very close distances. This assumption is invalid because animals are likely to leave the area to avoid intense sound exposure that could cause injury. Similarly, the modeling assumes that certain species known to avoid areas of high anthropogenic activity would remain in the very close vicinity of all Navy training activities, regardless of how many vessels or low-flying aircraft (i.e., helicopters) are involved. At close ranges and high sound levels approaching those that could cause PTS, avoidance of the area immediately around the sound source is the assumed behavioral response of exposed animals for most cases. In other words, the model estimates PTS impacts as though an animal would tolerate an injurious sound exposure without moving away from the sound source. The outputs of the model, therefore, present an unrealistically high estimate of acoustic impacts in close proximity to certain Navy training activities. The potential for avoidance is considered in the Navy's post-model analysis. This is discussed further in Gulf of Alaska FSEIS/OEIS (see sections on *Avoidance of Human Activity* and *Avoidance Behavior and Mitigation Measures as Applied to Sonar and other Active Acoustic Sources*).

3.1.2.2 *Mitigation*

The Navy implements mitigation measures during sound-producing activities, including halting or delaying use of sonar or another active acoustic source or an explosion when marine mammals are observed in the mitigation zone. Sound-producing activities would not begin or resume until the mitigation zone is observed to be free of marine mammals. The NAEMO estimates acoustic effects without any shutdown or delay of the activity in the presence of marine mammals; therefore, the model overestimates impacts to marine mammals within mitigation zones. The post-model analysis considers the potential for mitigation to reduce effects on ESA-listed marine mammals due to exposure to sonar and other active acoustic sources and explosions. The Navy's proposed mitigations were developed in cooperation with NMFS and are designed to reduce environmental impacts while being operationally feasible. It is difficult to assess the effectiveness of mitigation measures. However, NMFS assesses annual exercise reports and comprehensive summary reports to assess general trends in implementation and any observed responses to mitigation. Two factors are considered when quantifying the effectiveness of mitigation: (1) the extent to which the type of mitigation proposed for a sound-producing activity (e.g., active sonar) allows for observation of the mitigation zone prior to and during the activity; and (2) the sightability of each species that may be present in the mitigation zone, which is affected by species-specific characteristics.

The post-model acoustic effect analysis quantification process is summarized in Table 7 and presented in more detail in the technical report *Post-Model Quantitative Analysis of Animal Avoidance Behavior and Mitigation Effectiveness for the Gulf of Alaska Training Activities* (Navy 2014b) or Section 3.8.3.1.8 of the 2016 Gulf of Alaska SEIS/OEIS. In brief, the mitigation effectiveness score for an event (i.e., 1, 0.5, 0, depending on how much of the mitigation zone can be observed) is multiplied by the estimated sightability of each species to quantify the number of animals that were originally modeled as a mortality (explosives only) or injury (all sound-producing activities) exposure but would, in reality, be observed by Lookouts or shore-based observers prior to or during a sound-producing activity. Observation of marine mammals prior to or during a sound-producing event would be followed by stop or delay of the sound-producing activity, which would reduce actual marine mammal sound exposures. The Navy only quantitatively adjusted model-predicted effects within the range to mortality (explosives only) and injury (all sound-producing activities). Despite employing the required mitigation measures during an activity that will also reduce some TTS exposures, the Navy did not quantitatively adjust the model-predicted TTS effects or other predicted behavioral effects as a result of implemented mitigation. The total model-predicted number of animals affected is not reduced by the post-model mitigation analysis, since all reductions in mortality and injury effects are then added to and counted as TTS effects.

Table 7. Post Model Acoustic Impact Analysis Process

Is the Sound Source Sonar/Other Active Acoustic Source or Explosives?	
Sonar and Other Active Acoustic Sources	Explosives
S-1. Is the activity preceded by multiple vessel activity or hovering helicopter?	E-1. Is the activity preceded by multiple vessel activity or hovering helicopter?
Species sensitive to human activity (i.e., beaked whales) are assumed to avoid the activity area, putting them out of the range to Level A harassment. Model-estimated PTS to these species during these activities are unlikely to actually occur and, therefore, are considered to be TTS (animal is assumed to move into the range of potential TTS). The activities preceded by multiple vessel movements or hovering helicopters are listed in Tables 3.4-14 and 3.4-15 in Section 3.4.4.1.2 (Avoidance Behavior and Mitigation Measures as Applied to Sonar and Other Active Acoustic Sources) in the FEIS.	Species sensitive to human activity (i.e., beaked whales) are assumed to avoid the activity area, putting them out of the range to mortality. Model-estimated mortalities to these species during these activities are unlikely to actually occur and, therefore, are considered to be injuries (animal is assumed to move into the range of potential injury). The activities preceded by multiple vessel movements or hovering helicopters are listed in Table 3.4-20 in Section 3.4.4.2.2 (Avoidance Behavior and Mitigation as Applied to Explosives) in the FEIS.
S-2. Can Lookouts observe the activity-specific mitigation zone (see Chapter 5) up to and during the sound-producing activity?	E-2. Can Lookouts observe the activity-specific mitigation zone (see Chapter 5) up to and during the sound-producing activity?

Is the Sound Source Sonar/Other Active Acoustic Source or Explosives?	
Sonar and Other Active Acoustic Sources	Explosives
If Lookouts are able to observe the mitigation zone up to and during a sound-producing activity, the sound-producing activity would be halted or delayed if a marine mammal is observed and would not resume until the animal is thought to be out of the mitigation zone (per the mitigation measures in Chapter 5). Therefore, model-estimated PTS exposures are reduced by the portion of animals that are likely to be seen [Mitigation Effectiveness (1, 0.5, or 0) x Sightability, g(0)]. Any animals removed from the model-estimated PTS are instead assumed to be TTS (animal is assumed to move into the range of TTS). The g(0) value is associated with the platform (vessel or aircraft) with the dedicated Lookout(s). For activities with lookouts on both platforms, the higher g(0) is used for analysis. The g(0) values are provided in Table 3.4-8. The Mitigation Effectiveness values are provided in Table 3.4-16 in Section 3.4.4.1.2 (Avoidance Behavior and Mitigation Measures as Applied to Sonar and Other Active Acoustic Sources) in the FEIS.	If Lookouts are able to observe the mitigation zone up to and during an explosion, the explosive activity would be halted or delayed if a marine mammal is observed and would not resume until the animal is thought to be out of the mitigation zone (per the mitigation measures in Chapter 5). Therefore, model-estimated mortalities and injuries are reduced by the portion of animals that are likely to be seen [Mitigation Effectiveness (1, 0.5, or 0) x Sightability, g(0)]. Any animals removed from the model-estimated mortalities or injuries are instead assumed to be injuries or behavioral disturbances, respectively (animals are assumed to move into the range of a lower effect). The g(0) value is associated with the platform (vessel or aircraft) with the dedicated Lookout(s). For activities with lookouts on both platforms, the higher g(0) is used for analysis. The g(0) values are provided in Table 3.4-8. The Mitigation Effectiveness values are provided in Table 3.4-21 in Section 3.4.2.2 (Avoidance Behavior and Mitigation as Applied to Explosives) in the FEIS.

3.1.3 Discussion of Finneran and Schlundt 2010 and 2011 Dolphin Studies in the Context of Phase II Modeling

The Navy incorporated the data from two Finneran studies (2010 and 2011), in coordination with other scientific literature, to develop auditory weighting functions and "weighted" thresholds for auditory criteria. A summary of the findings from the two papers is provided below.

3.1.3.1 *Finneran and Schlundt (2010)*

Finneran and Schlundt (2010) measured temporary threshold shift (TTS) in a single female bottlenose dolphin (*Tursiops truncatus*) after exposure to tones at 3 and 20 kHz in order to examine the effects of exposure frequency on the onset and growth of TTS. The preliminary data provide evidence of frequency specific differences in TTS onset and growth between the 3 kHz and 20 kHz exposures. At 20 kHz, where bottlenose dolphin hearing sensitivity is better, TTS not only began at a lower exposure level compared to the 3 kHz exposures, but also grew at a faster rate. This demonstrated that damage risk criteria for dolphins exposed to underwater sound should account for the exposure frequency and that criteria developed for lower frequencies (e.g. 3 kHz) may underestimate the amount of TTS if applied to higher frequencies (e.g. 20 kHz),

where hearing sensitivity is better. This research suggests the need for analogous data across the entire audible range so that potential effects of various frequency tones can be properly assessed.

3.1.3.2 *Finneran and Schlundt (2011)*

For humans, acoustic damage-risk criteria rely on numeric thresholds based on "weighted" noise levels. Weighted noise levels are calculated by applying a frequency-dependent filter, or "weighting function" to the measured sound pressure before calculation of the overall sound pressure level (SPL). The weighting functions are designed to emphasize frequencies where sensitivity to sound is high and to de-emphasize frequencies where sensitivity is low. This technique allows for a single, weighted damage-risk criterion, regardless of the sound frequency. Weighting functions for humans are derived from equal loudness contours—graphs representing the SPLs that led to a sensation of equal loudness magnitude in the listener as a function of sound frequency (Suzuki and Takeshima 2004). Equal loudness contours are derived from loudness experiments where the listener is asked to judge the relative loudness of two tones with different frequencies. Prior to Finneran and Schlundt (2011a) there were no direct measurements of subjective loudness in non-human animals from which to develop equal loudness contours. Finneran and Schlundt (2011a) trained a bottlenose dolphin to perform a loudness comparison test, where the listener indicated which of two sequential tones was louder. This study demonstrated that a non-human animal could be conditioned for subjective loudness training and therefore, it was possible to directly measure loudness levels in some species. Additional data is required to more accurately predict the relationship below 2.5 kHz. The weighting function derived here is substantially different than the "M-weighting function" proposed for mid-frequency cetaceans in Southall et al. (Southall et al. 2007b), which is nearly flat over the range of approximately 1 to 30 kHz and thus does not mirror the change in equal loudness contours observed over that frequency range. Nor does the M-weighting function capture the difference in TTS onset and growth reported for a single bottlenose dolphin tested at 3 and 20 kHz in Finneran and Schlundt (2010).

3.1.4 Consideration of the National Oceanic and Atmospheric Administration's Marine Mammal Acoustic Technical Guidance

The criteria used for predicting acoustic impacts to marine mammals are described in section 6.3.14 of this opinion. On August 4, 2016, NMFS released its Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (new Guidance). This new Guidance established new thresholds and associated weighting functions for predicting auditory injury, or permanent threshold shift (PTS) and temporary threshold shift (TTS). NMFS uses acoustic thresholds to help quantify "take" and as part of more comprehensive effects analyses under several statutes, including the ESA. In the August 4, 2016, Federal Register notice announcing the new Guidance (81 FR 51694), NMFS explained the approach it would take during a transition period, during which we will balance the need to consider this new best available science with the fact that some applicants have already committed time and resources

to the development of analyses based on our previous thresholds and have constraints that preclude the recalculation of take estimates, as well as consideration of where the action is in the agency's decision-making "pipeline." In that notice, we included a non-exhaustive list of factors that would inform the most appropriate approach for considering the new Guidance, including: how far in the process the application or prospective application has progressed; when the activity is scheduled to begin or other timing constraints; the complexity of the analyses and the cost and practicality of redoing them; the temporal and spatial scope of anticipated effects; and the relative degree to which the new Guidance is expected to affect the results of the acoustic impact analyses.

In developing the new Guidance, NMFS compiled, interpreted, and synthesized scientific information currently available on the effects of anthropogenic sound on marine mammals, including a recent Technical Report by Dr. James Finneran (U.S. Navy-SPAWAR Systems Center Pacific) that proposed new weighting functions and thresholds for predicting the onset of both PTS and temporary threshold shifts (TTS) in marine mammals (Finneran, 2016). The methodologies presented within this paper (and in NMFS' new Guidance) build upon the methodologies used to develop the criteria applied within the proposed rule and Navy's GOA FSEIS/OEIS (Finneran and Jenkins, 2012), and incorporate relevant auditory research made available since 2012 (e.g., Kastelein *et al*., 2012a; Kastelein *et al*., 2012b; Finneran and Schlundt, 2013; Kastelein *et al*., 2013a; Kastelein *et al*., 2013b; Popov *et al*., 2013; Kastelein *et al*., 2014a; Kastelein *et al*., 2014b; Popov *et al*., 2014; Finneran *et al*., 2015; Kastelein *et al*., 2015a; Kastelein *et al*., 2015b; Popov *et al*., 2015). In light of limited data at the time, Finneran and Jenkins (2012) presented a conservative approach to development of auditory weighting functions. In 2016, with the benefit of newly-available data, Finneran was able to synthesize a wide range of auditory data, including newly-available studies, to predict refined auditory weighting functions and corresponding TTS and PTS thresholds across the complete hearing ranges of functional hearing groups. The new criteria were not available for the Navy's acoustic effects modeling used to calculate distances to harassment thresholds and resulting take estimates for this consultation. Therefore, the Navy did not directly use the new auditory weighting functions and PTS/TTS criteria in its acoustic modeling or the GOA FSEIS/OEIS.

It would be impractical for the Navy to entirely re-model its proposed action based on the new Guidance. The Navy committed substantial time and resources to the development of acoustic analyses based on previous acoustic thresholds. Data and information (*e.g.*, on marine species density) gathering for Phase II GOA modeling began in November 2011 and subsequent modeling occurred over a 20 month period from October 2012 to June 2014. The underlying science contained within Finneran (2016) (upon which NMFS' new Guidance is based) has been addressed qualitatively within the applicable sections of the GOA FSEIS/OEIS and this biological opinion. Although the writers of the base code for the model used for Phase II were not available to recode the model with the updated impulsive criteria in terms of weighting functions, the Navy was able to use the model to reprocess anticipated explosive ranges to effects

for PTS based on the criteria presented in the new Guidance to assess if the new criteria could result in any additional species-specific injury exposures. In short, the Navy quantitatively reanalyzed PTS ranges and exposures from explosive sources using the new Guidance, from which TTS and behavioral exposures could be estimated, but the sonar exposures were not remodeled because a qualitative assessment of the new Guidance and the activities showed that it was not necessary in order to support the analysis, in addition to being impractical.

For the sonar exposure estimates, if the new Guidance was quantitatively applied to the GOA TMAA effects analysis and new modeling conducted, predicted numbers of PTS and/or TTS would change to some small degree (even if only by fractions of a take). However, because the new Guidance relies on much of the same data as the auditory criteria used in the Navy's modeling, these changes would not be substantial, and in most cases would result in a reduction in the predicted impacts. Onset PTS thresholds for non-impulsive sound (sonar) are largely lower (*i.e.*, are more conservative) in Finneran and Jenkins 2012 (used in the Navy's modeling) compared to the new Guidance, while updated auditory weighting functions for most marine mammal hearing groups have changed minimally in the new Guidance. This means that the predicted ranges to PTS and TTS in the GOA FSEIS/OEIS and this opinion for non-impulsive sources would change only minimally (and for the most part are larger than what would result) if NMFS' new Guidance were quantitatively applied and new modeling conducted (*i.e.*, estimated numbers of takes resulting in PTS and TTS from sonar are, for the most part, larger in this opinion than would be expected if the Navy's activities were re-modeled using the new Guidance).

Specifically, PTS thresholds for non-impulsive sources for all taxa went up (*i.e.*, are less conservative), except for Otariids (e.g., Steller sea lions), for which they went down by one dB. Given that the PTS range to effects for Otariids was previously 10m, a 1dB change in the PTS threshold would not change the PTS range to effects by more than a couple of meters for any acoustic source. For TTS, the onset thresholds for cetaceans in the new Guidance all went up (*i.e.*, are less conservative) or stayed the same (*i.e.*, ranges to effects and take estimates for TTS would go down or stay the same for cetaceans if the Navy's activities were re-modeled using the new Guidance). The onset thresholds for TTS for Otariids went down by 7 dB. The previous range to effects was 230-570m for Otariids for the largest source (53C). If spherical spreading were conservatively considered, applying the new Guidance, the range to TTS for Otariids would likely be no more than approximately 500-1,300m. The originally modeled TTS for Steller sea lions was zero. When the lower likelihood of overlap of Steller sea lions with these activities is considered in combination with their densities and the change in the size of the ensonified zone, our analysis still suggests that TTS take is not likely to occur, and TTS take estimates have not been changed for this species.

For impulsive sound (explosives), the Navy was able to reprocess anticipated ranges to effects for Level A harassment (PTS), and subsequently ranges to effects for TTS and behavioral exposures, based on the new Guidance to assess if the new impulsive criteria could result in any additional

species-specific takes. The conclusion from that analysis was that the new impulsive criteria would not change previous species-specific quantities of impulsive PTS, TTS, or behavioral exposures for any ESA-listed species, and the mitigation zones described in the 2016 GOA FSEIS by explosives training activity remain sufficiently protective (*i.e.*, revised range to effects for PTS, even when larger, remain encompassed by the mitigation zones for all explosive types and hearing groups).

It is important to emphasize that these updated acoustic thresholds do not represent the entirety of an impact assessment, but rather serve as one tool (in addition to behavioral impact thresholds, auditory masking assessments, evaluations to help understand the ultimate effects of any particular type of impact on an individual's fitness, population assessments, etc.), to help evaluate the effects of a proposed action. Further, takes generated by modeling are used as estimates, not absolutes, and are factored into NMFS' analysis accordingly. The results of prior Navy modeling described in this opinion represent the best available estimate of the number and type of take that may result from the Navy's use of acoustic sources in the action area. Modeling that incorporated the updated acoustic thresholds could result in minor changes to the enumerations of take estimates. However, as described above, use of the new acoustic thresholds would not alter our assessment of the likely responses of affected ESA-listed species to acoustic sources employed by Navy in the action area, or the likely fitness consequences of those responses.

3.1.5 Criteria for Assessing Effects to Fish from Sonar

This section details sound exposure criteria for fishes from sonar that were proposed by the Navy and agreed to by NMFS for this consultation. These criteria were largely derived from the extensive review provided in Popper et al. (2014b) "Sound Exposure Guidelines for Fishes and Sea Turtles." Thresholds within that technical report are generally presented at the lowest level at which the effect occurred. In some cases the thresholds presented in Popper et al. (2014b) did not show any effect but are the only data available for that stressor. Therefore, these guidelines may be overly conservative. A description of each cell is presented below to explain the derivation of the threshold value proposed. For additional information on the methodology used to develop these criteria, see Renken (2015).

Thresholds for TTS are typically reported in cumulative sound exposure level (SEL_{cum}) so as to account for the duration of the exposure and therefore are presented in terms of SEL_{cum} metric.

Acoustic Units

SEL_{cum} - Cumulative sound exposure level (dB re 1 $\mu Pa^2 \cdot s$)
SPL_{rms} - Root mean square sound pressure level (dB re 1 μPa)
SPL_{peak} - Peak (0 – peak) sound pressure level (dB re 1 μPa)

Acoustic Calculations (see Richardson 1995)

$SEL_{cum} = SPL_{rms} + 10 \log t$
Where t = duration of exposure in seconds

Table 8. Sound exposure criteria for fishes exposed to sonar.

Column #	Low-Frequency Navy Sonar (< 1 kHz)					
	Row Letter	**A**	**B**	**C**	**D**	**E**
		Mortality & mortal injury	**Recoverable injury**	**TTS**	**Masking**	**Behavior**
1	Fish-no SB (swim bladder)	>> 218 dB SEL_{cum}	> 218 dB SEL_{cum}	> 218 dB SEL_{cum}	(N) Low (I) Low (F) Low	(N) Low (I) Low (F) Low
2	Fish w/ SB not involved in hearing (particle motion detection)	>> 218 dB SEL_{cum}	> 218 dB SEL_{cum}	210 dB SEL_{cum}	(N) Low (I) Low (F) Low	(N) Low (I) Low (F) Low
3	Fish w/ SB used in hearing (pressure detection)	>> 218 dB SEL_{cum}	> 218 dB SEL_{cum}	210 dB SEL_{cum}	(N) Mod (I) Low (F) Low	> 197 dB SPL_{rms}
Mid-Frequency Navy Sonar (1-10 kHz)						
		Mortality & mortal injury	Recoverable injury	TTS	Masking	Behavior
4	Fish-no SB	>> 221 dB SEL_{cum}	> 221 dB SEL_{cum}	NA	NA	NA
5	Fish w/ SB not involved in hearing (particle motion detection)	>> 221 dB SEL_{cum}	> 221 dB SEL_{cum}	NA	NA	NA
6	Fish w/ SB used in hearing (pressure detection)	>> 221 dB SEL_{cum}	> 221 dB SEL_{cum}	220 dB SEL_{cum}	(N) Low (I) Low (F) Low	200 dB SPL_{rms}

NA = No data available or threshold is not applicable to fish
(N) = near (i.e. tens of meters from the source)

(I) = intermediate (i.e. 100s of meters from the source)
(F) = far (thousands of meters form the source)
High, Mod (moderate), and Low = Probability of the effect occurring. For any cell containing these designations please see Popper et al. (2014) for meaning.

3.1.5.1 *Low-Frequency Navy Sonar*

The following sections outline criteria to assess effects to fish from low-frequency sonar.

3.1.5.1.1 Mortality, Mortal Injury, and Recoverable Injury All Fish = > 218 dB SEL_{cum} (cells A1, A2, A3, B1, B2 & B3 in Table 8 above)

Sonar has not been known to cause mortality, mortal injury, or recoverable injury to fish in the wild due to lack of fast rise times, lack of high peak pressures, and lack of high acoustic impulse associated with some impulsive sounds (e.g., explosives). Long duration exposures (up to 2 hours) of sonar to fish in laboratory settings has caused stunning and mortality in some cases but these exposures were much longer than any exposure a fish would normally encounter in the wild due to Gulf of Alaska proposed activities because both fish and vessels or aircraft using sonar would be moving, and not likely in the same direction. In addition, the subjects exposed in the lab were held in a cage for the duration of the exposure, unable to avoid the source (Hastings 1991; Hastings 1995a). Exposure to low-frequency sonar has been tested at levels up to 193 dB SPL (rms) for 324 seconds (218 dB SEL_{cum}) and has not been shown to cause mortality or any injury in fish with swim bladders (Kane et al. 2010; Popper et al. 2007). Lesser potential for injurious effects would be expected for fish without air cavities (i.e., swim bladders). Therefore the recommended threshold would be >> 218 dB SEL_{cum} for mortality and > 218 dB SELcum for recoverable injury.

3.1.5.1.2 Temporary Threshold Shift, Fish-no SB = > 218 dB SEL_{cum} (cell C1 in Table 8 above)

Exposure to low-frequency sonar has not been shown to induce TTS in fish species without swim bladders (Popper et al. 2014b).

3.1.5.1.3 Temporary Threshold Shift, Fish w/ SB = 210 dB SELcum (cells C2 & C3 in Table 8 above)

Exposure to sonar above 1 kHz has been known to induce TTS in some fish species with swim bladders (Halvorsen et al. 2012; Popper et al. 2007). Subjects from Popper et al. (2007) may have undergone varying husbandry treatments or possessed different genetics which may have resulted in higher than normal shifts. Criteria provided in Popper et al. (2014b) were reported in dB SPL_{rms}. This criteria was converted to SEL based on the signal durations reported in Popper et al. (2007) and Halvorsen et al. (2012) and was rounded down from the lowest sound exposure level as a conservative measure.

193dB SPLrms + 10log(324 sec) = 218 dB SEL_{cum} (Popper et al. 2007)

3.1.5.1.4 Masking, Fish w/out SB and Fish w/ SB not involved in hearing = (N)Low, (I)Low, (F)Low (cells D1 & D2 in Table 8 above)

No data are available on masking by sonar but it is unlikely that sonar would mask important sounds for fish. Risk of significant masking occurring within any distance from the source is low (Popper et al. 2014b). The narrow bandwidth of most sonar would result in only a limited range of frequencies being masked (Popper et al. 2014b). Furthermore most sonars are intermittent (i.e., low duty cycle) which further lowers the probability of any masking effects.

3.1.5.1.5 Masking, Fish w/ SB involved in hearing = (N)Mod, (I)Low, (F)Low (cell D3 in Table 8 above)

No data are available on masking by sonar but it is unlikely that sonar would mask important sounds for fish. The risk of masking occurring is moderate near the source and low at intermediate and far distances from the source (Popper et al. 2014b); however, the narrow bandwidth of most sonar would result in only a limited range of frequencies being masked (Popper et al. 2014b). Furthermore most sonars are intermittent (i.e., low duty cycle) which further lowers the probability of any masking effects.

3.1.5.1.6 Behavior, Fish no SB and Fish w/ SB not involved in hearing = (N)Low, (I)Low, (F)Low (cells E1 & E2 in Table 8 above)

No data are available on behavioral reactions to low-frequency sonar. Fish without a mechanism to sense pressure are unlikely to sense sound beyond the near-field. The risk that sonar would result in a behavioral response within near, intermediate or far distances from sonar is low (Popper et al. 2014b).

3.1.5.1.7 Behavior: Fish w/ SB involved in hearing = > 197 dB SPLrms (cell E3 in Table 8 above)

No reactions were seen in fish exposed to 1 to 2 kHz sonar which is categorized as mid-frequency sonar, not low-frequency sonar. Therefore criteria used for behavioral reactions to sonar was taken from Popper et al. (2014b), >197 dB SPLrms (Doksaeter et al. 2009; Doksaeter et al. 2012).

3.1.5.2 *Mid-Frequency Navy Sonar*

The following sections outline criteria to assess effects to fish from mid-frequency sonar.

3.1.5.2.1 Mortality, Mortal Injury & Recoverable Injury: >> 221 dB SEL_{cum} (cells A4, A5,

A6, B4, B5, & B6 in Table 8 above)

Sonar is not anticipated to cause mortality, mortal injury, or recoverable injury due to lack of fast rise times, lack of high peak pressures, and lack of high acoustic impulse associated with some impulsive sounds (e.g., explosives). Exposure to mid-frequency sonar has been tested and has not been shown to cause mortality or any injury in fish with swim bladders (Kane et al. 2010; Popper et al. 2007). Lesser potential for injurious effects would be expected for fish without air cavities (i.e., swim bladders). Therefore the recommended threshold would be >>221 dB SEL_{cum} for mortality and >221 dB SEL_{cum} for recoverable injury.

3.1.5.2.2 TTS: Fish-no SB and Fish w/SB not involved in hearing = NA (cells C4 & C5 in Table 8 above)

Exposure to mid-frequency sonar has not been known to induce TTS in fish species without swim bladders or in fish with swim bladders that are not involved in hearing (Halvorsen et al. 2012). In addition, fish without swim bladders involved in hearing (i.e. close connections to the inner ear) do not sense pressure well and cannot hear at frequencies above 1 kHz.

3.1.5.2.3 TTS: Fish w/ SB used in hearing = 220 dB SEL_{cum} (cell C6 in Table 8 above)

Exposure to mid-frequency sonar has been known to induce TTS in some fish species with swim bladders and better hearing capabilities (Halvorsen et al. 2012). Criteria from Popper et al. (2014b) was originally listed as >210 dB SPL_{rms}. As previously stated, TTS criteria reported as cumulative sound exposure level (SEL_{cum}) accounts for the duration of the exposure as well. Therefore, the criteria originally presented in the technical report was converted to this metric using the duration of the signal reported from the experiments and was rounded down as a conservative measure (Halvorsen et al. 2012).

$$210 \text{ dB } SPL_{rms} + 10\log(15 \text{ sec}) = 221 \text{ dB } SEL_{cum}$$

3.1.5.2.4 Masking: NA (cells D4, D5, & D6 in Table 8 above)

No data are available on masking by sonar but it is unlikely that sonar would mask important sounds for fish. The narrow bandwidth of most sonar would result in only a limited range of frequencies being masked (Popper et al. 2014b). Furthermore most sonars are intermittent (i.e., low duty cycle) which further lowers the probability of any masking effects. Most mid-frequency sonars are above the hearing range of most fish species and almost all marine fish species (including salmonids).

3.1.5.2.5 Behavior: Fish no SB and Fish w/ SB not involved in hearing = NA (cells E4 & E5

in Table 8 above)

Fish without swim bladders or without swim bladders involved in hearing would not be able to hear mid-frequency sonar; therefore, behavioral reactions would not occur.

3.1.5.2.6 Behavior: Fish w/ SB involved in hearing = 200 dB SPL_{rms} (cell E6 in Table 8 above)

No reactions were seen in herring exposed to 1 to 2 and 6 to 7 kHz sonar (Doksaeter et al. 2009; Doksaeter et al. 2012). Therefore it is recommended that this criterion be 200 dB SPL_{rms} as a conservative measure. This criterion only applies to mid-frequency sonars up to 2.5 kHz since even fish with swim bladders with connections to the inner ear cannot hear above these frequencies with the exception of the taxa *Alosa* spp. (e.g., herring). While improbable (Doksaeter et al. 2009; Doksaeter et al. 2012), *Alosa* spp. could have behavioral reactions over the full bandwidth of mid-frequency sonar (1 to 10 kHz).

3.1.6 Criteria for Assessing Effects to Fish from Explosives

During this consultation, NMFS needed to assess the effects of impulsive stressors (explosions) on ESA-listed salmonids. The most appropriate thresholds to assess are the onset of physical injury and onset of mortality from impulsive stressors. These thresholds must be based on the best scientific and commercial data[1] pursuant to section 7 of the ESA.

3.1.6.1 *Datasets/Thresholds Evaluated for Effects from Explosives*

The following thresholds were assessed by NMFS to establish suitable criteria for the ESA risk analysis. For additional information on the methodology used to develop these criteria, see Renken (2015).

[1] Best available scientific and commercial data - to assure the quality of the biological, ecological, and other information used in the implementation of the Act, it is the policy of the Services to: (1) evaluate all scientific and other information used to ensure that it is reliable, credible, and represents the best scientific and commercial data available; (2) gather and impartially evaluate biological, ecological, and other information disputing official positions, decisions, and actions proposed or taken by the Services; (3) document their evaluation of comprehensive, technical information regarding the status and habitat requirements for a species throughout its range, whether it supports or does not support a position being proposed as an official agency position; (4) use primary and original sources of information as the basis for recommendations; (5) retain these sources referenced in the official document as part of the administrative record supporting an action; (6) collect, evaluate, and complete all reviews of biological, ecological, and other relevant information within the schedules established by the Act, appropriate regulations, and applicable policies; and (7) require management-level review of documents developed and drafted by Service biologists to verify and assure the quality of the science used to establish official positions, decisions, and actions taken by the Services during their implementation of the Act. [59 FR 34271 (July 1, 1994)]

3.1.6.1.1 NMFS West Coast Region's Interim Impact Pile Driving Thresholds

In 2008, NMFS' West Coast Region established interim dual thresholds for the onset of physical injury from impact pile driving activities (Table 9) via participation in the Fisheries Hydroacoustic Working Group (FHWG 2008). The NMFS dual interim thresholds are expressed as peak pressure (dB_{peak}) and cumulative sound exposure level (SEL_{cum}) and account for vulnerability depending on fish size.

Table 9. NMFS Dual Interim Impact Pile Driving Injury Thresholds.

Interim Criteria for Injury	Agreement in Principle
Peak	206 dB (for all size of fish)
Cumulative SEL	187 dB - for fish size of two grams or greater. 183 dB - for fish size of less than two grams.

Because of limited data, the FHWG relied on data from a variety of surrogate impulsive sources (i.e., explosives: (Carlson and Hastings 2007; Govoni et al. 2003; Govoni et al. 2008; Yelverton et al. 1975); seismic airguns: (Popper et al. 2005; Song et al. 2008; Stadler and Woodbury 2009)) to derive dual interim thresholds for impact pile driving.

3.1.6.1.2 Popper et al. 2014 Thresholds for Explosives

The authors of Popper et al. (2014b) did a thorough review of available data associated with fishes and explosions and concluded "The problem for setting guidelines is that the studies that have examined the effects of explosions on fishes have each used different species, different types of explosives, and/or charges of different weights. Since the methodologies and data are so varied, the guidelines in the Tables are based on a paper representing the lowest amplitude that caused consistent mortality" based on data from Hubbs and Rechnitzer (1952) (Table 10). Thus, Popper et al. (2014b) did not provide any threshold recommendations beyond mortality/mortal injury. In addition to mortality/mortal injury threshold, this consultation required NMFS to use a threshold to determine onset injury.

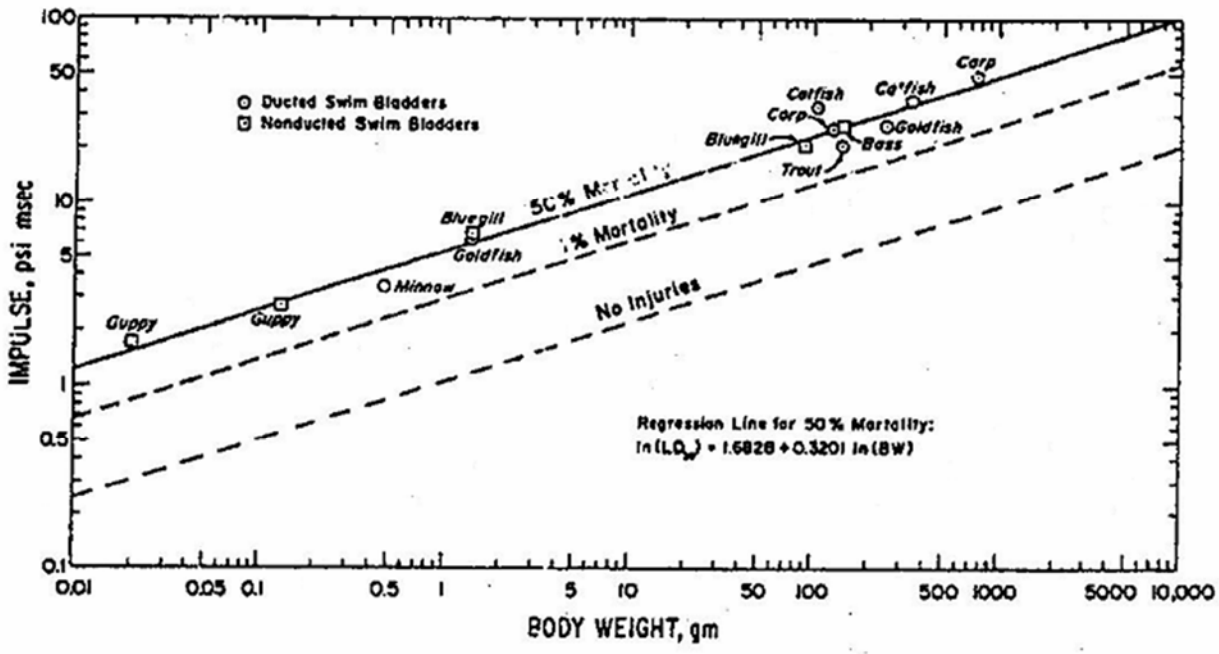
Table 10. Popper et al. 2014 Explosive Thresholds for Mortality/Mortal Injury (highlighted).

Guidelines for explosions. Levels other than for eggs and larvae from Hubbs and Rechnitzer (1952); levels for eggs and larvae from Wright and Hopky (1998). Guidelines are not provided for masking since the animals are not exposed to more than a few explosive events, and masking would not last beyond the period of exposure

Type of Animal	Mortality and potential mortal injury	Impairment			Behavior
		Recoverable injury	TTS	Masking	
Fish: no swim bladder (particle motion detection)	229-234 dB peak	(N) High (I) Low (F) Low	(N) High (I) Moderate (L) Low	NA	(N) High (I) Moderate (F) Low
Fish where swim bladder is not involved in hearing (particle motion detection)	229-234 dB peak	(N) High (I) High (F) Low	(N) High (I) Moderate (F) Low	NA	(N) High (I) High (F) Low
Fish where swim bladder is involved in hearing (primarily pressure detection)	229-234 dB peak	(N) High (I) High (F) Low	(N) High (I) High (F) Low	NA	(N) High (I) High (F) Low
Sea turtles	229-234 dB peak	(N) High (I) High (F) Low	(N) High (I) High (F) Low	NA	(N) High (I) High (F) Low
Eggs and larvae	>13 mm s^{-1} peak velocity	(N) High (I) Low (F) Low	(N) High (I) Low (F) Low	NA	(N) High (I) Low (F) Low
Notes: peak and rms sound pressure levels dB re 1 μPa; SEL dB re 1 $\mu Pa^2 \cdot s$. All criteria are presented as sound pressure even for fish without swim bladders since no data for particle motion exist. Relative risk (high, moderate, low) is given for animals at three distances from the source defined in relative terms as near (N), intermediate (I), and far (F).					

3.1.6.1.3 Yelverton et al. 1975

This study examined the effects of explosives, in terms of injury and mortality, on eight species of freshwater fishes (ducted and non-ducted swim bladders) ranging in size from 0.02 g to 744 g. This study found a direct correlation between fish body mass and mortality/injury (i.e., kidney and liver damage, swim bladder rupture) when explosives were described in terms of the impulse metric (pounds per square inch-milliseconds (psi-msec)) (Figure 3). Based on data derived during this study a model was provided to determine no injury, 1 percent mortality, and 50 percent mortality (1.7 to 49.5 psi-msec). The equation presented by Young (1991) for 10 percent mortality was modified by the Navy using Yelverton et al. (1975) to find no injury and 1 percent mortality.



The results of study by Yelverton et al. (1975) to determine the effects of underwater blasts on fishes. A direct correlation was found between body mass and the received sound impulse, characterized by psi-msec, which caused 50% mortality. The correlation was independent of peak overpressure, thus indicating that sound energy may be more indicative than peak pressure in determining damage thresholds. Fish with ducted swim bladders were found to be just as vulnerable to blast injury and death as those without ducts. (Note: Yelverton et al. reported no control test specimens in this study.)

Figure 3. Results from Yelverton et al. (1975) (caption from Hastings and Popper (2005)).

3.1.6.2 *NMFS' Selected Thresholds*

The most appropriate thresholds for explosives are those derived directly from explosive data. However, limited direct data associated with explosives complicates the ability to establish appropriate thresholds. Additionally, every dataset has its own set of caveats and considerations.

We determined that Yelverton et al. (1975) represents an appropriate study to consider in deriving thresholds for the onset of injury (e.g., no mortality model). The intent of the Popper et al. (2014b) explosive thresholds is to represent the onset of mortality. However, for NMFS's analysis, thresholds were also needed to account for the potential of sub-lethal impacts, specifically the onset of physical injury. The FWHG examined explosive data (e.g., (Govoni et al. 2003; Govoni et al. 2008)) when deriving interim thresholds for impact pile driving. If one examines how these data fit within the model derived by Yelverton et al. (1975) (as demonstrated by Hastings 2007; Figure 3), they lend additional support to this model.

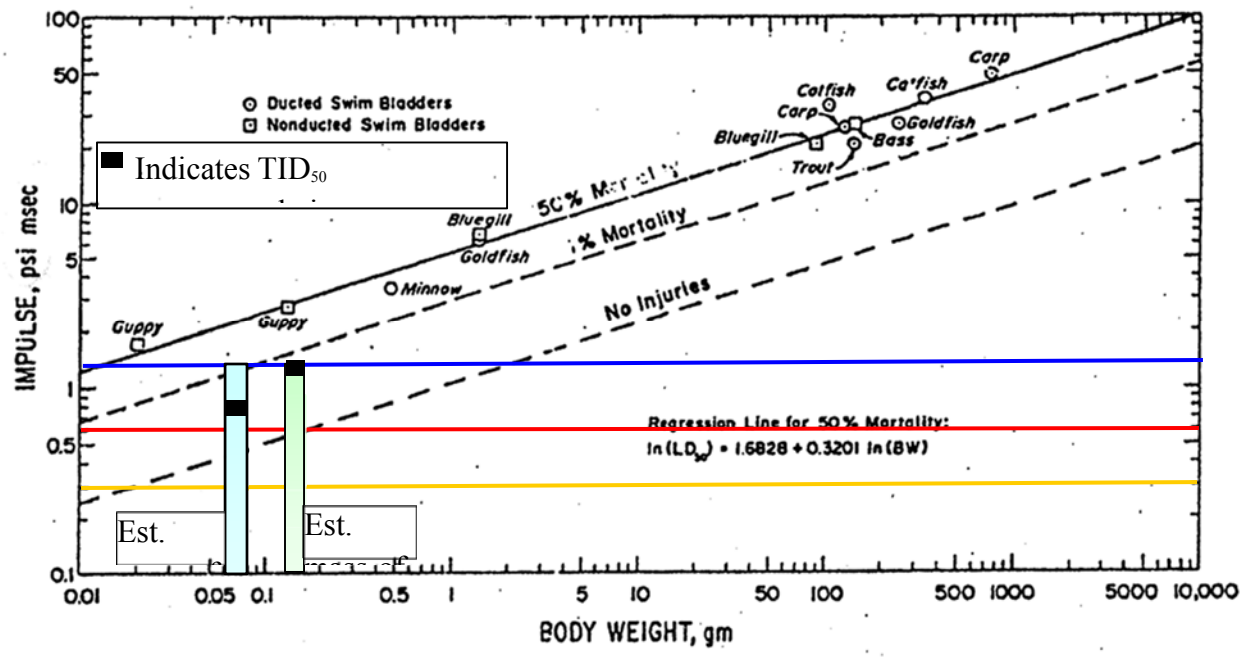


Figure 4. Average impulse values from Govoni et al. (2003, 2007; vertical aqua bars with black squares indicating total injury dose (TID_{50}) recommended by Govoni et al. 2007) compared with results of Yelverton et al. (1975) from Hasting 2007. Distance from the explosive source is indicated by the color of the line: blue – 3.6 m; red – 7.5 m; and yellow – 17 m.

Regarding potential effects beyond injury, NMFS acknowledges that explosives have a faster rise time and more high-frequency energy content than other impulsive sources, like pile driving and seismic. Nevertheless, the only TTS onset data available for impulsive sources comes from seismic exposure (Popper et al. 2005). Popper et al. (2014b) used these data to recommend TTS onset thresholds (186 dB SEL_{cum}) for both pile driving and seismic. Since no other data are available, this is the most appropriate dataset to also consider for explosives.

In summary, NMFS determined that the thresholds listed in Table 11 are appropriate to assess effects to salmonids resulting from impulsive stressors associated with Navy Gulf of Alaska activities. These thresholds may also be appropriate for other actions involving explosions in water.

Table 11. Thresholds for Assessing Effects of Explosives on Fish

Potential Effect	Threshold	Reference
Mortality/Mortal Injury*	229 dB_{peak}	Hubbs and Rechnitzer (1952)
Onset of Injury	TBD psi-msec (based on representative weight of fish species and age class)	Yelverton et al. (1975), Young (1991)
TTS	186 dB SEL_{cum}	Popper et al. (2005)
*** Yelverton et al. (1975) could possibly be used to establish dual thresholds using the impulse (psi-msec) metric (e.g., 1 percent mortality model)**		

3.2 Treatment of "Cumulative Impacts" (in the sense of NEPA)

The U.S. Council on Environmental Quality defined "cumulative effects" (which we refer to as "cumulative impacts" to distinguish between NEPA and ESA uses of the same term) as "the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions" (40 CFR §1508.7). The effects analyses of biological opinions considered the "impacts" on listed species and designated critical habitat that result from the incremental impact of an action by identifying natural and anthropogenic stressors that affect endangered and threatened species throughout their range (the *Status of Listed Resources*) and within an action area (the *Environmental Baseline*, which articulate the pre-existing *impacts* of activities that occur in an action area, including the past, contemporaneous, and future *impacts* of those activities). We assess the effects of a proposed action by adding its direct and indirect effects to the *impacts* of the activities we identify in an *Environmental Baseline* (50 CFR §402.02), in light of the impacts of the status of the listed species and designated critical habitat throughout their range; therefore, the results of our effects analyses are equivalent to those contained in the "cumulative impact" sections of NEPA documents.

We considered cumulative impacts as part of our consultation. Specifically, we considered (1) stressors that accumulate in the environment, and (2) effects that represent either the response of individuals, populations, or species to that accumulation of stressors. Further, we considered the likely impacts of these accumulative phenomena on an annual basis, over the duration of the five-year MMPA regulations, and under the assumption that these activities would continue into the reasonably foreseeable future. Given the ongoing nature of the proposed activities, we assume that the type, amount, and extent of training do not exceed maximum levels assessed in the action.

In the sense of Item 1, which captures the normal usage of "cumulative impacts," we concluded that phenomena like sound and ship strike do not accumulate in the environment (sound energy rapidly transforms into other forms of energy and ship strikes are independent events), although phenomena like the acreage of habitat destroyed and concentrations of toxic chemicals, sediment, and other pollutants accumulate.

In the sense of Item 2, we considered phenomena that accumulate in individuals and individually contribute or collectively determine the probable fitness of the individuals that comprise a population. These include, the passage of time and its corollary, the passage or loss of time (specifically, the loss of time to reproduce, to forage, and to migrate, etc.); reproductive success; longevity; energy debt, including allostatic loading; body burdens of toxic chemicals; the fitness costs of behavioral decisions (canonical costs); injuries and tissue damage; and overstimulation of sensory organs (which would include noise-induced losses of hearing sensitivity).

At the level of populations, phenomena that "accumulate" include population abundance; the number or percent of individuals in a population with lifetime reproductive success greater than 2.0 (successful offspring produced); the number or percent of individuals in a population with lifetime reproductive success equal to 2.0; the number or percent of individuals in a population with lifetime reproductive success less than 2.0; the number or percent of individuals that emigrate from a population per unit time; the number or percent of individuals that immigrate into a population per unit time; mortality within a particular age or stage over generation time; and the reservoir of juveniles in a population that have a high probability of surviving to the age of reproduction (population momentum or its absence).

At the species level, when feasible, we accumulate those phenomena that allow us to estimate the extinction risks facing a species. These include increases or decreases in the number of occurrences or populations; the extinction probability of particular occurrences; variance in the rates of population growth or decline; and demographic stochasticity.

Cumulative effects also include effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future Federal actions that are unrelated to the action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

3.3 Defining "Significance"

In this biological opinion, we focused on the potential physical, chemical, or biotic stressors that are "significant" in the sense of being distinct from ambient or background. We then asked if

a. exposing individuals to those potential stressors is likely to represent a "significant" negative experience in the life history of individuals that have been exposed; and if
b. exposing individuals to those potential stressors is likely to cause the individuals to experience "significant" physical, chemical, or biotic responses; and if
c. any "significant" physical, chemical, or biotic response are likely to have "significant" consequence for the fitness of the individual animal; and if
d. exposing the physical, chemical, or biotic phenomena that we identified as constituent elements in a critical habitat designation or, in the case of critical habitat designations that do not identify constituent elements, those physical, chemical or biotic phenomena that give designated critical habitat value for the conservation of endangered or threatened species is likely to represent a "significant" change in the quantity, quality, or availability of the physical, chemical, or biotic resource; and if
e. any "significant" change in the quantity, quality, or availability of a physical, chemical, or biotic resource is likely to "significantly" reduce the conservation value of the designated critical habitat.

In all of these cases, the term "significant" means "clinically or biotically significant" rather than statistically significant because the presence or absence of statistical significance do not imply the presence or absence of clinical significance (Achinstein 2001; Royall 2004).

For populations (or sub-populations, demes, etc.), we are concerned about whether the number of individuals that are likely to experience "significant" reductions in fitness and the nature of any fitness reductions are likely to have a "significant" consequence for the viability (= probability of demographic, ecological, or genetic extinction) of the population(s) those individuals represent. Here "significant" also means "clinically or biotically significant" rather than statistically significant.

For "species" (the entity that has been listed as endangered or threatened, not the biological species concept), we are concerned about whether the number of populations that are likely to experience "significant" reductions in viability (= increases in their extinction probabilities) and the nature of any reductions in viability are likely to have "significant" consequence for the viability (= probability of demographic, ecological, or genetic extinction) of the "species" those populations comprise. Here, again, "significant" also means "clinically or biotically significant" rather than statistically significant.

For designated critical habitat, we are concerned about whether the area that has been designated is likely to experience "significant" reductions in the quantity, quality, or availability of physical, chemical, or biotic resources that are likely to result in "significant" reductions in the conservation value (usually measured using the concept of "carrying capacity[2]") of the entire area contained in the designation.

3.4 Defining "Population"

For this opinion the term "population" refers to groups of individuals whose patterns of increase or decrease in abundance over time are determined by internal dynamics (births resulting from sexual interactions between individuals in the group and deaths of those individuals) rather than external dynamics (immigration or emigration). This definition is a reformulation of definitions articulated by Futuymda (1986) and Wells and Richmond (1995) and is more restrictive than those uses of 'population' that refer to groups of individuals that co-occur in space and time but do not have internal dynamics that determine whether the size of the group increases or decreases over time (see review by Wells and Richmond 1995). The definition we apply is important to section 7 consultations because such concepts as "population decline," "population collapse," "population extinction," and "population recovery" apply to the restrictive definition of "population" but do not explicitly apply to alternative definitions. As a result, we do not treat

[2] I.e., the maximum number of individuals of a particular species that a given environment (habitat) can support without detrimental effects to the environment.

the different whale "stocks" recognized by the International Whaling Commission or other authorities as populations unless those distinctions were clearly based on demographic criteria. We do, however, acknowledge those "stock" distinctions in these narratives.

4 STATUS OF LISTED RESOURCES

This section identifies the ESA-listed species that occur within the action area that may be affected by Navy Gulf of Alaska activities in the action area. It then summarizes the biology and ecology of those species and what is known about their life histories in the action area. The listed species including distinct population segments (DPS) or evolutionarily significant units (ESU) occurring within the action area that may be affected by the proposed action are listed in Table 12, along with their ESA listing status.

Table 12. Species listed under the Endangered Species Act (ESA) under NMFS' jurisdiction that may occur in the action area.

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Blue Whale (*Balaenoptera musculus*)	E - 35 FR 18319	-- --	1998
Fin Whale (*Balaenoptera physalus*)	E - 35 FR 18319	-- --	2010
Gray Whale – Western North Pacific Population (*Eschrichtius robustus*)	E - 35 FR 18319	-- --	-- --
North Pacific Right Whale (*Eubalaena japonica*)	E - 73 FR 12024	73 FR 19000	2013
Sei Whale (*Balaenoptera borealis*)	E - 35 FR 18319	-- --	2011
Sperm Whale (*Physeter macrocephalus*)	E - 35 FR 18319	-- --	2010
Humpback whale (*Megaptera novaeangliae*)			1991
Western North Pacific DPS	E – 81 FR 62259	-- --	-- --
Mexico DPS	T – 81 FR 62259	-- --	-- --
Marine Mammals – Pinnipeds			
Steller Sea Lion – Western DPS (*Eumetopias jubatus*)	E - 62 FR 24345	58 FR 45269	2008
Sea Turtles			
Leatherback Turtle (*Dermochelys coriacea*)	E - 61 FR 17	--	1998 (Pacific)
Green sea turtle – Central North Pacific and East Pacific DPSs (*Chelonia mydas*)	T – 81 FR 20057	--	1998
Olive ridley sea turtle (*Lepidochelys olivacea*)	T - 43 FR 32800	--	1998
Loggerhead sea turtle – North Pacific Ocean DPS (*Caretta caretta*)	E – 76 FR 58868	--	1998
Fish – Salmonids			
Chinook Salmon (*Oncorhynchus tshawytscha)* Evolutionarily Significant Units (ESU)			
Puget Sound ESU	T - 64 FR 14308	--	2007
Lower Columbia River ESU	T – 64 FR 14308	--	2013
Upper Columbia River spring-run ESU	E – 64 FR 14308	--	2007
Upper Willamette River ESU	T – 64 FR 14308	--	2011
Snake River spring/summer-run ESU	T – 59 FR 42529	--	--
Snake River fall-run ESU	T – 59 FR 42529	--	--

Species	ESA Status	Critical Habitat	Recovery Plan
Central Valley, spring-run ESU	T – 64 FR 50393	--	2014
California Coastal ESU	T – 64 FR 50393	--	2007
Sacramento River, Winter-run ESU	E – 59 FR 440	--	2014
Chum Salmon (*Oncorhynchus keta*) ESUs			
Columbia River ESU	T – 64 FR 14507	--	2013
Hood Canal Summer Run ESU	T - 64 FR 14507	--	2007
Coho Salmon (*Oncorhynchus kisutch*) ESUs			
Lower Columbia River ESU	T - 70 FR 37160	--	2013
Oregon Coast ESU	T – 63 FR 42587	--	--
So. Oregon Northern California Coast ESU	T- 76 FR 50447	--	2014
Central California Coast ESU	E – 61 FR 56138	--	2012
Sockeye Salmon (*Oncorhynchus nerka*) ESUs			
Ozette Lake ESU	T – 64 FR 14528	--	2009
Snake River ESU	E – 56 FR 58619	--	2015
Steelhead Trout (*Oncorhynchus mykiss*) Distinct Population Segments (DPS)			
Lower Columbia River DPS	T - 71 FR 834	--	2013
Upper Columbia River DPS	T – 71 FR 834	--	2007
Middle Columbia River DPS	T – 71 FR 834	--	2009
Puget Sound DPS	T - 72 FR 26722	--	-- --
Upper Willamette River DPS	T – 71 FR 834	--	2011
Northern California DPS	T - 71 FR 834	--	-- --
Central California Coast DPS	T - 71 FR 834	--	-- --
Snake River Basin DPS	T – 71 FR 834	--	-- --
California Central Valley DPS	T – 71 FR 834	--	2014
South-Central California Coast DPS	T – 71 FR 834	--	2013
Southern California DPS	E – 71 FR 834	--	2012

4.1 Species and Critical Habitat Not Considered Further in this Opinion

As described in the *Approach to the Assessment*, NMFS uses two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the various activities. The first criterion is exposure or some reasonable expectation of a co-occurrence between one or more potential stressors associated with the Navy's activities and a particular listed species or designated critical habitat: if we conclude that a listed species or designated critical habitat is not likely to be exposed to the activities, we must also conclude that

the species or critical habitat is not likely to be adversely affected by those activities. The second criterion is the probability of a response given exposure, which considers susceptibility: species that may be exposed to sound transmissions from active sonar, for example, but are likely to be unaffected by the sonar (at sound pressure levels they are likely to be exposed to) are also not likely to be adversely affected by the sonar. For designated critical habitat, we consider the susceptibility of the constituent elements or the physical, chemical, or biotic resources whose quantity, quality, or availability make the designated critical habitat valuable for an endangered or threatened species. We applied these criteria to the species and critical habitat listed at the beginning of this section; this subsection summarizes the results of those evaluations.

4.1.1 North Pacific Gray Whale - Western Population

Gray whales (*Eschrichtius robustus*) occur in two genetically distinct populations in the North Pacific Ocean (Brownell Jr. et al. 2009; Burdin et al. 2011; Kanda et al. 2010; Lang et al. 2004; Lang et al. 2005; Lang et al. 2010; Leduc et al. 2002; Swartz et al. 2006b; Weller et al. 2007; Weller et al. 2004; Weller et al. 2006). These are formally recognized as the western North Pacific stock that was listed as endangered in 1970 (35 FR 18319) under the Endangered Species Act and shows no apparent signs of recovery, and the Eastern North Pacific stock that appears to have recovered from exploitation and was removed from listing under the ESA in 1994 (Carretta et al. 2013a; Swartz et al. 2006a), and is not included in this biological opinion. Although the western North Pacific stock is listed as endangered under the ESA, there is no designated critical habitat for this species.

Gray whales are mysticetes, or baleen whales. Gray whales are the only species in the family Eschrichtiidae. These large whales can grow to about 50 ft (15 m) long, and weigh approximately 80,000 lb (35,000 kg). Females are slightly larger than males. They have a mottled gray body, with small eyes located just above the corners of the mouth. Their "pectoral fins" (flippers) are broad, paddle-shaped, and pointed at the tips. Lacking a dorsal fin, they instead have a "dorsal hump" located about two-thirds of the way back on the body, and a series of 8 to 14 small bumps, known as "knuckles," between the dorsal hump and the tail flukes. The tail flukes are more than 15 ft (3 m) wide, have S-shaped trailing edges, and a deep median notch.

Gray whales are frequently observed traveling alone or in small, unstable groups, although large aggregations may be seen on feeding and breeding grounds. Similar to other baleen whales, long-term bonds between individuals are rare. Gray whales are bottom feeders, and suck sediment and the "benthic" amphipods that are their prey from the sea floor. To do this, they roll on their sides and swim slowly along, filtering their food through coarse baleen plates, of which they have 130 to 180 on each side of the upper jaw. In doing so, they often leave long trails of mud behind them, and "feeding pits" in the sea floor.

Western North Pacific gray whales migrate annually along Asia during autumn, although migration routes are poorly known. Migration from summer foraging areas off the northeastern coasts of Sakhalin Island and south-eastern Kamchatka along the Japanese coasts to the South China Sea is suspected (Commission 2004; IWC 2003; Omura 1988; Tsidulko et al. 2005; Weller et al. 2008b; Weller et al. 2012c).

Eastern and western North Pacific gray whales were once considered geographically separated along either side of the ocean basin, but recent photoidentification, genetic, and satellite tracking data refute this. Two western North Pacific gray whales have been satellite tracked from Russian foraging areas east along the Aleutian Islands, through the Gulf of Alaska, and south to the Washington State and Oregonian coasts in one case (Mate et al. 2011) and to the southern tip of Baja California and back to Sakhalin Island in another (IWC 2012). Comparisons of eastern and western North Pacific gray whale catalogs have thus far identified 23 western North Pacific gray whales occurring on the eastern side of the basin during winter and spring (Weller et al. 2013). Burdin et al. (2011) found an additional individual. During one field season off Vancouver Island, western gray whales were found to constitute 6 of 74 (8.1 percent) of photoidentifications (Weller et al. 2012b). In addition, two genetic matches of western gray whales off Santa Barbara, California have been made (Lang et al. 2011). Individuals have also been observed migrating as far as central Baja Mexico (Weller et al. 2012c).

Group sizes vary, but are roughly 2 (range 1 to 14) for non-calf groups and slightly larger for groups containing calves (Weller et al. 2007; Weller et al. 2004; Weller et al. 2006; Weller et al. 1999; Yakovlev and Tyurneva 2004).

4.1.1.1 *Distribution*

Western North Pacific gray whales exhibit extensive plasticity in their occurrence, shifting use areas within and between years, as well as over longer time frames, such as in response to oceanic climate cycles (e.g., El Nino-Southern Oscillation, Pacific Decadal Oscillation, and Arctic Oscillation) (Gardner and Chavez-Rosales 2000; Meier et al. 2007; Tyurneva et al. 2009; Vladimirov et al. 2006a; Vladimirov et al. 2006b; Vladimirov et al. 2005; Vladimirov et al. 2008; Vladimirov et al. 2009; Vladimirov et al. 2010; Weller et al. 2012c; Yablokov and Bogoslovskaya 1984; Yakovlev and Tyurneva 2005). Species distribution extends south along Japan, the Koreas, and China from the Kamchatka Peninsula (IWC 2003; Kato and Kasuya. 2002; Omura 1988; Reeves et al. 2008; Weller et al. 2003). Other possible range states include Vietnam, the Philippines, and Taiwan, although only historical whaling records support occurrence in these areas (Henderson 1990; Ilyashenko 2009). Range has likely contracted from the Koreas and other southern portions of the range versus pre-whaling periods. Prey availability and, to a lesser extent, sea ice extent, are probably strong influences on the habitats used by western North Pacific gray whales (Clarke and Moore 2002; Moore 2000).

4.1.1.1.1 Occurrence in the Gulf of Alaska TMAA

Most gray whales follow the coast during migration and stay within 1.2 miles (mi.) (2 kilometers [km]) of the shoreline, except when crossing major bays, straits, and inlets from southeastern Alaska to the eastern Bering Sea (Braham 1984). However, gray whales are known to move farther offshore between the entrance to Prince William Sound and Kodiak Island and between Kodiak Island and the southern part of the Alaska Peninsula (Consiglieri et al. 1982a). Gray whales use the nearshore areas of the Alaska Peninsula during the spring and fall migrations and are often found within the bays and lagoons, primarily north of the peninsula, during the summer (Navy 2006a). During the April 2009 survey of the action area, one group of two gray whales was sighted while on-effort within the action area (Rone et al. 2009). There was one off-effort sighting (25 individuals) southeast of Kodiak Island during a survey of the action area in June and July 2013 (Rone et al. 2014).

Gray whale calls were detected during a single hour on a single day, 29 September 2012, at the High-frequency Acoustic Recording Package (HARP) deployed in the slope region of north-central Gulf of Alaska (Baumann-Pickering et al. 2012a). Since gray whales tend to stay close to shore during their migration, the HARP deployment locations are likely too far offshore to capture more gray whale signals (Baumann-Pickering et al. 2012b).

Previous sighting data suggested that the remaining population of gray whales in the western Pacific had a limited range extending between the Okhotsk Sea off the coast of Sakhalin Island (Russia) and the South China Sea (Weller et al. 2002). However, recent long-term studies of radio-tracked whales indicate that the coastal waters of eastern Russia, the Korean Peninsula, and Japan are part of the migratory route (Weller et al. 2012a). There is also photographic evidence of a match between a whale found off Sakhalin Island and the Pacific coast of Japan, more than 932 mi. (1,500 km) south of the Sakhalin feeding area (Weller et al. 2008a). Mate et al. (2010) and Mate et al. (2015) documented movement of a western Pacific gray whale from Sakhalin Island to the nearshore waters off Washington state. This whale tracked via long-term satellite tag traveled directly across the southern Gulf of Alaska via a direct path from the Aleutian Islands to Washington state. Further, photo-catalog comparisons of eastern and western North Pacific gray whale populations suggest that there is more exchange between the western and eastern populations than previously thought, since "Sakhalin" whales were sighted off Santa Barbara, California; British Columbia, Canada; and Baja California, Mexico (Weller et al. 2013). Due to the already low abundance of western North Pacific gray whales, their occurrence in the action area during the summer time period is considered rare.

4.1.1.2 *Abundance Estimate*

The western North Pacific gray whale population was once considered extinct, but now small numbers are known to exist (Weller et al. 2002). The most recent estimate of this population is 140 individuals (CV = 0.04; Carretta et al. 2015).

4.1.1.3 *Critical Habitat*

NMFS has not designated critical habitat for the western North Pacific gray whale population.

4.1.1.4 *Conclusion*

There are insufficient data to derive a species or stock-specific density for western gray whales. To quantify the likely effects to the western North Pacific gray whale population from acoustic stressors, a ratio based on the abundance of eastern North Pacific gray whales was used to prorate effects on western North Pacific gray whales. Predicted exposures to acoustic stressors would not exceed the current effects thresholds and thus would not rise to the level of "take" pursuant to the ESA. Therefore, effects to Western North Pacific gray whales will be insignificant and this species is not likely to be adversely affected by acoustic stressors.

We also conclude that because of the extremely low numbers of the western North Pacific gray whale stock in the North Pacific Ocean and rare occurrence in the Gulf of Alaska, exposure to stressors such as vessel strike and direct strike of expended materials would be unlikely. Therefore, we concur with the Navy's determination of NLAA and have determined that the western North Pacific stock of gray whales is not likely to be adversely affected by the proposed action. As a result, this species will not be considered further in this opinion.

4.1.2 Leatherback Sea Turtle

The leatherback turtle (*Dermochelys coriacea*) was listed under the Endangered Species Act as endangered throughout its range in 1970 (61 FR 17). The 2013-2014 Biennial Report to Congress states the status of the Pacific stock is considered decreasing (NMFS 2014). In the Pacific Ocean, leatherback sea turtles occur in two genetically and biologically distinct subpopulations, as the western and eastern Pacific leatherback sea turtles. Leatherback sea turtles are considered critically endangered by the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2010) and are protected by the Convention on International Trade in Endangered Species (CITES).

The Pacific Ocean leatherback population is generally smaller in size than that in the Atlantic Ocean. Because adult female leatherbacks frequently nest on different beaches, nesting population estimates and trends are especially difficult to monitor. In the Pacific, the IUCN notes that most leatherback nesting populations have declined more than 80 percent. In other areas of

the leatherback's range, observed declines in nesting populations are not as severe, and some population trends are increasing or stable. In the Atlantic, available information indicates that the largest leatherback nesting population occurs in French Guyana, but the trends are unclear. Some Caribbean nesting populations appear to be increasing, but these populations are very small when compared to those that nested in the Pacific less than 10 years ago. Nesting trends on U.S. beaches have been increasing in recent years.

The leatherback sea turtle is the largest turtle and the largest living reptile in the world. Mature turtles can be as long as six and a half feet (2 m) and weigh almost 2,000 lbs (900 kg). The leatherback is the only sea turtle that lacks a hard, bony shell. A leatherback's carapace is approximately 1.5 inches (4 cm) thick and consists of leathery, oil saturated connective tissue overlaying loosely interlocking dermal bones. The carapace has seven longitudinal ridges and tapers to a blunt point. Adult leatherbacks are primarily black with a pinkish white mottled ventral surface and pale white and pink spotting on the top of the head. The front flippers lack claws and scales and are proportionally longer than in other sea turtles; back flippers are paddle-shaped. The ridged carapace and large flippers are characteristics that make the leatherback uniquely equipped for long distance foraging migrations.

4.1.2.1 *Distribution*

Leatherback sea turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India and Sri Lanka and KwaZulu Natal, South Africa.

Leatherback sea turtles have been documented in Alaska waters as far north as approximately 60º latitude (approximately 50 miles north of the northern edge of the Temporary Maritime Activities Area) and as far west in the Gulf of Alaska as the Aleutian Islands (Eckert 1993). In contrast with other sea turtles, leatherback sea turtles have physiological traits that allow for the conservation of body heat which enable them to maintain body core temperatures well above the ambient water temperatures (Eckert 1993; Greer et al. 1973; Pritchard 1971). Shells, or carapaces, of adult leatherbacks are 4 cm (1.5 inches) thick on average, contributing to the leatherback's thermal tolerance that enables this species to forage in water temperatures far lower than the leatherback's core body temperature (Bostrom et al. 2010). In an analysis of

available sightings (Eckert 2002), researchers found that leatherback turtles with carapace lengths smaller than 100 cm (39 inches) were sighted only in waters 79 ºF or warmer, while adults were found in waters as cold as 32 ºF to 59 ºF off Newfoundland (Goff and Lien 1988). As a result, they are more capable of surviving for extended periods of time in cooler waters than the hard-shelled sea turtles (Bleakney 1965; Lazell Jr. 1980).

In the Pacific Ocean, leatherback turtles have the most extensive range of any living reptile and have been reported in all pelagic waters of the Pacific between 71° N and 47° S latitude and in all other major pelagic ocean habitats (NMFS and USFWS 1998). Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Few quantitative data are available concerning the seasonality, abundance, or distribution of leatherbacks in the central northern Pacific Ocean. Satellite tracking studies and occasional incidental captures of the species in the Hawaii-based longline fishery indicate that deep ocean waters are the preferred habitats of leatherback turtles in the central Pacific Ocean (NMFS and USFWS 2007). The primary migration corridors for leatherbacks are across the North Pacific Subtropical Gyre, with the eastward migration route possibly to the north of the westward migration.

4.1.2.1.1 Occurrence in the Gulf of Alaska TMAA

Few quantitative data are available concerning the seasonality, abundance, or distribution of leatherbacks in the North Pacific Ocean. In a review of 126 satellite tag tracks from western Pacfic leatherback sea turtles, Benson et al. (2011) documented Pacific wide movements of these individuals, none of which traveled within or near the action area. The movements of adult leatherback sea turtles appear to be linked to the seasonal availability of their prey and the requirements of their reproductive cycles (Collard 1990; Davenport and Balazs 1991). Leatherbacks prefer convergence zones and upwelling areas in the open ocean, along continental margins, or near large archipelagos. Leatherbacks from both eastern and western Pacific Ocean nesting populations migrate to northern Pacific Ocean foraging grounds (Dutton et al. 1998).

The occurrence of leatherback sea turtles in the Gulf of Alaska is considered uncommon or rare (Hodge and Wing 2000; Wing and Hodge 2002). Little is known about the seasonal occurrence patterns of this species in eastern Pacific waters north of Monterey Bay. McAlpine et al. (2004) suggest that leatherback occurrences off British Columbia are most frequent from July to September and that the species is an uncommon seasonal resident of those waters. It is likely that the same can be said for leatherbacks in Alaskan waters. Since 1960, there have been 19 documented occurrences in Alaska, ranging from Southeast Alaska to the Alaska Peninsula (Wing, B.L., NMFS-AFSC, pers. comm., 25 January 2006; Hodge and Wing 2000). The majority of these occurrences were reported in August during the 1970s and 1980s. Two summer

occurrences have been recorded in the waters near Cordova, located north of the action area and slightly east of Prince William Sound (Stinson 1984).

4.1.2.2 *Abundance Estimate*

Most stocks in the Pacific Ocean are faring poorly, as nesting populations there have declined more than 80 percent since 1982 (Sarti-Martinez 2000), while western Atlantic and South African populations are generally stable or increasing (TEWG 2007). Worldwide, the largest nesting populations now occur off of Gabon in equatorial West Africa (5,865 to 20,499 females nesting per year (Witt et al. 2009), in the western Atlantic in French Guiana (4,500 to 7,500 females nesting per year (Dutton et al. 2007) and Trinidad (estimated 6,000 turtles nesting annually (Eckert 2002), and in the western Pacific in West Papua (formerly Irian Jaya), Indonesia (about 600 to 650 females nesting per year (Dutton et al. 2007). By 2004, 203 nesting beaches from 46 countries around the world had been identified (Dutton 2006). Of these, 89 sites (44 percent) have generated data from beach monitoring programs. Although these data are beginning to form a global perspective, unidentified sites likely exist, and incomplete or no data are available for many known sites. Genetic studies have been used to identify two discrete leatherback populations in the Pacific Ocean (Dutton 2006): an eastern Pacific Ocean population, which nests between Mexico and Ecuador; and a western Pacific Ocean population, which nests in numerous countries, including Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu.

4.1.2.3 *Critical Habitat*

NMFS has designated critical habitat for leatherback sea turtles along the U.S. Pacific Coast (77 FR 4170) in 2012 and the U.S. Virgin Islands in 1998 (44 FR 17710); however these are not located in or near the action area.

4.1.2.4 *Conclusion*

Leatherback sea turtles have seldom been encountered in the Gulf of Alaska (e.g., only 19 sightings of the species in the Gulf of Alaska since 1960), and no data or density estimates are available for this species in the action area. Because of the rarity of this species in the TMAA and because there was no leatherback sea turtle density data available for the TMAA, the Navy did not include this species in the acoustic effects analysis using NAEMO. However, due to their low expected occurrence in the action area and the limited duration of the proposed action each year (i.e., 21 days or less), the species is not expected to co-occur with Navy training activities in the TMAA. Further, the non-impulsive acoustic sources proposed for use in the TMAA are all either mid- or high frequency sources. Since sea turtles detect sound at less than 1,000 Hz (Popper et al. 2014), they would likely not be able to hear or respond to the mid or high frequency non-impulsive acoustic sources proposed for use in the TMAA. Because they would not hear non-impulsive sources, they also would not be susceptible to thresholds shifts from

these sources. For these reasons, the likelihood of a leatherback sea turtle being impact by non-impulsive acoustic stressors is discountable. For explosives, the zone of effects is small (within a few kilometers) and given the low probability of leatherbacks being present within the TMAA, let alone in close enough proximity to an explosion to be adversely affected, the likelihood of a leatherback sea turtle being impacted by an explosion in the TMAA is discountable.

We would also not expect a Navy vessel to strike a leatherback sea turtle in the TMAA. First, as discussed above, leatherback sea turtles are rare in the action area and are not expected to co-occur with Navy activities that take place over a limited amount of time (i.e., 21 days or less) in the TMAA. Second, the Navy implements mitigation measures to avoid striking protected marine species including the use of lookouts. Finally, there has never been a documented case of a Navy vessel striking a leatherback sea turtle in the TMAA. For these reasons, the likelihood of a Navy vessel associated with training activities in the TMAA to strike a leatherback sea turtle is so low as to be discountable.

As discussed above, we have determined that the likelihood of Navy training activities in the TMAA impacting leatherback sea turtles is discountable. This conclusion is largely based on the low abundance of this species in the action area and the low likelihood that any leatherback turtles would occur in the action area during training activities. Therefore, we concur with the Navy's determination of NLAA and have determined that leatherback sea turtles are not likely to be adversely affected by the proposed action. As a result, this species will not be considered further in this opinion.

4.1.3 Chelonid sea turtles

Sea turtles from the Cheloniidae family have been documented in the Gulf of Alaska, but only rarely. Members of the Cheloniidae family (loggerhead, green, olive ridley sea turtles) typically occur in the warm, subtropical areas of the Pacific such as southern California and Hawai′i. Therefore, the Gulf of Alaska is considered beyond their normal range of occurrence because of cold water temperatures. The ocean waters of the TMAA have an average sea surface temperature in summer in the upper 100 m (328 ft) of approximately 51.8 degrees Fahrenheit (°F) (11 degrees Celsius [°C]). Most hard-shell turtles seek optimal seawater temperatures near 65°F and are cold-stressed at seawater temperatures below 50°F (Davenport 1997). At temperatures below 15°C (59°F), green and ridley sea turtles become semidormant, hardly move and come to the surface at intervals up to 3 hours (Milton and Lutz 2003). Loggerhead sea turtles exposed to excessive low temperatures have experienced abrupt failure in pH homeostasis and a sharp increase in blood lactate levels (Milton and Lutz 2003). At 10°C (50°F) loggerhead sea turtles were lethargic and "floated" (Milton and Lutz 2003).

In Alaska, only 9 green sea turtle occurrences, 2 olive ridley occurrences, and 2 loggerheads were documented between 1960 and 2006 (Hodge and Wing 2000; Navy 2006). Most of these sightings involved individuals that were either cold-stressed, likely to become cold-stressed, or

already deceased (Hodge and Wing 2000; McAlpine et al. 2002). Thus, the TMAA is considered to be outside the normal range for sea turtle species of the Cheloniidae family. Because Chelonid sea turtles occur in the Gulf of Alaska only rarely, we do not expect individual Chelonid sea turtles to co-occur with Navy activities in the Gulf of Alaska TMAA. Therefore, the likelihood of Chelonid sea turtles being exposed to Navy stressors is discountable and these species are not likely to be adversely affected by the proposed action. As a result, these species will not be considered further in this opinion.

4.1.4 ESA-listed Salmonids from California

As documented further in Section 6 of this opinion, the only stressor we determined would likely adversely affect ESA-listed fish species was the use of explosive ordnances. The salmonid ESUs and DPSs from California (inclusive of Southern Oregon/Northern California coho ESU) are either not expected to occur in the action area, or are expected to occur in the action area only rarely. Adverse effects to salmon ESUs and DPSs from California are so unlikely to occur as to be considered discountable; therefore, salmon ESUs and DPSs from California are not likely to be adversely affected by Navy explosive training activities in the Gulf of Alaska TMAA. This conclusion is based on our understanding of the migratory patterns of fish from these ESUs and DPSs. In addition to the discussion below, further information on the migratory patterns of ESA-listed salmonids from western North America is included in section 6.7.2 of this opinion.

Weitkamp (2010) examined coded wire-tag recovery data and found that Chinook salmon originating from a particular freshwater region share a common marine distribution. Weitkamp and Neely (2002) reported a similar pattern for coho. Chinook salmon originating from north of Cape Blanco in Oregon tend to migrate towards the Gulf of Alaska, whereas those originating south of Cape Blanco tend to migrate west and south to forage in waters off Oregon and California (PFMC 2014). Weitkamp (2010) found that Chinook originating from southern Oregon and California were generally only recovered off the coast of Oregon and California. Similarly, Masuda et al. (2015a) reported on coded-wire tag recoveries along the west coast of North America and found that of 1,278 Chinook salmon recovered with CWTs, only three individuals were found in the northern Gulf of Alaska. Similar results have been reported for coho salmon from southern Oregon and California. We do not have information to suggest (e.g., coded-wire tag data) that coho salmon or steelhead from California or southern Oregon (south of Cape Blanco) regularly migrate north to Gulf of Alaskan waters in close proximity to the action area. Myers et al. (1996b) did not report any steelhead or coho salmon from California in northern Gulf of Alaska waters in close proximity to the action area. Weitkamp and Neely (2002) also did not document coho from these more southern waters in the northern Gulf of Alaska in close proximity to the action area.

The information presented above suggests that it is unlikely that individuals from these ESUs would occur in the action area. Though the rare individual from these more southern ESUs/DPSs

could migrate to the northern Gulf of Alaska in close proximity to the action area, the vast majority will not. Because of the rarity of such a migratory pattern resulting in an extremely low abundance of individuals from these ESUs/DPSs occurring within the action area, and the infrequent nature of Navy explosive training activities in the Gulf of Alaska TMAA, the likelihood of Navy training activities in the Gulf of Alaska TMAA affecting individuals from these ESUs/DPSs is so low as to be discountable. Therefore, Navy training activities in the Gulf of Alaska TMAA are not likely to adversely affect the following ESA-listed salmon ESUs and steelhead DPSs: Chinook (Central Valley spring-run ESU, California Coastal ESU, Sacramento River winter-run ESU), coho (Southern Oregon Northern California Coast ESU, Central California Coast ESU), and steelhead (Northern California DPS, Central California Coast DPS, California Central Valley DPS, South-Central California Coast DPS, Southern California DPS).

4.1.5 Chinook salmon from Oregon, Washington, and Idaho

Chinook salmon are the largest of any salmon, with adults often exceeding 40 pounds (18 kg); individuals over 120 pounds (54 kg) have been reported. Chinook mature at about 36 inches and 30 pounds. Chinook salmon are blue-green back with silver flanks at sea, with small black spots on both lobes of the tail, and black pigment along the base of the teeth. The Chinook salmon's historical range in North America extended from the Ventura River in California to Point Hope, Alaska. The natural freshwater range for Chinook salmon extends throughout the Pacific Rim of North America. This species has been identified from the San Joaquin River in California to the Mackenzie River in northern Canada (Healey 1991). The oceanic range encompasses Washington, Oregon, California, throughout the north Pacific Ocean, and as far south as the U.S./Mexico border (PFMC 2000b).

ESA-listed Chinook salmon ESUs from Washington, Oregon, and Idaho are expected to occur in the Gulf of Alaska TMAA action area. Further discussion of the occurrence of ESA-listed Chinook salmon in the action area is in section 6.7.2 of this opinion. Each Chinook salmon ESU is treated as a separate species under the ESA (NMFS 2005b). Of the Chinook salmon from Oregon, Washington, and Idaho, one Chinook salmon ESU is endangered (Upper Columbia River spring-run) and five are threatened (Snake River spring/summer-run, Snake River fall-run, Puget Sound, Lower Columbia River, and Upper Willamette River) (70 FR 37160).

As documented further in Section 6 of this opinion, the only stressor we determined would likely adversely affect ESA-listed fish species was the use of explosive ordnances. Using the methodology described in section 6.7.2 of this opinion, we determined that no ESA-listed Chinook salmon from Oregon, Washington, or Idaho would be killed or injured by Navy explosive activities in the Gulf of Alaska TMAA. Therefore, the likelihood of Navy training activities in the Gulf of Alaska TMAA injuring or killing fish from these ESUs is discountable. Also, as documented in 6.7.2.3, we determined that any behavioral responses of ESA-listed salmonids (including those from these ESUs) to explosive ordnance would not create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal

behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, the effect of any instances of behavioral response to Navy explosive activities is insignificant. For these reasons we determined that Chinook salmon from the Upper Columbia River spring-run, Snake River spring/summer-run, Snake River fall-run, Puget Sound, Lower Columbia River, and Upper Willamette River ESUs are not likely to be adversely affected by the proposed action.

4.1.6 Snake River and Ozette Lake Sockeye salmon

Sockeye salmon (*Oncorhynchus nerka*) are the second most abundant of the seven Pacific salmon species. They have silvery sides with a green or blue back and white tips on the ventral and anal fins. Sockeye salmon have no large spots on back or tail, but some may have speckling on the back. They have no silver pigment on the tail, and they have a prominent gold eye color.

Sockeye salmon exhibit a very diverse life history, characteristically using both riverine and lake habitat throughout its range, exhibiting both freshwater resident and anadromous forms. The vast majority of sockeye salmon are anadromous fish that make use of lacustrine habitat for juvenile rearing. These "lake-type" fish typically spawn in the outlet streams of lakes and occasionally in the lakes themselves. Juvenile sockeye salmon will then use the lake environment for rearing for up to 3 years before migrating to sea. After 1 to 4 years at sea, sockeye salmon will return to their natal lake to spawn. Some sockeye, however, spawn in rivers without lake habitat for juvenile rearing. Offspring of these riverine spawners tend to use the lower velocity sections of rivers as the juvenile rearing environment for 1 to 2 years, or may migrate to sea in their first year.

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Sacramento River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia (Burgner 1991a). The largest populations, and hence the most important commercial populations, occur north of the Columbia River. Each sockeye salmon ESU is treated as a separate species under the ESA (NMFS 2005b). There are currently two ESA-listed ESUs of sockeye salmon, one of which is listed as threatened (Ozette Lake) and one of which is listed as endangered (Snake River). Individuals of both the Snake River and the Ozette Lake ESUs are expected to occur in the Gulf of Alaska TMAA. Further discussion of the occurrence of ESA-listed sockeye salmon in the action area is in section 6.7.2 of this opinion.

As documented further in Section 6 of this opinion, the only stressor we determined would likely adversely affect ESA-listed fish species was the use of explosive ordnances. Using the methodology described in section 6.7.2 of this opinion, we determined that no ESA-listed sockeye salmon from the Snake River or Ozette Lake ESUs would be killed or injured by Navy explosive activities in the Gulf of Alaska TMAA. Therefore, the likelihood of Navy training

activities in the Gulf of Alaska TMAA injuring or killing fish from these ESUs is discountable. Also, as documented in 6.7.2.3, we determined that any behavioral responses of ESA-listed salmonids (including those from these ESUs) to explosive ordnance would not create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, the effect of any instances of behavioral response to Navy explosive activities is insignificant. For these reasons we determined that Snake River and Ozette Lake sockeye salmon ESUs are not likely to be adversely affected by the proposed action.

4.1.7 Critical Habitat of North Pacific Right Whale

In April 2008 (73 FR 19000), NMFS clarified that two areas previously designated as critical habitat for right whales in the North Pacific (71 FR 38277) also applied to the listed North Pacific right whale. The areas encompass about 36,750 square miles of marine habitat, which include feeding areas within the Gulf of Alaska (Figure 5) and the Bering Sea that support the species.

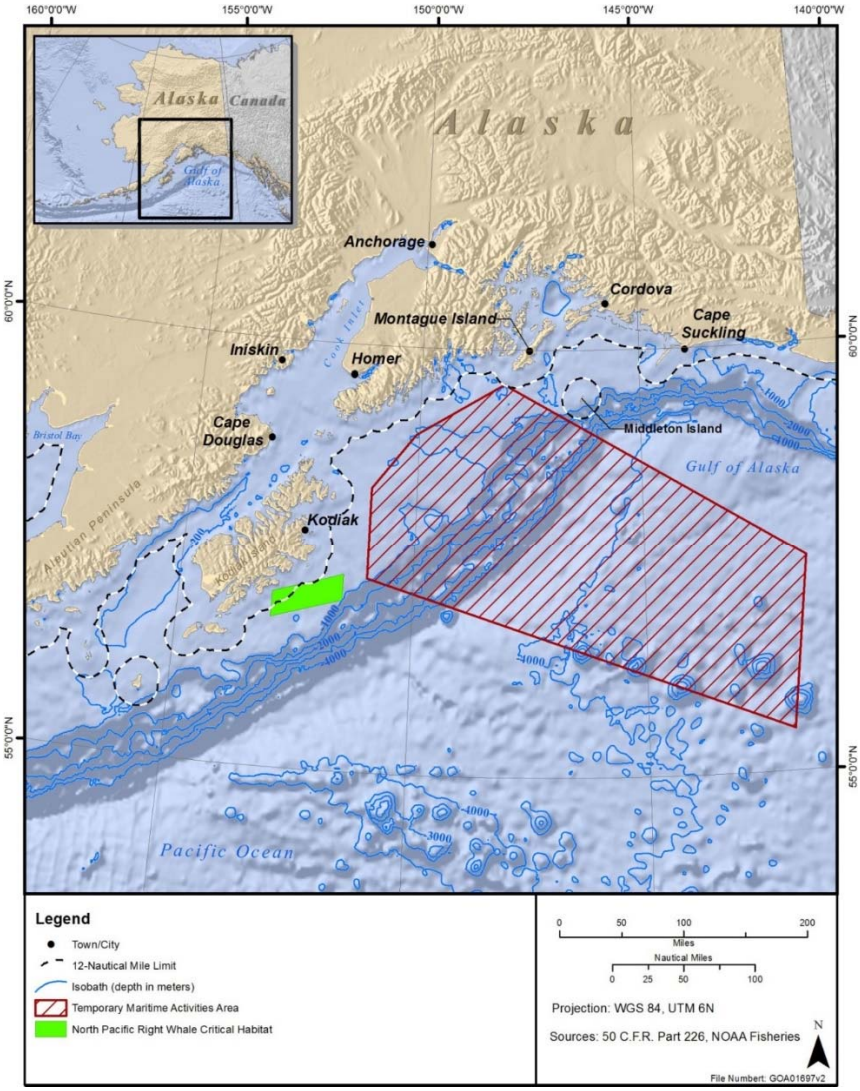


Figure 5. North Pacific Right Whale Critical Habitat and the Gulf of Alaska TMAA.

The TMAA is close to, but does not overlap North Pacific right whale critical habitat in the Gulf of Alaska. The nearest boundary of the Pacific right whale critical habitat is approximately 16 nm (30 km) west of the southwest corner of the TMAA. Sounds from training activities have the potential to reach North Pacific right whale critical habitat in the Gulf of Alaska, but as described further below, are expected to be reduced to negligible levels due to transmission loss by the time it reaches the critical habitat.

As described in the 2016 FSEIS/OEIS, the Navy rarely, if ever, operates near the corners or edge of the TMAA (i.e., areas of the TMAA closest to North Pacific right whale critical habitat). To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The

Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as Visit, Board, Search, and Seizure training events that are without sonar or explosives (Navy 2016a). North Pacific right whale critical habitat is at least 80 to 120 nautical miles away from areas in the TMAA where the majority of Navy sonar would be used (Navy 2016b) indicating that during the majority of training activities using sonar, sound levels would not reach North Pacific right critical habitat at detectable levels. Additionally, explosive use would generally occur great distances from North Pacific right whale critical habitat, in waters off the continental shelf (Navy 2017a) indicating that sound from explosives would also not be expected to reach North Pacific right whale critical habitat at detectable levels..

North Pacific right whale critical habitat is currently characterized by a single primary constituent element, the presence of large zooplankton for feeding; in particular, copepods and a euphausiid whose very large size, high lipid content, and occurrence in the region make it a preferred prey item for right whales (73 FR 19000). We are not aware of any research examing the effects of anthropogenic noise on zooplankton, though research has documented behavioral responses of other invertebrates (i.e., squid, crabs) to anthropogenic noise (McCauley et al. 2000) (Lagardere 1982; Wilson et al. 2007). Even if sound from Navy sonar activities were to reach North Pacific right whale critical habitat, sonar is not anticipated to cause mortality or injury to zooplankton due to the lack of fast rise times, lack of high peak pressures, and the lack of high acoustic impulse of sonar. Even if Navy sonar does reach critical habitat at detectable levels, at most we could only expect zooplankton to exhibit a short term behavioral reaction to the sound, with the animals resuming normal behaviors immediately after the sound exposure was over. We do not expect sound from explosives to reach critical habitat designated for North Pacific right whales because explosive use would generally occur great distances from North Pacific right whale critical habitat, in waters off the continental shelf and sound from explosives does not travel such distances. Since we do not anticipate zooplankton injury or mortality from Navy activities, all zooplankton would still be available to North Pacific right whales during and following any exposure to any sounds from Navy activities. This indicates that Navy activities would not be expected to affect the availability of zooplankton in North Pacific right whale critical habitat. For these reasons, the potential effects of Navy training activities on the essential features of critical habitat designated for North Pacific right whale are insignificant and the

proposed action is not likely to destroy or adversely modify the critical habitat that has been designated for North Pacific right whales. As a result, critical habitat of North Pacific right whales will not be considered further in this opinion.

4.1.8 Critical Habitat of Steller Sea Lion - Western DPS

NMFS designated Steller sea lion critical habitat on August 27, 1993 (58 CFR 45269). Steller sea lion critical habitat in Western Alaska includes a 20 nautical mile buffer around all major haulouts and rookeries, as well as associated terrestrial, air and aquatic zones, and three large offshore foraging areas (Figure 6). Essential features of Steller sea lion critical habitat include the physical and biological habitat features that support reproduction, foraging, rest, and refuge, and include terrestrial, air and aquatic areas. Specific terrestrial areas include major rookeries and haul-outs where breading, pupping, refuge and resting occurs. More than 100 major haulouts are documented. The principal, essential aquatic areas are the nearshore waters around rookeries and haulouts, their forage resources and habitats, and traditional rafting sites. Air zones around terrestrial and aquatic habitats are also designated as critical habitat to reduce disturbance in these essential areas. Specific activities that occur within the habitat that may disrupt the essential life functions that occur there include: (1) wildlife viewing, (2) boat and airplane traffic, (3) research activities, (4) timber harvest, (5) hard mineral extraction, (6) oil and gas exploration, (7) coastal development and pollutant discharge, and others.

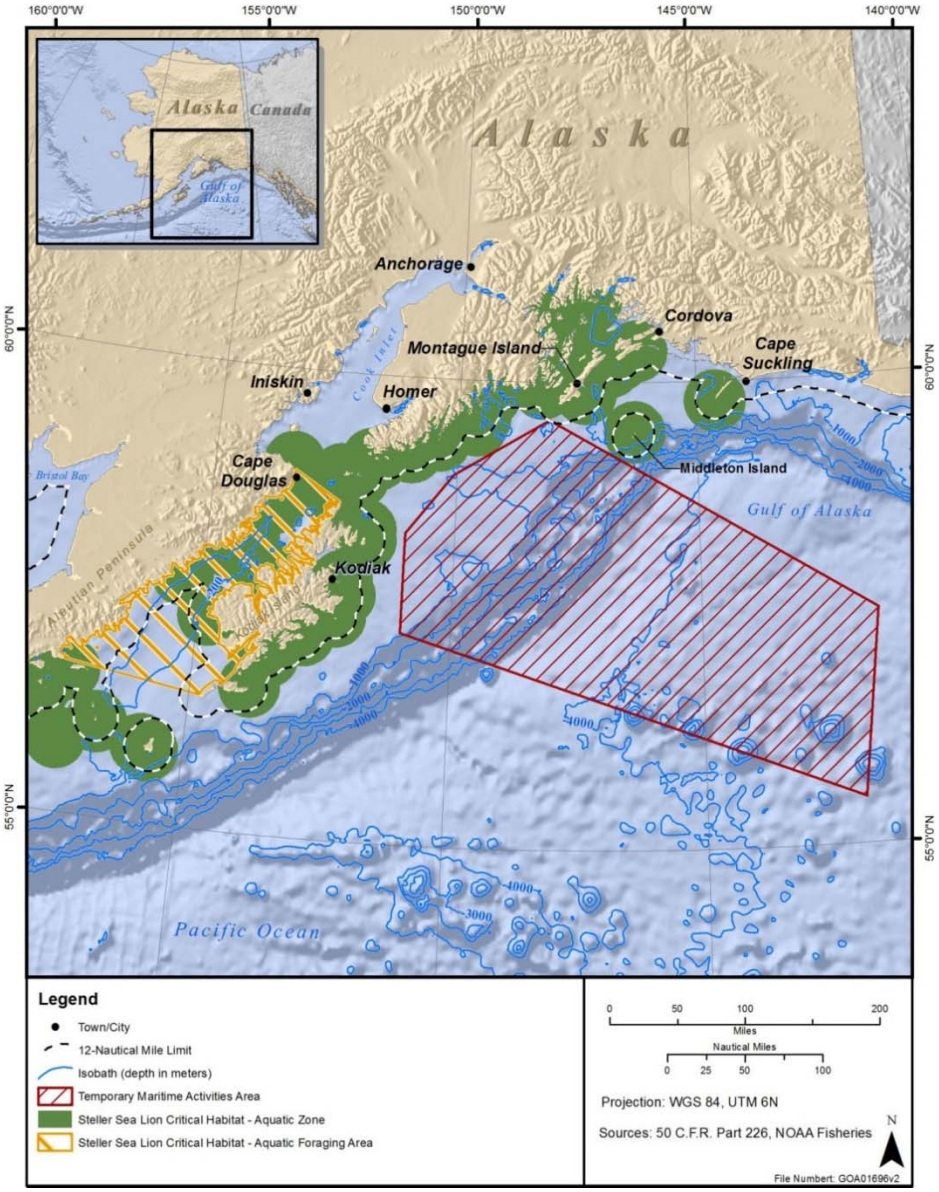


Figure 6. Steller Sea Lion Western DPS Critical Habitat and the Gulf of Alaska TMAA.

The Navy's training activities in the Gulf of Alaska TMAA are outside, but adjacent to, designated critical habitat for the western DPS of Steller sea lion (Figure 6). Sounds from training activities have the potential to reach areas of critical habitat, but as described further below, are expected to be reduced to negligible levels due to transmission loss by the time it reaches the critical habitat. For example, as described in Table 26 in section 6.3.14.4, results from the Navy's acoustic effects model lists a maximum range to effects for surface ship sonar (i.e., the source with the longest propogation range) of 58.7 to 67.4 nautical miles for odontocetes and pinnipeds (e.g., Steller sea lions). At this range, sound levels were estimated to be 132 to 138 dB SPL and only accounted for less than one percent of all behavioral response

exposures of these species. A higher percentage of behavioral response exposures from surface ship sonar were estimated to occur at much closer range (e.g., 86 percent of low-frequency harassment exposures were estimated to occur between 4.8 and 29.1 nautical miles from the source at 156-162 dB SPL). As described in section 6.3.14 of this opinion, sounds from explosives used in the TMAA travel a much shorter distance, with the largest range to effect on Steller sea lions being less than 530 meters.

As described in the 2016 FSEIS/OEIS, the Navy rarely, if ever, operates near the corners or edge of the TMAA (i.e., areas of the TMAA closest to Steller sea lion critical habitat). To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA (i.e., the portion of the TMAA closest to Steller sea lion critical habitat) is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as Visit, Board, Search, and Seizure training events that are without sonar or explosives (Navy 2016a). Steller sea lion critical habitat is at least 80 to 120 nautical miles away from areas in the TMAA where the majority of Navy sonar would be used (Navy 2017b) indicating that we would not expect sound from training activities using sonar to reach Steller sea lion critical habitat at levels that would be expected to disturb Steller sea lions. Additionally, explosive use would generally occur great distances from Steller sea lion critical habitat, in waters off the continental shelf (Navy 2017a).

These assumptions are supported by empirical data as well. Wiggins et al. (2017) recorded underwater ambient and anthropogenic sounds during the 2015 Northern Edge Navy exercise in the TMAA using High-frequency Acoustic Recording Packages (HARPs). HARPs are autonomous, battery-operated instruments capable of recording underwater sounds from 10 Hz to 100 kHz continuously over long periods (up to ~1 year) to provide a comprehensive time series of the marine soundscape. The HARP that was placed closest to nearshore habitats of the Gulf of Alaska during the 2015 Northern Edge exercise detected MFA sonar during the exercise, but received levels never exceeded 144 dB. The majority of detections at this site were at about 120 to 130 dB. Since Steller sea lion critical habitat is located even further away from areas where Navy sonar activities occur, received levels in these habitats would be even lower than 144 dB

due to transmission loss. The HARP located closest to Steller sea lion critical habitat also detected some explosions, but based on their spectral and temporal character, the explosions were likely related to fishing operations such as the use of "seal bombs" as pinniped deterrents (Wiggins et al. 2017).

As outlined above, it is unlikely that sound from Navy training activities will reach Steller sea lion critical habitat at detectable levels or levels that would be expected to cause behavioral disturbance. Therefore, the potential effects of Navy training activities on critical habitat designated for Steller sea lions are insignificant and the proposed action is not likely to destroy or adversely modify the critical habitat that has been designated for Steller sea lions. As a result, critical habitat of Steller sea lions will not be considered further in this opinion.

We note that direct effects to individual Western DPS Steller sea lions in the form of behavioral harassment are considered in section 6 of this opinion.

4.2 Species Considered Further in this Biological Opinion

The rest of this section of our opinion consists of narratives for each of the threatened and endangered species that occur in the Gulf of Alaska TMAA action area and that may be adversely affected by the readiness activities the Navy conducts. In each narrative, we present a summary of information on the distribution and population structure of each species to provide a foundation for the exposure analyses that appear later in this opinion. Then we summarize information on the threats to the species and the species' status given those threats to provide points of reference for the jeopardy determinations we make later in this opinion.

After the *Status* subsection of each narrative, we present information on the diving and social behavior of the different species because that behavior helps determine whether aerial and ship board surveys are likely to detect each species. We also summarize information on the vocalizations and hearing of the different species because that background information lays the foundation for our assessment of how the different species are likely to respond to sounds produced by sonar and detonations.

Additional background information on the status of these species and critical habitat can be found in a number of published documents prepared by NMFS including status reviews and recovery plans. Richardson et al. (1995d) and Tyack (2000), as well as a number of other literature sources cited in the effects analysis of this opinion, provide detailed analyses of the functional aspects of cetacean communication and their responses to active sonar. Finally, Croll et al. (1999), NRC (2005; 2000; 2003c), Martin et al. (2015b), Supin et al. (2011), Tyack (2010), Filadelfo et al. (2009), and Richardson and Wursig (1995), as well as a number of other literature sources cited in the effects analysis of this opinion, provide information on the potential and probable effects of active sonar on the marine animals considered in this opinion.

Many of the Pacific salmonid ESUs and DPSs include fish from artificial propagation programs (hatcheries). Hatchery fish are generally considered to be of less conservation value than individuals from the natural population (NMFS 2015b). ESA take prohibitions do not apply to hatchery fish with clipped adipose fins from threatened ESUs/DPSs.

4.2.1 Blue Whale

The blue whale (*Balaenoptera musculus*) is a cosmopolitan species of baleen whale. It is the largest animal ever known to have lived on Earth: adults in the Antarctic have reached a maximum body length of about 33 m and can weigh more than 150,000 kg. The largest blue whales reported from the North Pacific are a female that measured 26.8 m (88 ft) taken at Port Hobron in 1932 (Reeves et al. 1985) and a 27.1 m (89 ft) female taken by Japanese pelagic whaling operations in 1959 (NMFS 1998).

As is true of other baleen whale species, female blue whales are somewhat larger than males. Blue whales are identified by the following characteristics: a long-body and comparatively slender shape; a broad, flat "rostrum" when viewed from above; a proportionately smaller dorsal fin than other baleen whales; and a mottled gray color pattern that appears light blue when seen through the water.

4.2.1.1 *Distribution*

Blue whales are found along the coastal shelves of North America and South America (Clarke 1980; Donovan 1984; Rice 1998b). In the western North Atlantic Ocean, blue whales are found from the Arctic to at least the mid-latitude waters of the North Atlantic (CETAP 1982; Gagnon and Clark 1993; Wenzel et al. 1988; Yochem and Leatherwood 1985).

Blue whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea. Blue whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska. Nishiwaki (1966) reported that blue whales occur in the Aleutian Islands and in the Gulf of Alaska. An array of hydrophones, deployed in October 1999, detected two blue whale call types in the Gulf of Alaska (Stafford 2003a). Fifteen blue whale sightings off British Columbia and in the Gulf of Alaska have been made since 1997 (Calambokidis et al. 2009b). Three of these photographically verified sightings were in the northern Gulf of Alaska within 71 nm of each other and were less than 100 nm offshore (Calambokidis et al. 2009b).

4.2.1.1.1 Occurrence in the Gulf of Alaska TMAA

Blue whales from the Central North Pacific stock feed in summer off Kamchatka, the Aleutians, and in the Gulf of Alaska, and migrate to lower latitudes in the winter, including the Western

Pacific and to a lesser degree the Central Pacific, including Hawaii (Stafford 2003b; Stafford et al. 2001a).

There were no blue whale sightings during an August 1994 line-transect survey south of the Aleutian Islands that covered waters over the continental shelf, the Aleutian Trench, and the northern portion of the abyssal plains of the Gulf of Alaska (Forney and Brownell Jr. 1996). A large-scale, inter-disciplinary monitoring program for the North Pacific Ocean and the southern Bering Sea, conducted seasonally from June 2002 through October 2004, included surveys of marine birds and mammals. The cruises followed a survey track from British Columbia, Canada, to Hokkaido, Japan, crossing the Gulf of Alaska between roughly 51° N and 55° N. On six separate crossings, covering all seasons and including waters of all depths, no blue whales were seen (Sydeman et al. 2004). There also were no blue whale sightings during the Navy-funded survey of the action area in April 2009 (Rone et al. 2009a). However, during the 2013 GOALS II survey, there were five on-effort visual sightings and three acoustic detections on sonobuouys of blue whales in the action area (Rone et al. 2014). In 2012, four blue whales were observed south of the action area (Matsuoka et al. 2013).

Despite the lack of sighting data, blue whale calls have been acoustically detected in the Gulf of Alaska from mid-July to mid-December, with peak occurrence from August through November (Moore et al. 2006). Calls from the eastern North Pacific population are detected from late July to mid-December, and calls from the western (now central) North Pacific population are detected from mid-July to mid-December (Stafford et al. 2007a). More recently, two Navy-funded HARPs were deployed in the shelf and slope regions of north-central Gulf of Alaska and recordings collected from July 2011 through February 2012 (Baumann-Pickering et al. 2012b). Blue whale calls were detected from both the Eastern North Pacific and Central North Pacific stocks, although calls from the latter were substantially less common. Overall, blue whale calls were detected from the start of HARP deployment in July 2011 through early January 2012, when blue whale calling decreased dramatically (Baumann-Pickering et al. 2012b). A limited number of blue whale D calls were detected between January and April 2014 at two offshore HARPs.The highest number of hours with calls occurred from mid-August until early December, indicating the presence of blue whales in the action area from summer through early winter. Blue whale occurrence in the action area is considered seasonally likely, primarily from June through December (Debich et al. 2013; Rice et al. 2015).

4.2.1.2 ***Population Structure***

At least three subspecies of blue whales have been identified based on body size and geographic distribution (*B. musculus intermedia*, which occurs in the higher latitudes of the Southern Oceans, *B. m. musculus*, which occurs in the Northern Hemisphere, and *B. m. brevicauda* which occurs in the mid-latitude waters of the southern Indian Ocean and north of the Antarctic convergence); however, due to the way this species is listed under the ESA this consultation will

treat them as a single entity. Readers who are interested in these subspecies will find more information in Gilpatrick et al. (1997), Kato et al. (1995), Omura et al. (1970), and Ichihara (1966).

In addition to these subspecies, the International Whaling Commission's Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan 1991), although there is increasing evidence that there may be more than one blue whale population in the Pacific Ocean (Barlow et al. 1995; Gilpatrick Jr. et al. 1997; Mizroch et al. 1984; Ohsumi and Wada 1972). For example, studies of the blue whales that winter off Baja California and in the Gulf of California suggest that these whales are morphologically distinct from blue whales of the western and central North Pacific (Gilpatrick et al. 1997), although these differences might result from differences in the productivity of their foraging areas more than genetic differences (Barlow et al. 1997b; Calambokidis et al. 1990; Sears 1987). A population of blue whales that has distinct vocalizations inhabits the northeast Pacific from the Gulf of Alaska to waters off Central America (Gregr et al. 2000; Mate et al. 1998; Stafford 2003a). We assume that this population is the one affected by the activities considered in this opinion.

Blue whales from both the eastern and western North Pacific have been heard, tracked, or harvested in waters off Kodiak Island; acoustic detections are made in the Gulf of Alaska from mid-July to mid-December and a peak from August through November (COSEWIC 2002; Ivashin and Rovnin. 1967; Moore et al. 2006; Stafford 2003b; Stafford et al. 2007b; Yochem and Leatherwood 1985). Although acoustic detections in the Gulf of Alaska were absent since the late 1960s, recordings have increased during 1999 to 2002 and a few sightings have been made in the northern Gulf of Alaska (Calambokidis et al. 2009a; Moore et al. 2006; NOAA 2004; Stafford 2003b; Stafford et al. 2007b; Stafford and Moore 2005a). However, surveys in the western Gulf of Alaska and east of Kodiak Island have not found blue whales (Rone et al. 2010b; Zerbini et al. 2006b). Blue whales are rarely observed in nearshore Alaskan waters, but seem to prefer continental shelf edge waters; such areas in the Gulf of Alaska were formerly feeding grounds for blue whales prior to severe depletion (Rice and Wolman. 1982). Call detections of blue whales from the western North Pacific indicate a greater likelihood of these individual occurring southwest of Kodiak Island (Stafford 2003b).

4.2.1.3 *Abundance Estimate*

Widespread whaling over the last century is believed to have decreased the blue whale population drastically from its pre-whaling population size (Sirovic et al. 2004). The 2013 GOALS II provided the first abundance estimates for the eastern North Pacific population of blue whales in the central Gulf of Alaska to consist of 78 individuals (D = 0.0005, CV = 1.22; Rone et al. 2014).

Calambokidis et al. (2009a) suggested that when feeding conditions off California are not optimal, blue whales may move to other regions to feed, including waters further north. Calambokidis et al. (2009a) demonstrated that some of the blue whales found in the Gulf of Alaska and off British Columbia are part of the California feeding population. In July 2004, three blue whales were documented in the northern Gulf of Alaska, with photographic confirmation that one of these individuals previously identified off southern California in 1995 and 1998 (Calambokidis et al. 2009a). A comparison of survey data from the 1990s to 2008 indicates that there has been a northward shift in blue whale distribution within waters off California, Oregon, and Washington (Barlow 2010a). Subsequent mark-recapture estimates "indicated a significant upward trend in abundance of blue whales" at a rate of increase just under 3 percent per year for the U.S. west coast blue whale population in the Pacific (Calambokidis et al. 2009c). Consistent with the earlier suggested variability in the distribution patterns, Carretta et al. (2013b) reported that blue whales from the U.S. west coast have been increasingly found feeding to the north and south of the U.S. west coast during summer and fall. Although there has not been evidence to suggest an increase in the eastern North Pacific blue whale population, data provided by Monnahan et al. (2014b) indicate that population may have recovered near to its estimated pre-whaling size.

4.2.1.4 *Natural Threats*

Natural causes of mortality in blue whales are largely unknown, but probably include predation and disease (not necessarily in their order of importance). Blue whales are known to become infected with the nematode *Carricauda boopis* (Baylis 1928), which are believed to have caused fin whales to die as a result of renal failure (Lambertsen 1986); see additional discussion under Fin whales). Killer whales may also prey on blue whales, but this may be rare (Ford and Reeves 2008; Perry et al. 1999a).

4.2.1.5 *Anthropogenic Threats*

Two human activities are known to threaten blue whales; whaling and shipping. Historically, whaling represented the greatest threat to every population of blue whales and was ultimately responsible for listing blue whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing blue, fin, and other large whales using a fairly primitive open-water netting technique (Tonnessen and Johnsen 1982). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species.

From 1889 to 1965, whalers killed about 5,761 blue whales in the North Pacific Ocean (Hill et al. 1999). From 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984). Evidence of a population decline was seen in the catch data from Japan. In 1912, whalers captured 236 blue whales; in 1913, 58 blue whales; in 1914, 123 blue whales;

from 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984). In the eastern North Pacific, whalers killed 239 blue whales off the California coast in 1926. And, in the late 1950s and early 1960s, Japanese whalers killed 70 blue whales per year off the Aleutian Islands (Mizroch et al. 1984).

Although the International Whaling Commission banned commercial whaling in the North Pacific in 1966, Soviet whaling fleets continued to hunt blue whales in the North Pacific for several years after the ban. Surveys conducted in these former-whaling areas in the 1980s and 1990s failed to find any blue whales (Forney and Brownell Jr. 1996). By 1967, Soviet scientists wrote that blue whales in the North Pacific Ocean (including the eastern Bering Sea and Prince William Sound) had been so overharvested by Soviet whaling fleets that some scientists concluded that any additional harvests were certain to cause the species to become extinct in the North Pacific (Latishev 2007). As its legacy, whaling has reduced blue whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push blue whales closer to extinction. Otherwise, whaling currently does not threaten blue whale populations.

In 1980, 1986, 1987, and 1993, ship strikes have been implicated in the deaths of blue whales off California (Barlow 1997). More recently, Berman-Kowalewski et al. (2010) reported that between 1988 and 2007, 21 blue whale deaths were reported along the California coast, typically one or two cases annually. In addition, several photo-identified blue whales from California waters were observed with large scars on their dorsal areas that may have been caused by ship strikes. Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach, and speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than whales that are not feeding (Sears 1983).

Increasing oceanic noise may impair blue whale behavior. Although available data do not presently support traumatic injury from sonar, the general trend in increasing ambient low-frequency noise in the deep oceans of the world, primarily from ship engines, could impair the ability of blue whales to communicate or navigate through these vast expanses (Aburto et al. 1997c; Clark 2006). Blue whales off California altered call levels and rates in association with changes in local vessel traffic (Mckenna 2011). There is a paucity of contaminant data regarding blue whales. Available information indicates that organochlorines, including dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCB), benzene hexachloride (HCH), hexachlorobenzene (HCB), chlordane, dieldrin, methoxychlor, and mirex have been isolated from blue whale blubber and liver samples (Gauthier et al. 1997b; Metcalfe et al. 2004). Contaminant transfer between mother and calf occurs, meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional contaminant loads during life and passing higher loads to the next generation (Gauthier et al.

1997a; Metcalfe et al. 2004). This is supported by ear plug data showing maternal transfer of pesticides and flame retardants in the first year of life (Trumble et al. 2013). These data also support pulses of mercury in body tissues of the male studied (Trumble et al. 2013).

4.2.1.6 ***Status and Trends***

Blue whales (including all subspecies) were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. The 2013-2014 Biennial Report to Congress states the status of the species is considered stable. Blue whales are listed as endangered on the IUCN Red List of Threatened Animals (IUCN 2010). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for blue whales.

It is difficult to assess the current status of blue whales globally because (1) there is no general agreement on the size of the blue whale population prior to whaling and (2) estimates of the current size of the different blue whale populations vary widely. We may never know the size of the blue whale population in the North Pacific prior to whaling, although some authors have concluded that their population numbers about 200,000 animals before whaling. Similarly, estimates of the global abundance of blue whales are uncertain. Since the cessation of whaling, the global population of blue whales has been estimated to range from 11,200 to 13,000 animals (Maser et al. 1981). These estimates, however, are more than 20 years old.

The current best available abundance estimate for the eastern North Pacific population of blue whales that occur in the central Gulf of Alaska is 78 (D = 0.0005, CV = 1.22; Rone et al. 2014).There was a documented increase in the blue whale population size between 1979 and 1994, but there has not been evidence to suggest an increase in the population since then (Barlow 1994; Barlow and Taylor 2001a; Carretta et al. 2010b). In 2008, Cascadia Research conducted photographic identification surveys to make abundance estimates of blue whales along the U.S. West Coast, reflecting an upward trend in abundance of blue whales along the U.S. West Coast (Calambokidis et al. 2009d).

The information available on the status and trend of blue whales do not allow us to reach any conclusions about the extinction risks facing blue whales as a species, or particular populations of blue whales. The possible exception is the eastern North Pacific blue whale population which many not have been subject to as much commercial whaling as other blue whale populations and which may be recovering to a stable population level since the cessation of commercial whaling in 1971 (Campbell et al. 2015; Monnahan et al. 2014a; Monnahan et al. 2014b). With the limited data available on blue whales, we do not know whether these whales exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among

others, that cause their population size to become a threat in and of itself) or if blue whales are threatened more by exogenous threats such as anthropogenic activities (primarily whaling and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate).

4.2.1.7 *Diving and Social Behavior*

Blue whales spend more than 94 percent of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5 to 20 times at 12 to 20 sec intervals before a deep dive of 3 to 30 min (Croll et al. 1999a; Leatherwood et al. 1976; Maser et al. 1981; Yochem and Leatherwood 1985). Average foraging dives are 140 m deep and last for 7.8 min (Croll et al. 2001a). Non-foraging dives are shallower and shorter, averaging 68 m and 4.9 min (Croll et al. 2001a). However, dives of up to 300 m are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 m).

Blue whales occur singly or in groups of two or three (Aguayo 1974; Mackintosh 1965; Nemoto 1964; Pike and Macaskie 1969; Ruud 1956; Slijper 1962). However, larger foraging aggregations, even with other species such as fin whales, are regularly reported (Fiedler et al. 1998; Schoenherr 1991). Little is known of the mating behavior of blue whales.

4.2.1.8 *Vocalization and Hearing*

Blue whales produce prolonged low-frequency vocalizations that include moans in the range from 12.5 to 400 Hz, with dominant frequencies from 16 to 25 Hz, and songs that span frequencies from 16 to 60 Hz that last up to 36 seconds repeated every 1 to 2 minutes (see McDonald et al. 1995). Berchok et al. (2006a) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0 to 78.7 Hz. Reported source levels are 180 to 188 dB re 1μPa, but may reach 195 dB re 1μPa (Aburto et al. 1997b; Clark and Gagnon 2004; Ketten 1998; McDonald et al. 2001a). Samaran et al. (2010) estimated Antarctic blue whale calls in the Indian Ocean at 179 ± 5 dB re 1 μPa_{rms} -1 m in the 17 to 30 Hz range and pygmy blue whale calls at 175 ± 1 dB re 1 μPa_{rms} -1 m in the 17 to 50 Hz range.

As with other baleen whale vocalizations, blue whale vocalization function is unknown, although numerous hypotheses exist (maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources) (Edds-Walton 1997; Payne and Webb. 1971; Thompson et al. 1992). Intense bouts of long, patterned sounds are common from fall through spring in low latitudes, but these also occur less frequently while in summer high-latitude feeding areas. Short, rapid sequences of 30 to 90 Hz calls are associated with socialization and may be displays by males based upon call seasonality and structure. The low-frequency sounds produced by blue whales can, in theory, travel long distances, and it is possible that such long-distance communication occurs (Edds-Walton 1997; Payne and Webb.

1971). The long-range sounds may also be used for echolocation in orientation or navigation (Tyack 1999).

Cetaceans have an auditory anatomy that follows the basic mammalian pattern, with some modifications to adapt to the demands of hearing in the sea. The typical mammalian ear is divided into the outer ear, middle ear, and inner ear. The outer ear is separated from the inner ear by the tympanic membrane, or eardrum. In terrestrial mammals, the outer ear, eardrum, and middle ear function to transmit airborne sound to the inner ear, where the sound is detected in a fluid. Since cetaceans already live in a fluid medium, they do not require this matching, and thus do not have an air-filled external ear canal. The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997b) hypothesized that large mysticetes have acute infrasonic hearing.

Blue whale vocalizations tend to be long (>20 s), low-frequency (<100 Hz) signals (Thomson and Richardson 1995), with a range of 12 to 400 Hz and dominant energy in the infrasonic range of 12 to 25 Hz (Ketten 1998; McDonald et al. 2001b; Mellinger and Clark 2003). Vocalizations are predominantly songs and calls. Blue whale calls have high acoustic energy, with reports of 186 to 188 dB re 1 μPa-m (Cummings and Thompson 1971; McDonald et al. 2001b) and 195 dB re 1 μPa-m (Aburto et al. 1997a) source levels. Calls are short-duration sounds (2 to 5 s) that are transient and frequency-modulated, having a higher frequency range and shorter duration than song units and often sweeping down in frequency (80 to 30Hz), with seasonally variable occurrence.

Blue whale songs consist of repetitively patterned sounds produced over time spans of minutes to hours, or even days (Cummings and Thompson 1971; McDonald et al. 2001b). The songs are divided into pulsed/tonal units, which are continuous segments of sound, and phrases, which are repeated combinations of 1 to 5 units (Mellinger and Clark 2003; Payne and McVay 1971). A song is composed of many repeated phrases. Songs can be detected for hundreds, and even thousands of kilometers (Stafford et al. 1998), and have only been attributed to males (McDonald et al. 2001b; Oleson et al. 2007a). Worldwide, songs are showing a downward shift in frequency (Mcdonald et al. 2009). For example, a comparison of recordings from November 2003 and November 1964 and 1965 reveals a long-term shift in the frequency of blue whale calling near San Nicolas Island. In 2003, the spectral energy peak was 16 Hz compared to ~22.5 Hz in 1964 and 1965, illustrating a more than 30 percent shift in call frequency over four decades (McDonald et al. 2006b). McDonald et al. (2009) observed a 31 percent downward frequency shift in blue whale calls off the coast of California, and also noted lower frequencies in 7 of the

world's 10 known blue whale songs originating in the Atlantic, Pacific, Southern, and Indian Oceans. Many possible explanations for the shifts exist, but none have emerged as the probable cause.

Although general characteristics of blue whale calls are shared in distinct regions (McDonald et al. 2001b; Mellinger and Clark 2003; Rankin et al. 2005; Thompson et al. 1996), some variability appears to exist among different geographic areas (Rivers 1997). Sounds in the North Atlantic have been confirmed to have different characteristics (i.e., frequency, duration, and repetition) than those recorded in other parts of the world (Berchok et al. 2006b; Mellinger and Clark 2003). Clear differences in call structure suggestive of separate populations for the western and eastern regions of the North Pacific have also been reported (Stafford et al. 2001b); however, some overlap in calls from these geographically distinct regions have been observed, indicating that the whales may have the ability to mimic calls (Stafford and Moore 2005b).

Call types from both eastern and western North Pacific blue whale populations have been documented in the Gulf of Alaska (Stafford 2003b). The eastern North Pacific blue whales produce four call types: Types A, B and D (Oleson et al. 2007a; Thompson et al. 1996). A and B-calls are stereotypic calls of the blue whale population (McDonald et al. 2001a; McDonald et al. 2006b) and are produced exclusively by males and associated with mating behavior (Oleson et al. 2007a). These calls have long durations (20 sec) and low frequencies (10 to 100 Hz); they are produced either as repetitive sequences (song) or as singular calls. The B call has a set of harmonic tonals, and may be paired with a pulsed type A call. Blue whale D calls are down-swept in frequency (100 to 40 Hz) with duration of several seconds. These calls are similar worldwide and are associated with feeding animals; they may be produced as call- counter call between multiple animals and heard from both sexes (Oleson et al. 2007c). Calls from western North Pacific blue whales are shorter, consist primarily of frequency-modulated moans, and are typically higher in frequency than the eastern North Pacific blue whales (Stafford et al. 2001a).

Calling rates of blue whales tend to vary based on feeding behavior. Stafford et al. (2005b) recorded the highest calling rates when blue whale prey was closest to the surface during its vertical migration. Wiggins et al. (2005) reported the same trend of reduced vocalization during daytime foraging followed by an increase at dusk as prey moved up into the water column and dispersed. Blue whales make seasonal migrations to areas of high productivity to feed, and vocalize less at the feeding grounds than during migration (Burtenshaw et al. 2004). Oleson et al. (2007d) reported higher calling rates in shallow diving (<100 ft) whales, while deeper diving whales (>165 ft) were likely feeding and calling less.

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (Ketten 1997a; Richardson et al. 1995d). Based on vocalizations and anatomy, blue whales are assumed to predominantly hear low-frequency sounds below 400 Hz

(Croll et al. 2001b) (Croll et al. 2001c; Oleson et al. 2007d; Stafford and Moore 2005b). In terms of functional hearing capability, blue whales belong to the low-frequency group, which have a hearing range of 7 Hz to 22 kHz (Southall et al. 2007b).

Nevertheless, recent studies indicate that blue whales can hear and respond to sounds in the mid-frequency range. Nineteen controlled exposure experiments were conducted on blue whales during the Southern California-10 behavorial response study (Southall et al. 2011a) and 13 in the Southern California-11 behavorial response study (Southall et al. 2012a). Both controlled exposure experiments simulated exposure to Navy MFA sonar. Behavioral response was observed in some blue whales and consisted primarily of small changes in dive behavior and general avoidance of the sound source. Preliminary assessments showed behavior appearing to return to baseline shortly after the transmissions ended, however, it is possible that the changes observed were a direct response to the transmission or some other unknown or un-analyzed factors (Southall et al. 2012a). During other controlled exposure experiments, blue whales responded to a mid-frequency sound source, with a source level between 160 to 210 dB re 1 μPa at 1 m and a received sound level up to 160 dB re 1 μPa, by exhibiting generalized avoidance responses and changes to dive behavior (Goldbogen et al. 2013). However, reactions were temporary and were not consistent across individuals based on received sound levels alone. Results were likely the result of a complex interaction between sound exposure factors such as proximity to sound source and sound type (mid-frequency sonar simulation vs. pseudo-random noise), environmental conditions, and behavioral state. Surface feeding whales did not show a change in behavior during controlled exposure experiments, but deep feeding and non-feeding whales showed temporary reactions that often quickly abated after sound exposure. Distances of the sound source from the whales during controlled exposure experiments were sometimes less than a mile. Melcon et al. (2012) tested whether MFA sonar and other anthropogenic noises in the mid-frequency band affected the "D-calls" produced by blue whales in the Southern California Bight. The likelihood of an animal calling decreased with the increased received level of mid-frequency sonar, beginning at a sound pressure level of approximately 110 to 120 dB re 1 μPa. It is not known whether the lower rates of calling actually indicated a reduction in feeding behavior or social contact since the study used data from remotely deployed, passive acoustic monitoring buoys.

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, blue whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.1.9 *Critical Habitat*

NMFS has not designated critical habitat for blue whales.

4.2.2 North Pacific Right Whale

The North Pacific right whale (*Eubalaena japonica*) remains one of the most endangered whale species in the world, likely numbering fewer than 1,000 individuals between the eastern and western populations. This species is a large baleen whale that grows to between 45 and 55 feet in length and can weigh up to 70 tons. Females tend to be larger than males. Right whales are generally black (some with white belly patches) and stocky-bodied, lack a dorsal fin, and have large heads (about 1/4 of the body length) with strongly-bowed lower lips. Raised patches of rough skin, or callosities are found around their head, and frequently serve to differentiate individuals. Two rows of long, dark baleen plates hang from the upper jaw, with about 225 plates on each side. The tail is broad, deeply notched, and all black with a smooth trailing edge.

4.2.2.1 *Distribution*

Very little is known of the distribution of right whales in the North Pacific and very few of these animals have been seen in the past 20 years. North Pacific right whales occur in subpolar to temperate waters. They are generally migratory, with at least a portion of the population moving between summer feeding grounds in temperate or high latitudes and winter calving areas in warmer waters (Clapham et al. 2004b; Kraus et al. 1986). Historical whaling records provide virtually the only information on North Pacific right whale distribution (Gregr 2011). This species historically occurred across the Pacific Ocean north of 35° N, with concentrations in the Gulf of Alaska, eastern Aleutian Islands, south-central Bering Sea, Okhotsk Sea, and the Sea of Japan (Omura et al. 1969); (Scarff 1986a); (Clapham et al. 2004b); (Shelden et al. 2005); (Gregr 2011); (Ivashchenko et al. 2013a).

Presently, sightings are extremely rare, occurring primarily in the Okhotsk Sea and the eastern Bering Sea (Brownell Jr. et al. 2001) (Shelden et al. 2005) (Shelden and Clapham 2006) (Wade et al. 2006); (Zerbini et al. 2010a). Recent eastern sightings tend to occur over the continental shelf, although acoustic monitoring has identified whales over abyssal waters (Mellinger et al. 2004b; Sirovic et al. 2015). Some more southerly records also indicate species occurrence along Hawaii, California, Washington, and British Columbia (Herman et al. 1980; Scarff 1986b; Sirovic et al. 2015). However, records from Mexico and California may suggest historical wintering grounds in offshore southern North Pacific latitudes (Brownell Jr. et al. 2001; Gregr and Coyle. 2009).

Current information on the seasonal distribution and migration of right whales is spotty. Historical concentrations of sightings in the Bering Sea together with some recent sightings

indicate that this region, together with the Gulf of Alaska, may represent an important summer habitat for eastern North Pacific right whales (Brownell Jr. et al. 2001; Clapham et al. 2004a; Goddard and Rugh 1998a; Scarff 1986b; Shelden et al. 2005). North Pacific right whales summered in the North Pacific and southern Bering Sea from April or May to September, with a peak in sightings in coastal waters of Alaska in June and July (Klumov 1962; Maury 1852; Omura 1958; Omura et al. 1969; Townsend 1935). The summer range of the North Pacific right whale extended north of the Bering Strait (Omura et al. 1969), and occurrence in the Bering Sea during summer appears to be strongly influenced by the occurrence and abundance of the copepod *Calanus marshallae* (Baumgartner et al. 2013).

Fall and spring distribution was the most widely dispersed, with whales occurring in mid-ocean waters and extending from the Sea of Japan to the eastern Bering Sea. In winter, right whales have been found in the Ryukyu Islands (south of Kyushu, Japan), the Bonin Islands, the Yellow Sea, and the Sea of Japan. Whalers never reported winter calving areas in the North Pacific and where calving occurs remains unknown (Clapham et al. 2004a; Gregr and Coyle. 2009; Scarff 1986b). North Pacific right whales probably migrate north from lower latitudes in spring and may occur throughout the North Pacific from May through August north of 40º N from marginal seas to the Gulf of Alaska and Bering Sea, although absence from the central North Pacific has been argued due to inconsistencies in whaling records (Clapham et al. 2004c; Josephson et al. 2008). This follows generalized patterns of migration from high-latitude feeding grounds in summer to more temperate, possibly offshore waters, during winter (Braham and Rice 1984; Clapham et al. 2004a; Scarff 1986b).

Habitat preference data are sparse for North Pacific right whales as well. Sightings have been made with greater regularity in the western North Pacific, notably in the Okhotsk Sea, Kuril Islands, and adjacent areas (Brownell Jr. et al. 2001). In the western North Pacific, feeding areas occur in the Okhotsk Sea and adjacent waters along the coasts of Kamchatka and the Kuril Islands (IWC 2001). It has been suggested that North Pacific right whales have shifted their preferred habitat as a result of reduced population numbers, with oceanic habitat taking on a far smaller component compared to shelf and slope waters (Shelden et al. 2005). The area where North Pacific right whales are densest in the Gulf of Alaska is between 150 and 170° W and south to 52° N (Shelden and Clapham 2006), but present occurrence there is very rare (Wade et al. 2011b). However, four sightings were made from 2004 to 2006 off Kodiak Island in association with high zooplankton concentrations (Wade et al. 2011b). A right whale was sighted southeast of Kodiak Island in July 1998 and acoustic detections have been made off Kodiak Island, although no detections occurred from April to August 2003 or in April 2009 (Munger et al. 2008; Rone et al. 2010b; Waite et al. 2003a).

4.2.2.2 *Occurrence in the Gulf of Alaska TMAA*

Habitat modeling using historic whaling records suggests that the Gulf of Alaska currently provides suitable habitat for North Pacific right whales, although this has not been validated (Gregr 2011). Presently, sightings are extremely rare, occurring primarily in the Okhotsk Sea and the eastern Bering Sea (Brownell Jr. et al. 2001); (Shelden et al. 2005); (Wade et al. 2006); (Zerbini et al. 2010a). Recently, there are far fewer sightings of North Pacific right whales in the Gulf of Alaska than the Bering Sea (Brownell Jr. et al. 2001; Wade et al. 2011c; Zerbini et al. 2010b). In the summers of 2008 and 2009, satellite transmitters were deployed on four North Pacific right whales on the Bering Sea feeding grounds, and the results demonstrated that the movements of these animals were restricted to a relatively small region between 56° N and 58° N and 163° and 167° W in the Bering Sea (Zerbini et al. 2010a). From the 1960s through 2002, there were only two documented sightings of North Pacific right whales in the Gulf of Alaska. In March 1979, there was an opportunistic sighting near Yakutat Bay in the eastern Gulf of Alaska (Shelden et al. 2005). A single North Pacific right whale was sighted southeast of Kodiak Island in July 1998 during an aerial survey and, subsequently, two passive acoustic recorders were placed in the northern Gulf of Alaska near Kodiak Island (Waite et al. 2003a). Recordings from these instruments, and an additional five placed in the central Gulf of Alaska in 2000 to 2001, were later analyzed for North Pacific right whale calls. Very few right whale calls were positively identified, and all were detected on the westernmost recorder in the Gulf of Alaska during August and September (Moore et al. 2006).

From 2004 to 2006, there were an additional four sightings of North Pacific right whales in the Gulf of Alaska, all in the Barnabus Trough region on Albatross Bank, southeast of Kodiak Island (Wade et al. 2011a; Wade et al. 2011c). These sightings triple the number of sightings in the Gulf of Alaska over the last 40 years and suggest that this area represents important habitat for the remaining animals in this population (Wade et al. 2011a). A portion of this area, located to the west/southwest of the action area, was designated as critical habitat in 2006. Zerbini et al. (2010a) documented fine-scale localized small scale movements in the eastern Bering Sea between July and October based on satellite tag tracking of four North Pacific right whales.

During a marine mammal survey in July 2012, a lone North Pacific right whale was seen approximately 40 miles (mi.) south of the TMAA in deep water, approximately 130 mi. east of Kodiak Island (Matsuoka et al. 2013). In July 2013, during the Gulf of Alaska Line Transect (GOALS) II survey, three North Pacific right whales were acoustically detected in the Barnabus Trough region on Albatross Bank, southeast of Kodiak Island (Rone et al. 2014). This is the same area as the 2004 to 2006 sightings noted above (Wade et al. 2011a; Wade et al. 2011c). Over 2 days between June and August 2013, a Navy-funded passive acoustic monitoring device on Quinn seamount detected 3 hours of North Pacific right whale calls (Debich et al. 2013). Given the recording device location near the southwest border of the action area, inability of the

device as configured to determine call directionality, and likely signal propagation of several tens of miles, it remains uncertain if the detected calls originated within or outside of the TMAA. Previous related Navy-funded monitoring at multiple sites within the action area reported no North Pacific right whale detections from 2011 to 2015 (Baumann-Pickering et al. 2012c; Debich et al. 2013; Rice et al. 2015), but calls were detected in 2013 during two days (21 June and 3 August) from a device located at Quinn Seamount (Sirovic et al. 2015).

Ferguson et al. (2015) used existing published and unpublished data (i.e., from aerial-, land-, and vessel-based surveys; satellite-tagging data; passive acoustic monitoring; traditional ecological knowledge; photo- and genetic-identifica-tion data; whaling data, including catch and sighting locations and stomach contents; prey studies; and anecdotal information from fishermen) identify Biologically Important Areas (BIAs) for several cetacean species in the U.S. waters of the Gulf of Alaska. The feeding area identified by Ferguson et al. (2015) in the Gulf of Alaska overlaps slightly with the Gulf of Alaska TMAA's southwestern corner (Figure 1). This feeding area is applicable from June to September so there is temporal overlap with the proposed Navy training but there is minimal (<1 percent) spatial overlap between this feeding area and the Gulf of Alaska TMAA. It is worth noting here that, as described in section 2.3.3.2, the Navy has agreed to restrict the types of activities that will occur in the area of the TMAA that overlaps with the BIA.

Given their current extremely low population numbers, and the general lack of sightings in the Gulf of Alaska, the occurrence of right whales in the action area is considered rare. North Pacific right whales have not been visually observed inside the action area since at least the 1960s.

4.2.2.3 *Population structure and abundance estimate*

NMFS currently recognizes two stocks of North Pacific right whale: (1) an eastern North Pacific stock; and (2) a western North Pacific stock, thought to feed primarily in the Sea of Okhotsk (Allen and Angliss 2013). It is assumed that any North Pacific right whale in the action area would be from the eastern North Pacific stock.

Based upon mark-recapture studies, estimates of abundance suggested eastern North Pacific right whales in the Bering Sea and Aleutian Islands numbered 31 (95% CL 23 to 54, CV=0.22) and 28 (95% CL 24 to 42), respectively and composed of eight females and 20 males (Wade et al. 2011b). The most recent minimum population estimate of the eastern North Pacific right whale is 25.7 (Muto et al. 2016).

Abundance estimates and other vital rate indices in both the eastern and western North Pacific are not well established. Previous estimates of the size of the right whale population in the Pacific Ocean ranged from a low of 100 to 200 to a high of 220 to 500 (Berzin and Yablokov 1978; Braham and Rice 1984). Although Hill and Demaster (1998b) argued that it is not possible

to reliably estimate the population size or trends of right whales in the North Pacific, Reeves et al. (2003) and Brownell Jr. et al. (2001) concluded that North Pacific right whales in the eastern Pacific Ocean exist as a small population of individuals while the western population of right whales probably consists of several hundred animals; although Clapham et al. (2005) placed this population at likely under 100 individuals, Wade et al. (2011a) estimated 25 to 38 individuals, and Marques et al. (2011) estimated 25 animals. Brownell Jr. et al. (2001) reviewed sighting records and also estimated that the abundance of right whales in the western North Pacific was likely in the low hundreds.

4.2.2.4 *Natural threats*

Right whales have been subjects of killer whale attacks and, because of their robust size and slow swimming speed, tend to fight killer whales when confronted (Ford and Reeves 2008). Similarly, mortality or debilitation from disease and red tide events are not known, but have the potential to be significant problems in the recovery of right whales because of their small population size.

4.2.2.5 *Anthropogenic threats*

Whaling for North Pacific right whales was discontinued in 1966 with the IWC whaling moratorium. However, North Pacific right whales remain at high risk of extinction. Demographic stressors include but are not limited to the following: (1) life history characteristics such as slow growth rate, long calving intervals, and longevity; (2) distorted age structure of the population and reduced reproductive success; (3) strong depensatory or Allee effects; (4) habitat specificity or site fidelity; and (5) habitat sensitivity. The proximity of the other known right whale habitats to shipping lanes (e.g. Unimak Pass) suggests that collisions with vessels may also represent a threat to North Pacific right whales (Elvin and Hogart 2008).

Climate change may have a dramatic effect on survival of North Pacific right whales. Right whale life history characteristics make them very slow to adapt to rapid changes in their habitat (see Reynolds et al. 2002). They are also feeding specialists that require exceptionally high densities of their prey (e.g., Baumgartner and Mate 2003b). Zooplankton abundance and density in the Bering Sea has been shown to be highly variable, affected by climate, weather, and ocean processes and in particular ice extent (Baier and Napp 2003; Napp and G.L. Hunt 2001). The largest concentrations of copepods occurred in years with the greatest southern extent of sea ice (Baier and Napp 2003). It is possible that changes in ice extent, density, and persistence may alter the dynamics of the Bering Sea shelf zooplankton community and in turn affect the foraging behavior and success of right whales.

The recovery plan for North Pacific right whales released in 2013 indicated that though the main direct anthropogenic threat to the species was addressed by the International Whaling

Commission's 1982 moratorium on commercial whaling, several potential threats remain. Among the current threats are environmental contaminants; reduced prey abundance or location due to climate change; ship collisions; and exposure to anthropogenic noise, particularly from the use of the Arctic for energy development and commercial maritime traffic (NMFS 2013d). The recovery plan states that the most significant threat to the eastern population is the extremely small population size, posing a heightened risk for biological extinction if individuals are removed from the population (NMFS 2013d).

4.2.2.6 *Status and trends*

Right whales have been listed as endangered under the ESA since its passage in 1973 (35 FR 8495; June 2, 1970). The NMFS 2013-2014 Biennial Report to Congress states the status of the North Pacific right whale is considered unknown. The North Pacific right whale was originally listed as endangered as a part of the Northern right whale, or *Eubalaena spp*., which has been listed as endangered under the precursor to the ESA and under the ESA since its inception in 1973 (35 FR 8495; June 2, 1970). The original listing included both the North Atlantic and the North Pacific 'populations', although subsequent genetic studies conducted by Rosenbaum (2000) resulted in strong evidence that the North Atlantic and North Pacific right whales are separate species. Following a comprehensive status review, NMFS concluded that Northern right whales are indeed two separate species. On December 27, 2006, NMFS published two proposed rules to list these species separately as North Atlantic and North Pacific right whales (71 FR 77704 and 71 FR 77694). The final rule published on March 6, 2008 (73 FR 12024). North Pacific right whales are also listed as "threatened with extinction," or Appendix I, of the CITES. Their IUCN Redlist status is "endangered" or very high risk of extinction.

Very little is known about right whales in the eastern North Pacific, which were severely depleted by commercial whaling in the 1800s (Brownell Jr. et al. 2001). At least 11,500 individuals were taken by American whalers in the early- to mid-19th century, but harvesting continued into the 20th century (Best 1987). Illegal Soviet whaling took 661 individuals between 1962 and 1968, with 529 from the eastern North Pacific and 152 from the Okhotsk Sea, mostly of large mature individuals (Brownell Jr. et al. 2001; Ivashchenko and Clapham 2012; Ivashchenko et al. 2013b). In the last several decades there have been markedly fewer sightings due to a drastic reduction in number, caused by illegal Soviet whaling in the 1960s (Doroshenko 2000).

Scientists participating in a recent study utilizing acoustic detection and satellite tracking identified 17 right whales (10 males and 7 females) in the Bering Sea, which is almost threefold the number seen in any previous year in the last four decades (Wade et al. 2006). These sightings increased the number of individual North Pacific right whales identified in the genetic catalog for the eastern Bering Sea to 23. Amidst the uncertainty of the eastern North Pacific right whale's future, the discovery of females and calves gives hope that this endangered population may still

possess the capacity to recover (Wade et al. 2006). Available age composition of the North Pacific right whale population indicates most individuals are adults (Kenney 2002). Length measurements for two whales observed off California suggest at least one of these whales was not yet sexually mature and two calves have been observed in the Bering Sea (Carretta et al. 1994; Wade et al. 2006). However, to date, there is no evidence of reproductive success (i.e., young reared to independence) in the eastern North Pacific. No data are available for the western North Pacific.

4.2.2.7 *Diving and social behavior*

The maximum diving depth of the North Pacific right whale is unknown, however other species of right whales can dive as deep as 175 (Baumgartner and Mate 2003a) to 300 meters (Mate et al. 1992) for feeding. Baumgartner and Mate (2003a) reported right whale feeding dives were characterized by a rapid descent from the surface to a particular depth between 80 and 175 meters (262 to 574 ft). These animals would remain at those depths for 5 to 14 minutes, then ascend quickly to the surface. Longer surface intervals have been observed for reproductively active females and their calves (Baumgartner and Mate 2003a). In the Great South Channel, average diving durations is close to 2 minutes, with depths averaging 7.3 meters and reaching a maximum of 85.3 meters (Winn et al. 1995). In the U.S. Outer Continental Shelf, the average diving durations were about 7 minutes (CETAP 1982).

In the North Pacific Ocean, sightings have been of single animals or pairs; however, groups numbering six to ten individuals have been sighted in the northeastern Pacific Ocean (Goddard and Rugh 1998b).

4.2.2.8 *Feeding*

North Pacific right whales forage from the surface to accessible depths (Gregr and Coyle. 2009). Stomach contents from North Pacific right whales indicate copepods and, to a lesser extent, euphausiid crustaceans are the whales' primary prey (Omura et al. 1969). North Pacific right whales have also been observed feeding on coccolithophore blooms (Tynan et al. 2001). Their diet is likely more varied than North Atlantic right whales, likely due to the multiple blooms of different prey available in the North Pacific from January through August (Gregr and Coyle. 2009). Based upon trends in prey blooms, it is predicted that North Pacific right whales may shift from feeding offshore to over the shelf edge during late summer and fall (Gregr and Coyle. 2009). North Pacific right whales, due to the larger size of North Pacific copepods, have been proposed to be capable to exploit younger age classes of prey as well as a greater variety of species. Also as a result, they may require prey densities that are one-half to one-third those of North Atlantic right whales (Gregr and Coyle. 2009). Right whales feed, sometimes at the surface, by continuously filtering prey through their baleen while moving, mouth agape, through patches of planktonic crustaceans. Right whales are believed to rely on a combination of

experience, matrilinear learning, and sensing of oceanographic conditions to locate prey concentrations in the open ocean (Gregr and Coyle. 2009; Kenney 2001).

4.2.2.9 *Vocalization and hearing*

Right whales vocalize to communicate over long distances and for social interaction, including communication apparently informing others of prey path presence (Biedron et al. 2005; Tyson and Nowacek 2005). Vocalization patterns amongst all right whale species are generally similar, with six major call types: scream, gunshot, blow, up call, warble, and down call (McDonald and Moore 2002; Parks and Tyack 2005). A large majority of vocalizations occur in the 300-600 Hz range with up- and down sweeping modulations (Vanderlaan et al. 2003). Vocalizations below 200 Hz and above 900 Hz are rare (Vanderlaan et al. 2003). Calls tend to be clustered, with periods of silence between clusters (Vanderlaan et al. 2003). Gunshot bouts last 1.5 hours on average and up to seven hours (Parks et al. 2012a). Blows are associated with ventilation and are generally inaudible underwater (Parks and Clark 2007). Up calls are 100 to 400 Hz (Gillespie and Leaper 2001). Gunshots appear to be a largely or exclusively male vocalization (Parks et al. 2005b). Smaller groups vocalize more than larger groups and vocalization is more frequent at night (Matthews et al. 2001). Moans are usually produced within 10 m of the surface (Matthews et al. 2001). Up calls were detected year-round in Massachusetts Bay except July and August and peaking in April (Mussoline et al. 2012). Individuals remaining in the Gulf of Maine through winter continue to call, showing a strong diel pattern of up call and gunshot vocalizations from November through January possibly associated with mating (Bort et al. 2011; Morano et al. 2012; Mussoline et al. 2012). Estimated source levels of gunshots in non-surface active groups are 201 dB re 1 μPa p-p (Hotchkin et al. 2011). While in surface active groups, females produce scream calls and males produce up calls and gunshot calls as threats to other males; calves (at least female calves) produce warble sounds similar top their mothers' screams (Parks et al. 2003; Parks and Tyack 2005). Source levels for these calls in surface active groups range from 137 to 162 dB rms re: 1 μPa-m, except for gunshots, which are 174 to 192 dB rms re: 1 μPa-m (Parks and Tyack 2005). Up calls may also be used to reunite mothers with calves (Parks and Clark 2007). Atlantic right whales shift calling frequencies, particularly of up calls, as well as increase call amplitude over both long and short term periods due to exposure to vessel noise (Parks and Clark 2007; Parks et al. 2005a; Parks et al. 2007b; Parks et al. 2011; Parks et al. 2010; Parks et al. 2012b; Parks et al. 2006), particularly the peak frequency (Parks et al. 2009). North Atlantic right whales respond to anthropogenic sound designed to alert whales to vessel presence by surfacing (Nowacek et al. 2003; Nowacek et al. 2004b).

No direct measurements of right whale hearing have been undertaken (Parks and Clark 2007). Models based upon right whale auditory anatomy suggest a hearing range of 10 Hz to 22 kHz (Parks et al. 2007c).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. For the purposes of this analysis, North Pacific right whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.2.10 *Critical habitat*

See Section 4.1.7 on North Pacific Right whale critical habitat.

4.2.3 Humpback Whale – Mexico and Western North Pacific DPSs

Humpback whales (*Megaptera novaeangliae*) are distinguished from other whales in the same Family (Balaenopteridae) by extraordinarily long flippers (up to 5 m or about 1/3 total body length), a more robust body, fewer throat grooves (14 to 35), more variable dorsal fin, and utilization of very long (up to 30 min.), complex, repetitive vocalizations (songs) (Payne and McVay 1971) during courtship. Their grayish-black baleen plates, approximately 270 to 440 on each side of the jaw, are intermediate in length (6,570 cm) to those of other baleen whales. Humpbacks in different geographical areas vary somewhat in body length, but maximum recorded size is 18 m (Winn and Reichley 1985). The whales are generally dark on the back, but the flippers, sides and ventral surface of the body and flukes may have substantial areas of natural white pigmentation plus acquired scars (white or black). Researchers distinguish individual humpbacks by the apparently unique black and white patterns on the underside of the flukes as well as other individually variable features (Glockner and Venus 1983; Katona and Whitehead 1981; Kaufman and Osmond 1987).

On September 8, 2016, NMFS revised the ESA listing for humpback whales to identify 14 DPSs, listing one as threatened, four as endangered, and identify nine others as not warranted for listing (81 FR 40870). Humpback whales from the threatened Mexico DPS, endangered Western North Pacific DPS, and Hawaii DPS, which was identified as not warranted for listing, could all occur in the action area.

4.2.3.1 *Distribution and Occurrence in the Gulf of Alaska TMAA*

Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In both regions, humpback whales tend to occupy shallow, coastal waters. However,

migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985). In the North Pacific Ocean, the summer range of humpback whales includes coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Tomlin 1967, Nemoto 1957, Johnson and Wolman 1984 as cited in NMFS 1991). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter. Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992b; Tynan et al. 2005).

Humpback whales have been known to occur within the Gulf of Alaska primarily in summer and fall, migrating to southerly breeding grounds in winter and returning to the north in spring (Calambokidis et al. 2008). However, based on recordings from moored hydrophones deployed in six locations in the Gulf of Alaska from October 1999 to May 2002, humpback calls were most commonly detected during the fall and winter (Stafford et al. 2007a). More recently, High-frequency Acoustic Recording Packages (HARPs) deployed in the shelf and slope regions of the action area confirmed that some humpbacks remain in the area throughout the winter (Baumann-Pickering et al. 2012c; Debich et al. 2013). Based on both sighting data and acoustic detections, some humpback whales are known to occur year-round in the Gulf of Alaska, although they occur in higher numbers during summer (Baumann-Pickering et al. 2012c; Debich et al. 2013; Stafford et al. 2007a). Humpback whale occurrence in the action area during the summer time period is considered likely. Ferguson et al. (2015) identified areas around Kodiak Island west of the TMAA as a Biologically Important Area for humpback whale feeding (Figure 7).

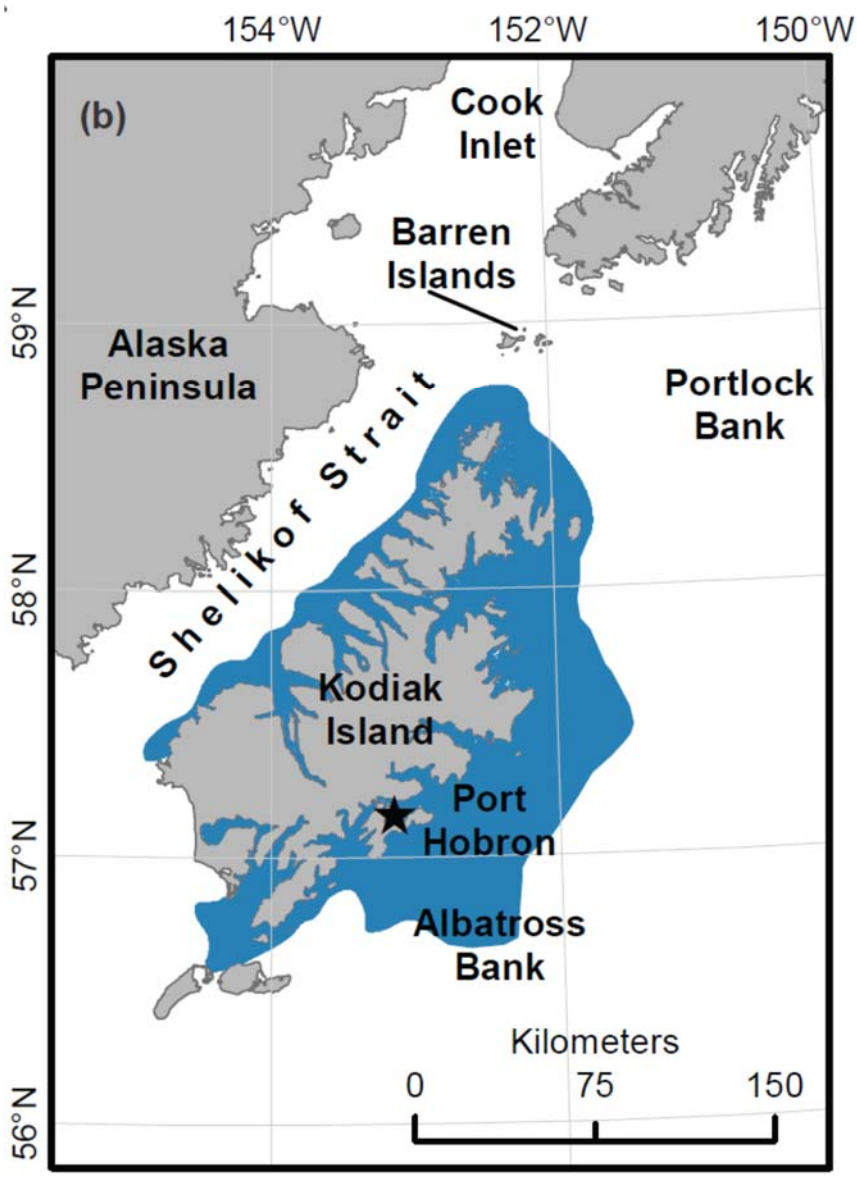


Figure 7. Humpback whale feeding area identified by Ferguson et al. (2015) around Kodiak Island in the Gulf of Alaska. The feeding area occurs northwest of the TMAA.

Identifications made between feeding and wintering areas indicate that the majority of humpbacks in the Gulf of Alaska winter in Hawaii, with the remainder wintering in Mexican waters around the Revillagigedo Islands, Baja, and the Mexican mainland (Barlow et al. 2011; Calambokidis et al. 2008). This suggests that whales migrating between breeding areas in Hawaii and feeding areas in northern British Columbia and southeast Alaska must cross paths with whales migrating between breeding areas near Mexico's offshore islands and feeding areas in the Gulf of Alaska (Barlow et al. 2011). Mexico DPS humpback whales breed along the Pacific coast of mainland Mexico, the Baja California Peninsula, and the Revillagigedos Islands. Whales

from this DPS feed across a broad geographic range from California to the Aleutian Islands (Betteridge et al. 2015). Western North Pacific DPS humpback whales breed and winter in the area of Okinawa and the Pillipines, another unidentified breeding area, and those transiting the Ogasawara area. Whales from this DPS migrate to feeding grounds in the northern Pacific, primarily off the coast of Russia (Betteridge et al 2015). A small number of animals from this DPS migrate to the northern Gulf of Alaska (Calambokidis et al. 2008). While this population structure has been identified, Calambokidis et al. (1997) observed that individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure.

There were eight on-effort humpback whale sightings during the Navy-funded line-transect survey of the action area in April 2009, and only one of these sightings was in the offshore stratum in waters deeper than 2,000 m (Rone et al. 2009a). Results from a recent study of humpback whales in the Gulf of Alaska suggest that there may be regional feeding aggregations within the Gulf of Alaska (Witteveen et al. 2011). This study confirmed that humpback whale feeding aggregations exhibit high site fidelity and indicated that, while inshore and offshore aggregations of humpbacks off Kodiak Island and southeastern Alaska represent single feeding aggregations, inshore and offshore whale aggregations off Prince William Sound may be unique (Witteveen et al. 2011).

4.2.3.2 *Diving and Social Behavior*

Maximum diving depths are approximately 170 m, with a very deep dive (240 m) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1 to 5.1 min in the north Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). Because most humpback prey is likely found within 300 m of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen et al. 2008).

During the feeding season, humpback whales form small groups that occasionally aggregate on concentrations of food that may be stable for long-periods of times. Humpbacks use a wide variety of behaviors to feed on various small, schooling prey including krill and fish (Hain et al. 1982; Hain et al. 1995; Jurasz and Jurasz 1979; Weinrich et al. 1992). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994; Clapham 1996; Tyack 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995).

Humpback whales feed on pelagic schooling euphausiids and small fish including capelin, herring and mackerel. Like other large mysticetes, they are a "lunge feeder" taking advantage of dense prey patches and engulfing as much food as possible in a single gulp. They also blow nets, or curtains, of bubbles around or below prey patches to concentrate the prey in one area, then lunge with open mouths through the middle. Dives appear to be closely correlated with the depths of prey patches, which vary from location to location. In the north Pacific (southeast Alaska), most dives were of fairly short duration (<4 min) with the deepest dive to 148 m (Dolphin 1987), while whales observed feeding on Stellwagen Bank in the North Atlantic dove to <40 m (Hain et al. 1995). Hamilton et al. (1997) tracked one possibly feeding whale near Bermuda to 240 m depth.

4.2.3.3 *Vocalization and Hearing*

Humpback whale vocalization is much better understood than is hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop et al. 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144 to 174 dB (Au et al. 2006a; Au et al. 2000b; Frazer and Mercado III 2000; Richardson et al. 1995d; Winn et al. 1970). Males also produce sounds associated with aggression, which are generally characterized as frequencies between 50 Hz to 10 kHz and having most energy below 3 kHz (Silber 1986; Tyack 1983). Such sounds can be heard up to 9 km away (Tyack 1983). Other social sounds from 50 Hz to 10 kHz (most energy below 3 kHz) are also produced in breeding areas (Richardson et al. 1995d; Tyack 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25 to 89 Hz), and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz) which can be very loud (175 to 192 dB re 1 μPa at 1 m) (Au et al. 2000b; Erbe 2002a; Payne 1985; Richardson et al. 1995d; Thompson et al. 1986b). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995d).

Humpback whales are known to produce three classes of vocalizations: (1) "songs" in the late fall, winter, and spring by solitary males; (2) social sounds made by calves (Zoidis et al. 2008) or within groups on the wintering (calving) grounds; and (3) social sounds made on the feeding grounds (Thomson and Richardson 1995). The best-known types of sounds produced by humpback whales are songs, which are thought to be reproductive displays used on breeding grounds only by adult males (Clark and Clapham 2004; Gabriele and Frankel. 2002; Helweg et al. 1992; Schevill et al. 1964; Smith et al. 2008). Singing is most common on breeding grounds during the winter and spring months, but is occasionally heard in other regions and seasons (Clark and Clapham 2004; Gabriele and Frankel. 2002; McSweeney et al. 1989). Au et al. (2000a) noted that humpbacks off Hawaii tended to sing louder at night compared to the day. There is geographical variation in humpback whale song, with different populations singing a basic form of a song that is unique to their own group. However, the song evolves over the

course of a breeding season, but remains nearly unchanged from the end of one season to the start of the next (Payne et al. 1983). The song is an elaborate series of patterned vocalizations that are hierarchical in nature, with a series of songs ('song sessions') sometimes lasting for hours (Payne and McVay 1971). Components of the song range from below 20 Hz up to 4 kHz, with source levels measured between 151 and 189 dB re 1 µPa-m and high-frequency harmonics extending beyond 24 kHz (Au et al. 2006a; Winn et al. 1970).

Social calls range from 20 Hz to 10 kHz, with dominant frequencies below 3 kHz (D'Vincent et al. 1985b; Dunlop et al. 2008; Silber 1986; Simao and Moreira 2005). Female vocalizations appear to be simple; Simao and Moreira (2005) noted little complexity.

"Feeding" calls, unlike song and social sounds are a highly stereotyped series of narrow-band trumpeting calls. These calls are 20 Hz to 2 kHz, less than 1 second in duration, and have source levels of 162 to 192 dB re 1 µPa-m (D'Vincent et al. 1985b; Thompson et al. 1986b). The fundamental frequency of feeding calls is approximately 500 Hz (D'Vincent et al. 1985a; Thompson et al. 1986a). The acoustics and dive profiles associated with humpback whale feeding behavior in the northwest Atlantic has been documented with Digital Acoustic Recording Tags (DTAGs[3]) (Stimpert et al. 2007). Underwater lunge behavior was associated with nocturnal feeding at depth and with multiple bouts of broadband click trains that were acoustically different from toothed whale echolocation: Stimpert et al. (2007) termed these sounds "mega-clicks" which showed relatively low received levels at the DTAGs (143 to 154 dB re 1 µPa), with the majority of acoustic energy below 2 kHz.

Humpback whale audiograms using a mathematical model based on the internal structure of the ear estimate sensitivity is from 700 Hz to 10 kHz, with maximum relative sensitivity between 2 kHz and 6 kHz (Ketten and Mountain 2014). Previously mentioned research by Au et al. (2001) and Au et al. (2006b) off Hawaii indicated the presence of high-frequency harmonics in vocalizations up to and beyond 24 kHz. While recognizing this was the upper limit of the recording equipment, it does not demonstrate that humpbacks can actually hear those harmonics, which may simply be correlated harmonics of the frequency fundamental in the humpback whale song. The ability of humpbacks to hear frequencies around 3 kHz may have been demonstrated in a playback study. Maybaum (1990) reported that humpback whales showed a mild response to a handheld sonar marine mammal detection and location device with frequency of 3.3 kHz at 219 dB re 1µPa-m or frequency sweep of 3.1 kHz to 3.6 kHz (although it should be noted that this system is significantly different from the Navy's hull mounted sonar). In addition, the system

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[3] *DTAG is a novel archival tag, developed to monitor the behavior of marine mammals, and their response to sound, continuously throughout the dive cycle. The tag contains a large array of solid-state memory and records continuously from a built-in hydrophone and suite of sensors. The sensors sample the orientation of the animal in three dimensions with sufficient speed and resolution to capture individual fluke strokes. Audio and sensor recording is synchronous so the relative timing of sounds and motion can be determined precisely Johnson, M. P., and P. L. Tyack. 2003. A digital acoustic recording tag for measuring the response of wild marine mammals to sound. IEEE Journal of Oceanic Engineering 28(1):3-12.*

had some low frequency components (below 1 kHz) which may have been an artifact of the acoustic equipment. This possible artifact may have affected the response of the whales to both the control and sonar playback conditions.

Results of studies on blue whales (Goldbogen et al. 2013; Southall et al. 2011a), which have similar auditory physiology compared to humpback whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. In terms of functional hearing capability humpback whales belong to low-frequency cetaceans which have a hearing range of 7 Hz to 22 kHz (Southall et al. 2007b).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, humpback whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.3.4 *Status and trends*

Humpback whales were originally listed as endangered in 1970 (35 FR 18319). In September 2016, NMFS revised the ESA listing for humpback whales to identify 14 DPSs, listing one as threatened, four as endangered, and identify nine others as not warranted for listing (81 FR 40870).

A large-scale photo-identification sampling study of humpback whales was conducted from 2004 to 2006 throughout the North Pacific (Barlow et al. 2011; Calambokidis et al. 2008). Known as the SPLASH (Structure of Populations, Levels of Abundance, and Status of Humpbacks) Project, the study was designed to sample all known North Pacific feeding and breeding populations. Overall humpback whale abundance in the North Pacific based on the SPLASH Project was estimated at 21,808 individuals (CV = 0.04), confirming that this population of humpback whales has continued to increase and is now greater than some pre-whaling abundance estimates (Barlow et al. 2011). The 2015 humpback whale status review estimated a growth rate for the North Pacific population of 4.9% (Betteridge et al. 2015).

Within the Gulf of Alaska, the abundance estimate for humpback whales is estimated to be 2,089 (CV = 0.09) animals and includes whale from the Hawaii DPS (89%), Mexico DPS (10.5%), and Western North Pacific DPS (0.5%[4]) (NMFS 2016h; Wade et al. 2016a).

Humpback whale abundance for the Mexico DPS based on the SPLASH project totals 6,000 to 7,000 individuals (Calambokidis et al. 2008). The Final Rule to designate humpback whale DPSs provided an updated abundance estimate for the Mexico DPS of 3,264 individuals (81 FR 62259; Wade et al. 2016b). The status review noted that while a reliable, quantitative estimate of a population growth rate for the Mexico DPS is unavailable, evidence indicates that abundance is not likely declining and likely growing at a rate of 4.9% or higher (Betteridge et al. 2015). The Final Rule to designate humpback whale DPSs provided an abundance estimate for the Western North Pacific DPS of 1,059 individuals (81 FR 62259; Wade et al. 2016b). The status review referenced a population growth rate of 6.9% for the Western North Pacific DPS (Calambokidis et al. 2008; Betteridge et al. 2015), but also noted that this estimate could be biased upwards. The status review noted that overall recovery of this population seems to be slower than in the Central and Eastern North Pacific. The final rule to designate humpback whale DPSs concluded that the population trend of Western North Pacific DPS humpback whales is unknown (81 FR 62259).

4.2.3.5 ***Threats***

Natural sources and rates of mortality of humpback whales are not well known. Based upon prevalence of tooth marks, attacks by killer whales appear to be highest among humpback whales migrating between Mexico and California, although populations throughout the Pacific Ocean appear to be targeted to some degree (Steiger et al. 2008). Juveniles appear to be the primary age group targeted. Humpback whales engage in grouping behavior, flailing tails, and rolling extensively to fight off attacks. Calves remain protected near mothers or within a group and lone calves have been known to be protected by presumably unrelated adults when confronted with attack (Ford and Reeves 2008).

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999b). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period.

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[4] For the endangered Western North Pacific DPS, NMFS chose the upper limit of the 95% confidence interval from the Wade et al. (2016) estimate in order to be conservative due the their status.

Human activities are known to threaten humpback whales. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for listing several species as endangered, but this threat has largely been curtailed. No whaling occurs within the range of Mexico DPS humpbacks, but some "commercial bycatch whaling" has been documented in both Japan and South Korea (within the range of Western North Pacific DPS humpbacks) (Betteridge et al. 2015). Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries in those two provinces between 1969 and 1990, of which 94 died (Lien 1994; Perkins and Beamish 1979). NMFS estimates that between 2002 and 2006, there were incidental serious injuries to 0.2 humpback annually in the Bering Sea/Aleutian Islands sablefish longline fishery. However, NMFS does not consider this estimation reliable because observers have not been assigned to a number of fisheries known to interact with the Central and western North Pacific stocks of humpback whale. In addition, the Canadian observation program is also limited and uncertain (Angliss and Allen 2009). More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997b).

The 2015 humpback whale status review identified underwater noise from human activity as a threat and suggested that exposure is likely chronic and at relatively high levels (Betteridge et al. 2015). However, the authors noted that overall population-level effects of exposure to underwater noise are not well-established. Sources of underwater noise identified in the status review include commercial and recreational vessel traffic, and activities in U.S. Navy training and testing ranges.

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (Gauthier et al. 1997b). Higher PCB levels have been observed in Atlantic waters versus Pacific waters along the United States and levels tend to increase with individual age (Elfes et al. 2010). Although humpback whales off Southern California tend to have the highest PCB concentrations of all North Pacific humpback whales, overall levels are on par with other baleen whales, which are generally lower than odontocete cetaceans (Elfes et al. 2010). As with blue whales, these contaminants are transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe et al. 2004). Available information does not suggest contaminant levels in humpback whales are having a significant impact on their persistence (Elfes et al. 2010).

4.2.3.6 *Critical Habitat*

The NMFS has not designated critical habitat for humpback whales.

4.2.4 Fin Whale

The fin whale, (*Balaenoptera physalus)* is a well-defined, cosmopolitan species of baleen whale (Gambell 1985a). Fin whales have two recognized subspecies: *Balaenoptera physalus physalus* occurs in the North Atlantic Ocean while *B. p. quoyi* (Fischer 1829) occurs in the Southern Ocean. Fin whales are the second-largest whale species by length. Fin whales are long-bodied and slender, with a prominent dorsal fin set about two-thirds of the way back on the body. The streamlined appearance can change during feeding when the pleated throat and chest area becomes distended by the influx of prey and seawater, giving the animal a tadpole-like appearance. The basic body color of the fin whale is dark gray dorsally and white ventrally, but the pigmentation pattern is complex. The lower jaw is gray or black on the left side and creamy white on the right side. This asymmetrical coloration extends to the baleen plates as well, and is reversed on the tongue. Individually distinctive features of pigmentation, along with dorsal fin shapes and body scars, have been used in photo-identification studies (Agler et al. 1990). Fin whales live 70 to 80 years (Kjeld 1982).

4.2.4.1 *Distribution*

Fin whales are distributed widely in every ocean except the Arctic Ocean. In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985a). The overall distribution may be based on prey availability. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

4.2.4.1.1 Occurrence in the Gulf of Alaska TMAA

In previous years, fin whales have been acoustically detected in the Gulf of Alaska year-round, with highest call occurrence rates from August through December and lowest call occurrence rates from February through July (Moore et al. 2006; Stafford et al. 2007a). More recently, two Navy-funded HARPs were deployed in the shelf and slope regions of north-central Gulf of Alaska and recordings collected from July 2011 through February 2012 (Baumann-Pickering et al. 2012b). Fin whale calls were recorded at both sites during all months, with a peak in calling from late August until the end of December. From up to five Navy-funded HARPs deployed in 2014 to 2015 from the shelf to southern seamounts within the Gulf of Alaska, similar 20-Hz and 40-Hz call patterns and seasonality peaks (September to December) were reported (Rice et al. 2015). In 2013 and 2014, 20 Hz fin whale calls associated with singing and call-countercall among animals was the dominant fin whale detection (Debich et al. 2014). Peaks in 20 Hz calling occurred from September to December 2013. Fin whale 40 Hz calls were frequently

detected from June through December 2013 and again from late February to May 2014 (Debich et al. 2014). Debich et al. (2014) went on to hypothesize that the different peaks in 20 Hz and 40 Hz calls may represent distinct behavioral states associated with these call types.

There were 20 on-effort fin whale sightings (56 total animals) during the Navy-funded line-transect survey of the action area in April 2009; animals were distributed in both the inshore and offshore strata (Rone et al. 2009b). During a 2012 survey in summer and early fall, Matsuoka et al. (2013) reported 149 fin whale sightings of 210 individuals. These sightings were made across both shelf and offshore strata within and adjacent to the Gulf of Alaska. During the June through July 2013 GOALS II survey, there were 172 on-effort fin whale sightings of 317 total animals (Rone et al. 2014). Fin whale occurrence in the action area during the summer time period is considered likely. Ferguson et al. (2015) identified areas around Kodiak Island north and west of the TMAA as a Biologically Important Area for fin whale feeding (Figure 8).

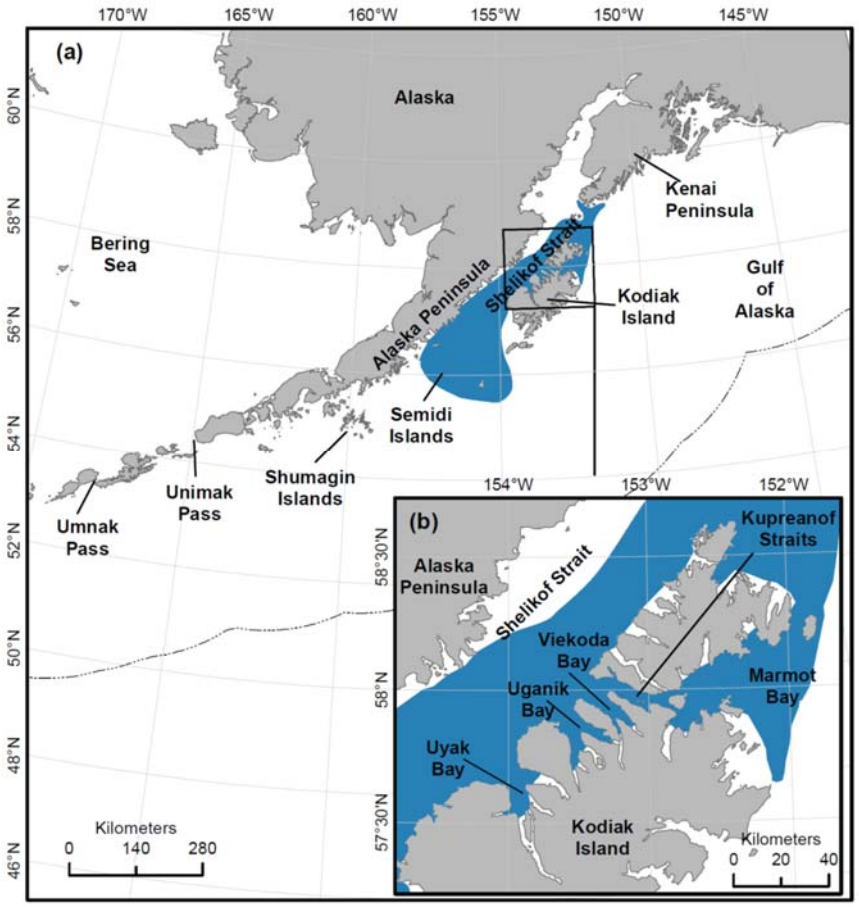

Figure 8. Fin whale feeding area identified by Ferguson et al. (2015) around Kodiak Island in the Gulf of Alaska. The feeding area occurs northwest of the TMAA.

4.2.4.2 *Population Structure*

In U.S. waters, fin whales have been divided into four stocks for management purposes: Hawaii, Western North Atlantic, Alaska (Northeast Pacific), and California/Oregon/Washington (NMFS 2015). The Alaska (Northeast Pacific) stock occurs in the action area.

In the North Pacific Ocean, the International Whaling Commission recognizes two "stocks": (1) East China Sea and (2) rest of the North Pacific (Donovan 1991). However, Mizroch et al. (1984) concluded that there were five possible "stocks" of fin whales within the North Pacific based on histological analyses and tagging experiments: (1) East and West Pacific that intermingle around the Aleutian Islands; (2) East China Sea; (3) British Columbia; (4) Southern-Central California to Gulf of Alaska; and (5) Gulf of California.

Hatch (2004) reported that fin whale vocalizations among five regions of the eastern North Pacific were heterogeneous: the Gulf of Alaska, the northeast North Pacific (Washington and British Columbia), the southeast North Pacific (California and northern Baja California), the Gulf of California, and the eastern tropical Pacific.

4.2.4.3 *Abundance Estimate*

Currently there are no reliable population estimates for the entire Alaska/Northeast Pacific stock of fin whales. The current best available minimum abundance estimate for the fin whale population west of the Kenai Peninsula is 1,368 from 2008 and 2010 surveys, however an estimate for the entire stock remains unknown (Allen and Angliss 2014). Zerbini et al. (2006a) have provided evidence of an increasing abundance trend for fin whales in Alaskan waters. Survey estimate numbers for these stocks are considered to be an underestimate because large whales that could not be identified in the field (due to distance, bad sighting conditions, etc.) were recorded in these and other surveys as "unidentified rorqual" or "unidentified large whale" (Carretta et al. 2010a). A recent study indicates that the abundance of fin whales in waters off the U.S. west coast has increased during the 1991 to 2008 survey period, most likely from *in situ* population growth combined with distribution shifts (Moore and Barlow 2011).

The current best available abundance estimate of fin whales in California, Oregon, and Washington waters is 3,051 (CV = 0.18) and 58 (CV = 1.12) for Hawaii (Carretta et al. 2014). Fin whales of the north Pacific appear to be increasing in abundance although the trend is unclear or declining throughout the rest of their range (NMFS 2011c). Sirovic et al. (2015) used passive acoustic monitoring of fin whale calls to estimate the spatial and seasonal distribution of fin whales in the Southern California Bight. An increase in the number of calls detected between 2006 and 2012 suggest that the population of fin whales off the U.S. west coast may be increasing.

4.2.4.4 *Natural Threats*

Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggested annual natural mortality rates might range from 0.04 to 0.06 for northeast Atlantic fin whales. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure and may be preventing some fin whale populations from recovering (Lambertsen 1983). Adult fin whales engage in flight responses (up to 40 km/h) to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Killer whale or shark attacks may also result in serious injury or death in very young and sick individuals (Perry et al. 1999b).

4.2.4.5 *Anthropogenic Threats*

Fin whales have undergone significant exploitation, but are currently protected under the IWC. Fin whales are still hunted in subsistence fisheries off West Greenland. In 2004, five males and six females were killed, and two other fin whales were struck and lost. In 2003, two males and four females were landed and two others were struck and lost (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. However, the scientific recommendation was to limit the number killed to four individuals until accurate populations could be produced (IWC 2005).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (Carretta et al. 2007; Douglas et al. 2008b; Lien 1994; Perkins and Beamish 1979; Waring et al. 2007). Between 1969 and 1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979). In 1999, one fin whale was reported killed in the Gulf of Alaska pollock trawl fishery and one was killed the same year in the offshore drift gillnet fishery (Angliss and Outlaw 2005; Carretta and Chivers. 2004). According to Waring et al. (2007), four fin whales in the western North Atlantic died or were seriously injured in fishing gear, while another five were killed or injured as a result of ship strikes between January 2000 and December 2004.

Jensen and Silber (2004) review of the NMFS's ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26 percent of the recorded ship strikes [n = 75/292 records]), with most collisions occurring off the east coast, followed by the west coast of the U.S. and Alaska/Hawai′i. Between 1999 to 2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole et al. 2005; Nelson et al. 2007). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al. 2008b). Similarly, 2.4 percent of living fin whales from the Mediterranean show ship strike injury and 16 percent of stranded individuals were killed by

vessel collision (Panigada et al. 2006). There are also numerous reports of ship strikes off the Atlantic coasts of France and England (Jensen and Silber 2004a).

The organochlorines DDE, DDT, and PCBs have been identified from fin whale blubber, but levels are lower than in toothed whales due to the lower level in the food chain that fin whales feed at (Aguilar and Borrell 1988; Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983; Marsili and Focardi 1996). Females contained lower burdens than males, likely due to mobilization of contaminants during pregnancy and lactation (Aguilar and Borrell 1988; Gauthier et al. 1997b). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males (Aguilar and Borrell 1988).

Climate change also presents a potential threat to fin whales, particularly in the Mediterranean Sea, where fin whales appear to rely exclusively upon northern krill as a prey source. These krill occupy the southern extent of their range and increases in water temperature could result in their decline and that of fin whales in the Mediterranean Sea (Gambaiani et al. 2009).

4.2.4.6 ***Status and Trends***

Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. The 2013-2014 Biennial Report to Congress states the status of the species is considered unknown as of 2011 (NMFS 2014). Although fin whale population structure remains unclear, various abundance estimates are available. Pre-exploitation fin whale abundance is estimated at 464,000 individuals worldwide; the estimate for 1991 was roughly 25 percent of this (Braham 1991). Historically, worldwide populations were severely depleted by commercial whaling, with more than 700,000 whales harvested in the twentieth century (Cherfas 1989).

The status and trend of fin whale populations is largely unknown. Over 26,000 fin whales were harvested between 1914 and 1975 (Braham 1991 as cited in Perry et al. 1999b). NMFS estimates roughly 3,000 individuals occur off California, Oregon, and Washington based on ship surveys in summer/autumn of 1996, 2001, and 2005, of which estimates of 283 and 380 have been made for Oregon and Washington alone (Barlow 2003; Barlow and Taylor 2001b; Forney 2007).

Fin whales were extensively hunted in coastal waters of Alaska as they congregated at feeding areas in the spring and summer (Mizroch et al. 2009). There has been little effort in the Gulf of Alaska since the cessation of whaling activities to assess abundance of large whale stocks. Fin whale calls have been recorded year-round in the Gulf of Alaska, but are most prevalent from August-February (Moore et al. 1998; Moore et al. 2006).

Regardless of which of these estimates, if any, have the closest correspondence to the actual size and trend of the fin whale population, all of these estimates suggest that the global population of fin whales consists of tens of thousands of individuals.

Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, fin whales appear to exist at population sizes that are large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself). As a result, we assume that fin whales are likely to be threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) than endogenous threats caused by the small size of their population.

Nevertheless, based on the evidence available, the number of fin whales that are recorded to have been killed or injured in the past 20 years by human activities or natural phenomena, does not appear to be increasing the extinction probability of fin whales, although it may slow the rate at which they recover from population declines that were caused by commercial whaling.

4.2.4.7 *Diving and Social Behavior*

The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5 to 20 shallow dives, each of 13 to 20 s duration, followed by a deep dive of 1.5 to 15 min (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale's most common dives last 2 to 6 min (Hain et al. 1992; Watkins 1981c). The most recent data support average dives of 98 m and 6.3 min for foraging fin whales, while non-foraging dives are 59 m and 4.2 min (Croll et al. 2001a). However, Lafortuna et al. (1999) found that foraging fin whales have a higher blow rate than when traveling. Foraging dives in excess of 150 m are known (Panigada et al. 1999). In waters off the U.S. Atlantic Coast, individuals or duos represented about 75 percent of sightings during the Cetacean and Turtle Assessment Program (Hain et al. 1992). Individuals or groups of less than five individuals represented about 90 percent of the observations.

4.2.4.8 *Vocalization and Hearing*

Fin whales produce a variety of low-frequency sounds in the 10 Hz to 200 Hz range (Edds 1988; Thompson et al. 1992; Watkins 1981b; Watkins et al. 1987). Typical vocalizations are long, patterned pulses of short duration (0.5 to 2 s) in the 18 Hz to 35 Hz range, but only males are known to produce these (Clark et al. 2002; Patterson and Hamilton 1964). Richardson et al. (1995d) reported the most common sound as a 1 second vocalization of about 20 Hz, occurring

in short series during spring, summer, and fall, and in repeated stereotyped patterns in winter. Au (Au and Green 2000b) reported moans of 14 Hz to 118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34 Hz 150 Hz, and songs of 17 Hz to 25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981b). Source levels for fin whale vocalizations are 140 to 200 dB re 1μPa-m (see also Clark and Gagnon 2004; as compiled by Erbe 2002b). The source depth of calling fin whales has been reported to be about 50 m (Watkins et al. 1987).

Although their function is still in doubt, low-frequency fin whale vocalizations travel over long distances and may aid in long-distance communication (Edds-Walton 1997; Payne and Webb. 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999).

The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997b) hypothesized that large mysticetes have acute infrasonic hearing. In a study using computer tomography scans of a calf fin whale skull, Cranford and Krysl (2015) found sensitivity to a broad range of frequencies between 10 Hz and 12 kHz and a maximum sensitivity to sounds in the 1 kHz to 2 kHz range.

Direct studies of fin whale hearing have not been conducted, but it is assumed that fin whales can hear the same frequencies that they produce (low) and are likely most sensitive to this frequency range (Ketten 1997a; Richardson et al. 1995d).

Fin whales produce a variety of low-frequency (< 1 kHz) sounds, but the most typically recorded is a 20 Hz pulse lasting about 1 second, and reaching source levels of 189 ± 4 dB re 1 μPam (Charif et al. 2002; Clark et al. 2002; Edds 1988; Richardson et al. 1995d; Sirovic et al. 2007; Watkins 1981b; Watkins et al. 1987). These pulses frequently occur in long sequenced patterns, are down swept (e.g., 23 to 18 Hz), and can be repeated over the course of many hours (Watkins et al. 1987). In temperate waters, intense bouts of these patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high-latitude feeding areas (Clarke and Charif 1998). The seasonality and stereotypic nature of these vocal sequences suggest that they are male reproductive displays (Watkins 1981b; Watkins et al. 1987); a notion further supported by recent data linking these vocalizations to male fin whales only (Croll et al. 2002). In Southern California, the 20 Hz pulses are the dominant fin whale call type associated both with call-counter-call between multiple animals and with singing (Navy 2010; Navy 2012). An additional fin whale sound, the 40 Hz call described by Watkins (1981b), was also frequently recorded, although these calls are not as common as the 20 Hz fin whale pulses. Seasonality of

the 40 Hz calls differed from the 20 Hz calls, since 40 Hz calls were more prominent in the spring, as observed at other sites across the northeast Pacific (Sirovic et al. 2012). Source levels of Eastern Pacific fin whale 20-Hz calls has been reported as 189 +/- 5.8 dB re 1uPa at 1m (Weirathmueller et al. 2013). Although acoustic recordings of fin whales from many diverse regions show close adherence to the typical 20 Hz bandwidth and sequencing when performing these vocalizations, there have been slight differences in the pulse patterns, indicative of some geographic variation (Thompson et al. 1992; Watkins et al. 1987).

Responses to conspecific sounds have been demonstrated in a number of mysticetes, and there is no reason to believe that fin whales do not communicate similarly (Edds-Walton 1997). The low-frequency sounds produced by fin whales have the potential to travel over long distances, and it is possible that long-distance communication occurs in fin whales (Edds-Walton 1997; Payne and Webb. 1971). Also, there is speculation that the sounds may function for long range echolocation of large-scale geographic targets such as seamounts, which might be used for orientation and navigation (Tyack 1999).

Although no studies have directly measured the sound sensitivity of fin whales, experts assume that fin whales are able to receive sound signals in roughly the same frequencies as the signals they produce. This suggests fin whales, like other baleen whales, are more likely to have their best hearing capacities at low frequencies, including frequencies lower than those of normal human hearing, rather than at mid- to high-frequencies (Ketten 1997a). Several fin whales were tagged during the Southern California-10 behavioral response study (BRS) and no obvious responses to a mid-frequency sound source were detected by the visual observers or in the initial tag analysis (Southall et al. 2011a). However, results of studies on blue whales (Goldbogen et al. 2013; Southall et al. 2011b), which have similar auditory physiology compared to fin whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. In terms of functional hearing capability fin whales belong to the low-frequency group, which have a hearing range of 7 Hz to 22 kHz (Southall et al. 2007b).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c) and for the purposes of this analysis, fin whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.4.9 ***Critical Habitat***

The NMFS has not designated critical habitat for fin whales.

4.2.5 Sei Whale

Sei whales (pronounced "say" or "sigh"; *Balaenoptera borealis*) are members of the baleen whale family and are considered one of the "great whales" or rorquals. Two subspecies of sei whales are recognized, *B. b. borealis* in the Northern Hemisphere and *B. b. schlegellii* in the Southern Hemisphere.

These large animals can reach lengths of about 40 to 60 ft (12 to 18 m) and weigh 100,000 lbs (45,000 kg). Females may be slightly longer than males. Sei whales have a long, sleek body that is dark bluish-gray to black in color and pale underneath. The body is often covered in oval-shaped scars (probably caused from cookie-cutter shark and lamprey bites) and sometimes has subtle mottling. This species has an erect falcate dorsal fin located far down (about two-thirds) the animals back. They often look similar in appearance to Bryde's whales, but can be distinguished by the presence of a single ridge located on the animal's "rostrum". Bryde's whales, unlike other rorquals, have three distinct prominent longitudinal ridges on their rostrum. Sei whales have 219 to 410 baleen plates that are dark in color with gray/white fine inner fringes in their enormous mouths. They also have 30 to 65 relatively short ventral pleats that extend from below the mouth to the naval area. The number of throat grooves and baleen plates may differ depending on geographic population.

The sei whale is regarded as the fastest swimmer among the great whales, reaching bursts of speed in excess of 20 knots. When a sei whale begins a dive it usually submerges by sinking quietly below the surface, often remaining only a few meters deep, leaving a series of swirls or tracks as it move its flukes. When at the water's surface, sei whales can be sighted by a columnar or bushy blow that is about 10 to 13 feet (3 to 4 m) in height. The dorsal fin usually appears at the same time as the blowhole, when the animal surfaces to breathe. This species usually does not arch its back or raise its flukes when diving. Sei whales have an estimated lifespan of 50 to 70 years.

4.2.5.1 ***Distribution***

The sei whale occurs in all oceans of the world except the Arctic. The migratory pattern of this species is thought to encompass long distances from high-latitude feeding areas in summer to low-latitude breeding areas in winter; however, the location of winter areas remains largely unknown (Perry et al. 1999b). Sei whales are often associated with deeper waters and areas along continental shelf edges (Hain et al. 1985). This general offshore pattern is disrupted during occasional incursions into shallower inshore waters (Waring et al. 2004b). The species appears to lack a well-defined social structure and individuals are usually found alone or in small groups of up to six whales (Perry et al. 1999b). When on feeding grounds, larger groupings have been observed (Gambell 1985b).

In the western Atlantic Ocean, sei whales occur from Nova Scotia and Labrador in the summer months and migrate south to Florida, the Gulf of Mexico, and the northern Caribbean (Gambell 1985b). In the eastern Atlantic Ocean, sei whales occur in the Norwegian Sea (as far north as Finnmark in northeastern Norway), occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Gambell 1985b).

In the North Pacific Ocean, sei whales occur from the Bering Sea south to California (on the east) and the coasts of Japan and Korea (on the west). During the winter, sei whales are found from 20° to 23° N (Gambell 1985b; Masaki 1977).

Sei whales occur throughout the Southern Ocean during the summer months, although they do not migrate as far south to feed as blue or fin whales. During the austral winter, sei whales occur off Brazil and the western and eastern coasts of Southern Africa and Australia. During the winter, sei whales are found from 20° to 23° N (Gambell 1985b; Masaki 1977). Sasaki et al. (2013) demonstrated that sei whale in the North Pacific are strongly correlated with sea surface temperatures between 13.1 and 16.8 degrees C.

4.2.5.1.1 Occurrence in the Gulf of Alaska TMAA

Whaling records from the 1900s indicate there were high densities of sei whales in the northwestern and northeastern portions (i.e., near Portlock Bank) of the Gulf of Alaska from May through August (Navy 2006b). There were no sei whales sighted during the April 2009 survey of the action area (Rone et al. 2009b). During a 2012 survey in summer and early fall, Matsuoka et al. (2013) reported 87 sei whale sightings of 1,647 individuals. The majority of these sightings were in the offshore waters in the central to southern Gulf of Alaska and adjacent eastern North Pacific south of the Gulf of Alaska. Most of the sei whales visually detected during this International Whaling Commission-Power Cruise were south of the Navy's action area (Hakamada and Matsuoka 2013). During the 2013 GOALS II survey in the action area, although sei whales were acoustically detected there were no confirmed visual sightings of sei whale (Rone et al. 2014). There were no sei whale detections reported from five bottom-mounted passive acoustic devices deployed within the action area from 2013 to 2014 (Debich et al. 2014; Rice et al. 2015). Sei whale occurrence in the action area during the summer time period is considered rare.

4.2.5.2 ***Population Structure***

The population structure of sei whales is not well defined, but presumed to be discrete by ocean basin (north and south), except for sei whales in the Southern Ocean, which may form a ubiquitous population or several discrete ones.

Some mark-recapture, catch distribution, and morphological research indicate more than one population may exist in the North Pacific Ocean—one between 155° and 175° W, and another east of 155° W (Masaki 1976; Masaki 1977). Sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska and south to California to the east and Japan and Korea to the west (Leatherwood et al. 1982b; Nasu 1974). Sightings have also occurred in Hawaiian waters (Smultea et al. 2010). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998a). Whaling data suggest that sei whales do not venture north of about 55° N (Gregr et al. 2000). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July to September, although other researchers question these observations because no other surveys have reported sei whales in the northern and western Bering Sea. Harwood (1987) evaluated Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea. Harwood (1987) reported that 75 to 85 percent of the North Pacific population resides east of 180°. During winter, sei whales are found from 20° to 23° N (Gambell 1985b; Masaki 1977). Considering the many British Columbia whaling catches in the early to mid-1900s, sei whales have clearly utilized this area in the past (Gregr et al. 2000; Pike and Macaskie 1969).

Sei whales appear to prefer to forage in regions of steep bathymetric relief, such as continental shelf breaks, canyons, or basins situated between banks and ledges (Best and Lockyer 2002; Gregr and Trites 2001; Kenney and Winn 1987), where local hydrographic features appear to help concentrate zooplankton, especially copepods. In their foraging areas, sei whales appear to associate with oceanic frontal systems (Horwood 1987). In the north Pacific, sei whales are found feeding particularly along the cold eastern currents (Perry et al. 1999b).

In the early to mid-1900s, sei whales were hunted off the coast of British Columbia (Gregr et al. 2000; Pike and Macaskie 1969). Masaki (1977) presented sightings data on sei whales in the North Pacific from the mid-1960s to the early 1970s. Over that time interval sei whales did not appear to occur in waters of Washington State and southern British Columbia in May or June, their densities increased in those waters in July and August (1.9 to 2.4 and 0.7 to 0.9 whales per 100 miles of distance for July and August, respectively), then declined again in September. More recently, sei whales have become known for an irruptive migratory habit in which they appear in an area then disappear for time periods that can extend to decades.

4.2.5.3 *Abundance Estimate*

In the North Pacific, the pre-exploitation sei whale population was estimated at 42,000 whales (Tillman 1977a). Results from the 2012 International Whaling Commission-Pacific Ocean Whale and Ecosystem Research (IWC-POWER) cruise suggest an abundance estimate of 27,197 individuals (CV = 0.236) in the central and eastern North Pacific (north of 40° N, south of Alaskan Peninsula, between 170° E and 135° W; Hakamada et al. 2012). However, the best

current estimate of abundance for the eastern North Pacific stock of sei whales that occur off California, Oregon, and Washington waters out to 300 nm is 126 animals (CV = 0.53) and 178 individuals (CV = 0.90) off of Hawaii (Carretta et al. 2014). No data on the current population trend are available; however, the population in the North Pacific is expected to have increased since sei whales began receiving protection in 1976 (Carretta et al. 2013a).

4.2.5.4 *Natural Threats*

Andrews (1916) suggested that killer whales attacked sei whales less frequently than fin and blue whales in the same areas. Sei whales engage in a flight responses to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Endoparasitic helminths (worms) are commonly found in sei whales and can result in pathogenic effects when infestations occur in the liver and kidneys (Rice 1977).

4.2.5.5 *Anthropogenic Threats*

Human activities known to threaten sei whales include whaling, commercial fishing, and maritime vessel traffic. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting may occur in some areas.

Sei whales, because of their offshore distribution and relative scarcity in U.S. Atlantic and Pacific waters, probably have a lower incidence of entrapment and entanglement than fin whales. Data on entanglement and entrapment in non-U.S. waters are not reported systematically. Heyning and Lewis (1990) made a crude estimate of about 73 rorquals killed/year in the southern California offshore drift gillnet fishery during the 1980s. Some of these may have been fin whales instead of sei whales. Some balaenopterids, particularly fin whales, may also be taken in the drift gillnet fisheries for sharks and swordfish along the Pacific coast of Baja California, Mexico (Barlow et al. 1997b). Heyning and Lewis (1990) suggested that most whales killed by offshore fishing gear do not drift far enough to strand on beaches or to be detected floating in the nearshore corridor where most whale-watching and other types of boat traffic occur. Thus, the small amount of documentation may not mean that entanglement in fishing gear is an insignificant cause of mortality. Observer coverage in the Pacific offshore fisheries has been too low for any confident assessment of species-specific entanglement rates (Barlow et al. 1997b). The offshore drift gillnet fishery is the only fishery that is likely to "take" sei whales from this stock, but no fishery mortalities or serious injuries to sei whales have been observed. Sei whales, like other large whales, may break through or carry away fishing gear. Whales carrying gear may die later, become debilitated or seriously injured, or have normal functions impaired, but with no evidence recorded.

Sei whales are occasionally killed in collisions with vessels. Of three sei whales that stranded along the U.S. Atlantic coast between 1975 and 1996, two showed evidence of collisions (Laist et al. 2001). Between 1999 and 2005, there were three reports of sei whales being struck by vessels along the U.S. Atlantic coast and Canada's Maritime Provinces (Cole et al. 2005; Nelson et al. 2007).

Sei whales are known to accumulate DDT, DDE, and PCBs (Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring.

4.2.5.6 *Status and Trends*

The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status has remained since the inception of the ESA in 1973. The 2013-2014 Biennial Report to Congress states the status of the species is considered unknown as of 2012 (NMFS 2014).

Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000 to 38,000 whales by 1967, and reduced again to 20,600 to 23,700 whales by 1973. From 1910 to 1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Harwood and Hembree. 1987; Perry et al. 1999b). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales: 300 to 600 sei whales were killed per year from 1911 to 1955. The sei whale catch peaked in 1959, when 1,340 sei whales were killed. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters. Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968 to 1969, after which the sei whale population declined rapidly (Mizroch et al. 1984). When commercial whaling for sei whales ended in 1974, the population in the North Pacific had been reduced to 7,260 to 12,620 animals (Tillman 1977b). There have been no direct estimates of sei whale populations for the eastern Pacific Ocean (or the entire Pacific). Between 1991 and 2001, during aerial surveys, there were two confirmed sightings of sei whales along the U.S. Pacific coast.

Sei whales are known to occur in the Gulf of Alaska and as far north as the Bering Sea in the north Pacific. However, their distribution is poorly understood. The only stock estimate for U.S. waters is for the eastern north Pacific stock offshore California, Oregon and Washington (Carretta et al. 2009); abundance in Alaskan waters is unknown and they have not been sighted during recent surveys (Rone et al. 2010a; Waite et al. 2003b).

4.2.5.7 ***Diving and Social Behavior***

Generally, sei whales make 5 to 20 shallow dives of 20 to 30 sec duration followed by a deep dive of up to 15 min (Gambell 1985b). The depths of sei whale dives have not been studied; however the composition of their diet suggests that they do not perform dives in excess of 300 meters. Sei whales are usually found in small groups of up to 6 individuals, but they commonly form larger groupings when they are on feeding grounds (Gambell 1985b).

Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2007). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Mizroch et al. 1984; Rice 1977). Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95 percent of their diets (Calkins 1986b). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollack, capelin, and Atka mackerel (Nemoto and Kawamura 1977). In the Southern Ocean, analysis of stomach contents indicates sei whales consume *Calanus spp*. and small-sized euphasiids with prey composition showing latitudinal trends (Kawamura 1974). Evidence indicates that sei whales in the Southern Hemisphere reduce direct interspecific competition with blue and fin whales by consuming a wider variety of prey and by arriving later to feeding grounds (Kirkwood 1992). Rice (1977) suggested that the diverse diet of sei whales may allow them greater opportunity to take advantage of variable prey resources, but may also increase their potential for competition with commercial fisheries.

Little is known about the actual social system of these animals. Groups of 2 to 5 individuals are typically observed, but sometimes thousands may gather if food is abundant. However, these large aggregations may not be dependent on food supply alone, as they often occur during times of migration. Norwegian workers call the times of great sei whale abundance "invasion years." During mating season, males and females may form a social unit, but strong data on this issue are lacking.

4.2.5.8 ***Vocalization and Hearing***

Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100 Hz to 600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200 Hz to 600 Hz range of 1 to 3 s durations (McDonald et al. 2005). Differences may exist in vocalizations between ocean basins (Rankin et al. 2009). Vocalizations from the North Atlantic consisted of paired sequences (0.5 to 0.8 sec, separated by 0.4 to 1.0 sec) of 10 to 20 short (4 msec) FM sweeps between 1.5 to 3.5 kHz (Richardson et al. 1995d).

Recordings made in the presence of sei whales have shown that they produce sounds ranging from short, mid-frequency pulse sequences (Knowlton et al. 1991; Thompson et al. 1979) to low

frequency broadband calls characteristic of mysticetes (Baumgartner et al. 2008; McDonald et al. 2005; Rankin and Barlow 2007). Off the coast of Nova Scotia, Canada, Knowlton et al. (1991) recorded two-phased calls lasting about 0.5 to 0.8 s and ranging in frequency from 1.5 kHz to 3.5 kHz in the presence of sei whales—data similar to that reported by Thompson et al. (1979). These mid-frequency calls are distinctly different from low-frequency tonal and frequency swept calls recorded in later studies. For example, calls recorded in the Antarctic averaged 0.45 ± 0.3 s in duration at 433 ± 192 Hz, with a maximum source level of 156 ± 3.6 dB re 1 μPa-m (McDonald et al. 2005). During winter months off Hawaii, (Rankin and Barlow 2007) recorded down swept calls by sei whales that exhibited two distinct low frequency ranges of 100 Hz to 44 Hz and 39 Hz to 21 Hz, with the former range usually shorter in duration. Similar sei whale calls were also found near the Gulf of Maine in the northwest Atlantic, ranging from 82.3 Hz to 34.0 Hz and averaging 1.38 s in duration (Baumgartner et al. 2008). These calls were primarily single occurrences, but some double or triple calls were noted as well. It is thought that the difference in call frequency may be functional, with the mid-frequency type serving a reproductive purpose and the low frequency calls aiding in feeding/social communication (McDonald et al. 2005). Sei whales have also been shown to reduce their calling rates near the Gulf of Maine at night, presumably when feeding, and increase them during the day, likely for social activity (Baumgartner and Fratantoni 2008). Off the Mariana Islands, 32 sei whale calls were recorded, 25 of which were backed up by sightings (Norris et al. 2012). The peak mean frequency of these calls ranged from 890.6 Hz to 1,046.9 Hz with a mean duration of 3.5 to 0.2 seconds.

While no data on hearing ability for this species are available, Ketten (1997b) hypothesized that mysticetes have acute infrasonic hearing. Results of studies on blue whales (Goldbogen et al. 2013) (Southall et al. 2011a), which have similar auditory physiology compared to sei whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. In terms of functional hearing capability, sei whales belong to low-frequency cetaceans which have a hearing range of 7 Hz to 22 kHz (Southall et al. 2007b). There are no tests or modeling estimates of specific sei whale hearing ranges.

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c) and for the purposes of this analysis, sei whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.5.9 ***Critical Habitat***

The NMFS has not designated critical habitat for sei whales.

4.2.6 Sperm Whale

Sperm whales (*Physeter macrocephalus*) are the largest of the odontocetes (toothed whales) and the most sexually dimorphic cetaceans, with males considerably larger than females. Adult females may grow to lengths of 36 feet (11 m) and weigh 15 tons (13,607 kg). Adult males, however, reach about 52 feet (16 m) and may weigh as much as 45 tons (40,823 kg).

The sperm whale is distinguished by its extremely large head, which takes up to 25 to 35 percent of its total body length. It is the only living cetacean that has a single blowhole asymmetrically situated on the left side of the head near the tip. Sperm whales have the largest brain of any animal (on average 17 pounds (7.8 kg) in mature males), however, compared to their large body size, the brain is not exceptional in size.

There are between 20 to 26 large conical teeth in each side of the lower jaw. The teeth in the upper jaw rarely erupt and are often considered to be vestigial. It appears that teeth may not be necessary for feeding, since they do not break through the gums until puberty, if at all, and healthy sperm whales have been caught that have no teeth.

Sperm whales are mostly dark gray, but oftentimes the interior of the mouth is bright white, and some whales have white patches on the belly. Their flippers are paddle-shaped and small compared to the size of the body, and their flukes are very triangular in shape. They have small dorsal fins that are low, thick, and usually rounded.

4.2.6.1 ***Distribution***

Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Barlow et al. 1997a; Perry et al. 1999b), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Barlow et al. 1997a). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring 1993) where adult males join them to breed.

4.2.6.1.1 Occurrence in the Gulf of Alaska TMAA

Summer surveys in the coastal waters around the central and western Aleutian Islands have found sperm whales to be the most frequently sighted large cetacean (Allen and Angliss 2013). Acoustic surveys have detected the presence of sperm whales year-round in the Gulf of Alaska, although about twice as many are present in summer as in winter (Mellinger et al. 2004a; Moore et al. 2006). Sperm whale echolocation clicks were detected by two HARPs deployed in the shelf and slope region of north-central Gulf of Alaska in July 2011; however, there were much higher detection rates at the deeper site (Baumann-Pickering et al. 2012b). In contrast to the findings of Mellinger et al. (2004a), Baumann-Pickering et al. (2012b) found high numbers of sperm whale

detections in November and December, with a drop off to low numbers of detections throughout January and February. Sperm whale echolocation clicks were prevalent year-round based on reported detections from a slope-deployed bottom-mounted passive acoustic device (Debich et al. 2014; Rice et al. 2015). From analysis of five Navy-funded HARPS in 2014 to 2015, Rice et al. (2015) reported the highest numbers of sperm whale clicks associated with a slope site from June through late November, and a second peak between April and May. Unlike previous reporting, sperm whale detection peaks in 2014 to 2015 occurred from August to September at one seamount in the southern Gulf of Alaska, and another nearby seamount had peak detections from March through April. Three of four other similar devices deployed in the offshore action area reported less frequent echolocation clicks but still with a year-round pattern. During the April 2009 survey of the action area, there were no sperm whale sightings, but they were acoustically detected on 28 different occasions (Rone et al. 2009b). During a 2012 survey in summer and early fall, Matsuoka et al. (2013) reported 50 sightings of 57 individual sperm whales. All sightings were of large male sperm whales and distributed on the shelf and offshore waters of the Gulf of Alaska and adjacent areas of the eastern North Pacific. As noted above, during the 2013 GOALS II survey there were 19 sightings of sperm whales totaling 22 individuals, and sperm whales were acoustically detected from the towed hydrophone array on 241 occasions (Rone et al. 2014). Sperm whale occurrence in the action area during the summer time period is considered likely.

4.2.6.2 ***Population Structure***

There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). The International Whaling Commission currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Barlow et al. 1997a; Dufault et al. 1999). The NMFS recognizes six stocks under the MMPA- three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; Perry et al. 1999c; Waring et al. 2004a). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead 2008).

Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 40º N in winter (Gosho et al. 1984; Miyashita et al. 1995 as cited in Carretta et al. 2005; Rice

1974). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995; Dohl 1983; Forney et al. 1995; Shallenberger 1981).

In the Gulf of Alaska, sperm whales have been sighted along the Aleutian Trench as well as over deeper waters and have been detected acoustically throughout the year (Forney and Brownell Jr. 1996; Mellinger et al. 2004a). Occurrence is higher from July through September than January through March (Mellinger et al. 2004a; Moore et al. 2006). The vast majority of individuals in the region are likely male based upon whaling records and genetic studies; the area is a summer foraging area for these individuals (Allen and Angliss 2010a; Reeves et al. 1985; Straley and O'Connell 2005; Straley et al. 2005). Mean group size has been reported to be 1.2 individuals (Wade et al. 2003; Waite 2003). However, female groups may rarely occur at least up to the central Aleutian Islands (Fearnbach et al. 2012).

4.2.6.3 *Abundance Estimate*

No recent world-wide abundance for sperm whales exists, however a global estimate from 2002 indicates 300,000 to 450,000 individuals (Whitehead 2002a). Currently there is no reliable abundance estimate for the Northeast Pacific stock of sperm whales (Allen and Angliss 2013; NMFS 2015e). The number of sperm whales estimated for the North Pacific is 26,300 to 32,100 individuals, and within the eastern temperate North Pacific (between 20° N and 45° N) was estimated at 26,300 (CV = 0.81) from visual surveys and 32,100 (CV = 0.36) from acoustic detections (Barlow and Taylor 2005).

The California/Oregon/Washington stock is estimated at approximately 1,332 to 2,106 individuals (CV = 0.58) (Moore and Barlow 2014), and the Hawaii stock at 2,539 to 3,354 (CV = 0.34) individuals (Carretta et al. 2014).

4.2.6.4 *Natural Threats*

Sperm whales are known to be occasionally predated upon by killer whales (Jefferson et al. 1991; Pitman et al. 2001) by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989; Weller et al. 1996; Whitehead et al. 1997) and large sharks (Best et al. 1984) and harassed by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989; Weller et al. 1996; Whitehead et al. 1997). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (Goold et al. 2002; Wright 2005), direct widespread causes remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

4.2.6.5 *Anthropogenic Threats*

Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959 to 1983). However, other estimates have included 436,000 individuals killed between 1800 and 1987 (Carretta et al. 2005). However, all of these estimates are likely underestimates due to illegal killings and inaccurate reporting by Soviet whaling fleets between 1947 and 1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (Yablokov 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber 2004a). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006).

In U.S. waters in the Pacific, sperm whales have been incidentally taken only in drift gillnet operations, which killed or seriously injured an average of nine sperm whales per year from 1991 to 1995 (Barlow et al. 1997b).

Interactions between sperm whales and longline fisheries in the Gulf of Alaska have been reported since 1995 and are increasing in frequency (Hill and DeMaster 1998a; Hill et al. 1999; Rice 1989). Between 2002 and 2006, there were three observed serious injuries (considered mortalities) to sperm whales in the Gulf of Alaska from the sablefish longline fishery (Angliss and Outlaw 2008). Sperm whales have also been observed in Gulf of Alaska feeding off longline gear (for sablefish and halibut) at 38 of the surveyed stations (Angliss and Outlaw 2008). Recent findings suggest sperm whales in Alaska may have learned that fishing vessel propeller cavitation (as gear is retrieved) are an indicator that longline gear with fish is present as a predation opportunity (Thode et al. 2007).

Contaminants have been identified in sperm whales, but vary widely in concentration based upon life history and geographic location, with northern hemisphere individuals generally carrying higher burdens (Evans et al. 2004). Contaminants include dieldrin, chlordane, DDT, DDE, PCBs, HCB and HCHs in a variety of body tissues (Aguilar 1983; Evans et al. 2004), as well as several heavy metals (Law et al. 1996). However, unlike other marine mammals, females appear to bioaccumulate toxins at greater levels than males, which may be related to possible dietary differences between females who remain at relatively low latitudes compared to more migratory

males (Aguilar 1983; Wise et al. 2009). Chromium levels from sperm whales skin samples worldwide have varied from undetectable to 122.6 µg Cr/g tissue, with the mean (8.8 µg Cr/g tissue) resembling levels found in human lung tissue with chromium-induced cancer (Wise et al. 2009). Older or larger individuals did not appear to accumulate chromium at higher levels.

4.2.6.6 *Status and Trends*

Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. The 2013-2014 Biennial Report to Congress states the status of the species is considered unknown (NMFS 2014). Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead and Mesnick 2003).

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947 and 1987. Although the IWC protected sperm whales from commercial harvest in 1981, Japanese whalers continued to hunt sperm whales in the North Pacific until 1988 (Barlow et al. 1997a). In 2000, the Japanese Whaling Association announced plans to kill 10 sperm whales in the Pacific Ocean for research. Although consequences of these deaths are unclear, the paucity of population data, uncertainty regarding recovery from whaling, and re-establishment of active programs for whale harvesting pose risks for the recovery and survival of this species. Sperm whales are also hunted for subsistence purposes by whalers from Lamalera, Indonesia, where a traditional whaling industry has been reported to kill up to 56 sperm whales per year.

4.2.6.7 *Diving*

Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km down and durations in excess of 2 hours (Clarke 1976; Watkins 1985; Watkins et al. 1993). However, dives are generally shorter (25 to 45 min) and shallower (400 to 1,000 m). Dives are separated by 8 to 11 min rests at the surface (Gordon 1987; Watwood et al. 2006) (Jochens et al. 2006; Papastavrou et al. 1989). Sperm whales typically travel approximately 3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Unlike other cetaceans, there is a preponderance of dive information for this species, most likely because it is the deepest diver of all cetacean species so generates a lot of interest. Sperm whales

feed on large and medium-sized squid, octopus, rays and sharks, on or near the ocean floor (Clarke 1986; Whitehead 2002b). Some evidence suggests that they do not always dive to the bottom of the sea floor (likely if food is elsewhere in the water column), but that they do generally feed at the bottom of the dive. Davis et al. (2007) report that dive-depths (100 to 500 m) of sperm whales in the Gulf of California overlapped with depth distributions (200 to 400 m) of jumbo squid, based on data from satellite-linked dive recorders placed on both species, particularly during daytime hours. Their research also showed that sperm whales foraged throughout a 24-hour period, and that they rarely dove to the sea floor bottom (>1000 m). The most consistent sperm whale dive type is U-shaped, during which the whale makes a rapid descent to the bottom of the dive, forages at various velocities while at depth (likely while chasing prey) and then ascends rapidly to the surface. There is some evidence that male sperm whales, feeding at higher latitudes during summer months, may forage at several depths including <200 m, and utilize different strategies depending on position in the water column (Teloni et al. 2007).

Sperm whales have a strong preference for waters deeper than 1,000 m (Barlow et al. 1997a; Watkins and Schevill 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956a; Rice 1989). Sperm whales have been observed near Long Island, New York, in water between 40 and 55 m deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956a). Such areas include oceanic islands and along the outer continental shelf.

4.2.6.8 ***Social Behavior***

Movement patterns of Pacific female and immature male groups appear to follow prey distribution and, although not random, movements are difficult to anticipate and are likely associated with feeding success, perception of the environment, and memory of optimal foraging areas (Whitehead 2008). However, no sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred miles are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several days. However, general transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

Gaskin (1973) proposed a northward population shift of sperm whales off New Zealand in the austral autumn based on reduction of available food species and probable temperature tolerances of calves.

Sperm whales are frequently found in locations of high productivity due to upwelling or steep underwater topography, such as continental slopes, seamounts, or canyon features (Jaquet 1996; Jaquet and Whitehead 1996). Cold-core eddy features are also attractive to sperm whales in the Gulf of Mexico, likely because of the large numbers of squid that are drawn to the high concentrations of plankton associated with these features (Biggs et al. 2000; Davis et al. 2000; Davis et al. 2002). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet and Whitehead 1996; Waring et al. 1993). Sperm whales over George's Bank were associated with surface temperatures of 23.2 to 24.9°C (Waring et al. 2004b).

Local information is inconsistent regarding sperm whale tendencies. Gregr and Trites (2001) reported that female sperm whales off British Columbia were relatively unaffected by the surrounding oceanography. However, Tynan et al. (2005) reported increased sperm whales densities with strong turbulence associated topographic features along the continental slope near Heceta Bank. Two noteworthy strandings in the region include an infamous incident (well publicized by the media) of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. In addition, a mass stranding of 47 individuals in Oregon occurred during June 1979 (Norman et al. 2004a; Rice et al. 1986).

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6 to 12 individuals) versus the Pacific (25 to 30 individuals) (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

4.2.6.9 *Vocalization and Hearing*

Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broad-band clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200 to 236 dB re 1μPa), although lower source level energy has been suggested at around 171 dB re 1 μPa (Goold and Jones 1995; Møhl et al. 2003; Weilgart and Whitehead 1993a; Weilgart and Whitehead 1997a). Most of the energy in sperm

whale clicks is concentrated at around 2 kHz to 4 kHz and 10 kHz to 16 kHz (Goold and Jones 1995; NMFS 2006d; Weilgart and Whitehead 1993a). The highly asymmetric head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Cranford 1992; Norris and Harvey 1972; Norris and Harvey. 1972). Long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993a; Weilgart and Whitehead 1997a). However, clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Weilgart and Whitehead 1993a). They may also aid in intra-specific communication. Another class of sound, "squeals", are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

Our understanding of sperm whale hearing stems largely from the sounds they produce. The only direct measurement of hearing was from a young stranded individual from which auditory evoked potentials were recorded (Carder and Ridgway 1990). From this whale, responses support a hearing range of 2.5 kHz to 60 kHz. However, behavioral responses of adult, free-ranging individuals also provide insight into hearing range; sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins et al. 1985a; Watkins and Schevill 1975a). They also stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995). Because they spend large amounts of time at depth and use low-frequency sound, sperm whales are likely to be susceptible to low frequency sound in the ocean (Croll et al. 1999b).

Recordings of sperm whale vocalizations reveal that they produce a variety of sounds, such as clicks, gunshots, chirrups, creaks, short trumpets, pips, squeals and clangs (Goold 1999b). Sperm whales typically produce short-duration repetitive broadband clicks with frequencies below 100 Hz to >30 kHz (Watkins 1977) and dominant frequencies between 1 kHz to 6 kHz and 10 kHz to 16 kHz. The source levels can reach 236 dB re 1 μPa-m (Mohl et al. 2003). The clicks of neonate sperm whales are very different from typical clicks of adults in that they are of low directionality, long duration, and low-frequency (between 300 Hz and 1.7 kHz) with estimated source levels between 140 to 162 dB re 1 μPa-m (Madsen et al. 2003). Clicks are heard most frequently when sperm whales are engaged in diving and foraging behavior (Miller et al. 2004; Whitehead and Weilgart 1991). Creaks (rapid sets of clicks) are heard most frequently when sperm whales are foraging and engaged in the deepest portion of their dives, with inter-click intervals and source levels being altered during these behaviors (Laplanche et al. 2005; Miller et al. 2004).

When sperm whales are socializing, they tend to repeat series of group-distinctive clicks (codas), which follow a precise rhythm and may last for hours (Watkins and Schevill 1977). Codas are shared between individuals in a social unit and are considered to be primarily for intragroup communication (Rendell and Whitehead 2004; Weilgart and Whitehead 1997b). Recent research

in the South Pacific suggests that in breeding areas the majority of codas are produced by mature females (Marcoux et al. 2006). Coda repertoires have also been found to vary geographically and are categorized as dialects, similar to those of killer whales (Pavan et al. 2000; Weilgart and Whitehead 1997b). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean and those in the Pacific (Weilgart and Whitehead 1997b). Three coda types used by male sperm whales have recently been described from data collected over multiple years: these include codas associated with dive cycles, socializing, and alarm (Frantzis and Alexiadou 2008).

Direct measures of sperm whale hearing have been conducted on a stranded neonate using the auditory brainstem response technique: the whale showed responses to pulses ranging from 2.5 kHz to 60 kHz and highest sensitivity to frequencies between 5 kHz to 20 kHz (Ridgway and Carder 2001). Other hearing information consists of indirect data. For example, the anatomy of the sperm whale's inner and middle ear indicates an ability to best hear high-frequency to ultrasonic hearing (Ketten 1992). The sperm whale may also possess better low-frequency hearing than other odontocetes, although not as low as many baleen whales (Ketten 1992). Reactions to anthropogenic sounds can provide indirect evidence of hearing capability, and several studies have made note of changes seen in sperm whale behavior in conjunction with these sounds. For example, sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echo-sounders and submarine sonar (Watkins et al. 1985b; Watkins and Schevill 1975b). In the Caribbean, Watkins et al. (1985b) observed that sperm whales exposed to 3.25 kHz to 8.4 kHz pulses (presumed to be from submarine sonar) interrupted their activities and left the area. Similar reactions were observed from artificial noise generated by banging on a boat hull (Watkins et al. 1985b). André et al. (1997) reported that foraging whales exposed to a 10 kHz pulsed signal did not ultimately exhibit any general avoidance reactions: when resting at the surface in a compact group, sperm whales initially reacted strongly, and then ignored the signal completely (André et al. 1997). Thode et al. (2007) observed that the acoustic signal from the cavitation of a fishing vessel's propeller (110 dB re 1 μPa^2 between 250 Hz and 1.0 kHz) interrupted sperm whale acoustic activity and resulted in the animals converging on the vessel. The full range of functional hearing for the sperm whale is estimated to occur between approximately 150 Hz and 160 kHz, placing them among the group of cetaceans that can hear mid-frequency sounds (Southall et al. 2007c).

Sperm whales have been observed by marine mammal observers aboard Navy surface ships during training events and detected on the PMRF range hydrophones; however, MFAS was not active so no behavioral response data exists during naval training events. However, a sperm whale was tagged for a controlled exposure experiment during BRS-10. The sperm whale did not appear to demonstrate obvious behavioral changes in dive pattern or production of clicks (Miller et al. 2012; Sivle et al. 2012; Southall et al. 2011c).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c) and for the purposes of this analysis, sperm whales were considered part of the mid-frequency cetacean group, with a nominal hearing range between approximately 150 Hz and up to 160 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.6.10 *Critical Habitat*

The NMFS has not designated critical habitat for sperm whales.

4.2.7 Steller Sea Lion – Western DPS

Steller sea lions (*Eumetopias jubatus*) are distributed along the rim of the North Pacific Ocean from San Miguel Island (Channel Islands) off Southern California to northern Hokkaido, Japan (Loughlin et al. 1984; Nowak 2003). Their centers of abundance and distribution are in Gulf of Alaska and the Aleutian Islands (NMFS 1992). In the Bering Sea, the northernmost major rookery is on Walrus Island in the Pribilof Island group. The northernmost major haul-out is on Hall Island off the northwestern tip of St. Matthew Island. Their distribution also extends northward from the western end of the Aleutian chain to sites along the eastern shore of the Kamchatka Peninsula. The NMFS has designated two DPSs of Steller sea lion, the ESA listed endangered western (62 FR 24345) and eastern Steller sea lion DPSs.

4.2.7.1 *Distribution*

The eastern DPS of Steller sea lions includes animals east of Cape Suckling, Alaska (144° W) south to California waters (55 FR 49204). The western DPS of Steller sea lions includes animals west of Cape Suckling, Alaska (144° W; 62 FR 24345). However, individuals move between rookeries and haul out sites regularly, even over long distances between eastern and western DPS locations (Calkins and Pitcher 1982a; Raum-Suryan et al. 2002; Raum-Suryan et al. 2004). Most adult Steller sea lions occupy rookeries during the summer pupping and breeding season and exhibit a high level of site fidelity. During the breeding season, some juveniles and non-breeding adults occur at or near the rookeries, but most are on haulouts (sites that provide regular retreat from the water on exposed rocky shoreline, gravel beaches, and wave-cut platforms or ice; (Ban 2005; Call and Loughlin 2005; Rice 1998a). Adult males may disperse widely after the breeding season. Males that breed in California move north after the breeding season and are rarely seen in California or Oregon except from May through August (Mate 1973). During fall and winter many sea lions disperse from rookeries and increase use of haulouts, particularly on terrestrial sites but also on sea ice in the Bering Sea.

4.2.7.2 *Occurrence in the Gulf of Alaska TMAA*

Given the wide dispersal of individuals, both the western DPS and eastern DPS may occur in the action area (Allen and Angliss 2013). Steller sea lions do not migrate, but they often disperse widely outside of the breeding season. An area of high occurrence extends from the shore to water depths of 273 fathoms (500 m). In the Gulf of Alaska, foraging habitat is primarily shallow, nearshore, and continental shelf waters 4.3 to 13 nm offshore with a secondary occurrence inshore of the 3,280 ft. (1,000 m) isobath, and a rare occurrence seaward of the 3,280 ft. (1,000 m) isobath. Six groups of Steller sea lions, which totaled 28 individuals, were sighted during the April 2009 survey of the action area, in both the inshore and offshore strata (Rone et al. 2009b). No Steller sea lions were identified during the recent (June and July 2013) survey of the action area, although there were six sightings of unidentified pinnipeds (Rone et al. 2014). Steller sea lion occurrence in the action area during the summer time period is considered likely.

4.2.7.3 *Reproduction*

Female Steller sea lions reach sexual maturity and first breed between three and eight years of age and the average age of reproducing females (generation time) is about 10 years (Calkins and Pitcher 1982b; Pitcher and Calkins 1981; York 1994). They give birth to a single pup from May through July and then breed about 11 days after giving birth. Females normally ovulate and breed annually after maturity although there is a high rate of reproductive failures. The gestation period is believed to be about 50 to 51 weeks (Pitcher and Calkins 1981). The available literature indicates an overall reproductive (birth) rate on the order of 55 percent to 70 percent or greater (Gentry 1970; Pike and Maxwell 1958; Pitcher and Calkins 1981). However, natality was reported to be low in the western DPS in recent years (2003-2009; 69%) versus earlier years (43%); (Maniscalco et al. 2010). Survival through the first three weeks can be less than 50 perecnt at some sites, while others can be over 90 percent (Kaplan et al. 2008). Twinning has been reported (Maniscalco and Parker. 2009).

Mothers with newborn pups will make their first foraging trip about a week after giving birth, but trips are short in duration and distance at first, then increase as the pup gets older (Maniscalco et al. 2006; Merrick and Loughlin 1997; Milette 1999; Milette and Trites 2003; Pitcher et al. 2001). Females attending pups tend to stay within 37 km of the rookery (Calkins 1996; Merrick and Loughlin 1997). Newborn pups are wholly dependent upon their mother for milk during at least their first three months of life, and observations suggest they continue to be highly dependent upon their mother through their first winter (Porter 1997; Scheffer 1945; Trites et al. 2006). Generally, female Steller sea lion will nurse their offspring until they are one to two years old (Calkins and Pitcher 1982b; Gentry 1970; Pitcher and Calkins 1981; Sandegren 1970; Trites et al. 2006). Pups may enter the water after 2 to 4 weeks (Sandegren 1970).

Males reach sexual maturity at about the same time as females (three to seven years of age, reported in (Loughlin et al. 1987)), but generally do not reach physical maturity and participate in breeding until about eight to ten years of age (Pitcher and Calkins 1981). The sex ratio of pups at birth is assumed to be about 1:1 or biased toward slightly greater production of males, but non-pups are biased towards females (Calkins and Pitcher 1982b; NMFS 1992; Pike and Maxwell 1958; Trites and Larkin 1992; York 1994).

4.2.7.4 *Habitat*

Steller sea lions are not known to make regular migrations but do move considerable distances. Adult males may disperse hundreds of miles after the breeding season (Calkins 1986a; Calkins and Pitcher 1982b; Loughlin 1997). During fall and winter many sea lions disperse from rookeries and increase use of haulouts, particularly on terrestrial sites but also on sea ice in the Bering Sea. Western Stellers appear to be moving from western Alaska to the central and eastern Gulf of Alaska areas (Fritz et al. 2013).

Adult females may travel far out to sea into water greater than 1,000 m deep (Merrick and Loughlin 1997). Studies on immature Steller sea lions indicate three types of movements: long-range trips (greater than 15 km and greater than 20 hours), short-range trips (less than 15 km and less than 20 hours), and transits to other sites (NMFS 2007). Long-range trips started around 9 months of age and likely occur most frequently around the time of weaning, while short-range trips happen almost daily. Young individuals generally remain within 480 km of rookeries their first year before moving further away in subsequent years (Raum-Suryan et al. 2004). Many animals also use traditional rafting sites, which are places where they rest on the ocean surface in a tightly packed group (Bigg 1985). Frontal features with small-scale temperature gradients appear to be attractive foraging sites for juvenile Steller sea lions (Lander et al. 2010). Large numbers of Steller sea lions are found near the 200 m isobath year round (Consiglieri et al. 1982b). Foraging generally occurs within 8 to 24 km of shore (Fiscus and Braham 1976). However, foraging can occur hundreds of kilometers from shore over extended periods (Merrick et al. 1997).

4.2.7.5 *Feeding*

Steller sea lions are generalist predators that eat various fish (arrowtooth flounder, rockfish, hake, flatfish, Pacific salmon, Pacific herring, Pacific cod, sand lance, skates, cusk eel, lamprey, walleye, Atka mackerel), squids, and octopus and occasionally birds and marine mammals (Brown et al. 2002; Calkins and Goodwin 1988; Daniel and Schneeweis 1992; Jones 1981; McKenzie and Wynne 2008; Olesiuk et al. 1990; Pitcher and Fay 1982; Sinclair and Zeppelin 2002; Womble and Conlon. 2010). Diet is likely strongly influenced by local and temporal changes in prey distribution and abundance (McKenzie and Wynne 2008; Sigler et al. 2009). Haulout selection appears to be driven at least in part by local prey density (Winter et al. 2009).

Adult females embark on foraging trips of at night for 7 to 26 hours during the breeding season, while adult males rarely or never eat while on breeding grounds (Andrews et al. 2001; Loughlin 2002a).

4.2.7.6 *Diving*

Diving activity is highly variable in Steller sea lion by sex and season. During the breeding season, when both males and females occupy rookeries, adult breeding males rarely, if ever, leave the beach (Loughlin 2002b). However, females tend to feed at night on one to two day trips and return to nurse pups (NRC 2003a). Female foraging trips during winter are longer (130 km) and dives are deeper (frequently greater than 250 m). Summer foraging dives, however, are closer to shore (about 16 km) and shallower (100-250 m; (Loughlin 2002b; Merrick and Loughlin 1997)). As pups mature and start foraging for themselves, they develop greater diving ability until roughly 10 years of age (Pitcher et al. 2005). Juveniles usually make shallow dives of 70 to 140 m over 1 to 2 minutes, but much deeper dives in excess of 300 m are known (Loughlin et al. 2003; Merrick and Loughlin 1997; Rehberg et al. 2001). Young animals also tend to stay in shallower water less than 100 m deep and within 20 km from shore (Fadely et al. 2005).

4.2.7.7 *Acoustics and Hearing*

Males and females apparently have different hearing sensitivities, with males hearing best at 1 to 16 kHz (best sensitivity at the low end of the range) and females hearing from 16 to 25 kHz (best hearing at the upper end of the range) (Kastelein et al. 2005a).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Steller sea lions are part of the Otariid group, with hearing limits estimimated to be 100 Hz to 50 kHz in water (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.3.14.

4.2.7.8 *Status and Trends*

Steller sea lions were originally listed as threatened under the ESA on November 26, 1990 (55 FR 49204), following a decline in the U.S. of about 64 percent over the previous three decades. In 1997, the species was split into two separate populations based on demographic and genetic differences (Bickham et al. 1996; Loughlin 1997), and the western population was reclassified to endangered (62 FR 24345) while the eastern population remained threatened (62 FR 30772). On April 18, 2012, the NMFS proposed to delist the eastern DPS of Steller sea lions (77 FR 23209). On November 4, 2013, the NMFS announced that as of December 4, 2013, the eastern DPS of

Steller sea lions would be delisted and no longer protected under the ESA (78 FR 66139). The 2013-2014 Biennial Report to Congress states the status of the Western DPS of this species is considered mixed. The Steller sea lion is listed as near threatened on the 2012 IUCN Red List.

Loughlin et al.(1984) estimated the worldwide population of Steller sea lions was between 245,000 and 290,000 animals (including pups) in the late 1970s. Though the genetic differences between the eastern and western DPSs were not known at the time, Loughlin et al. (1984) noted that 90 percent of the worldwide population of Steller sea lions was in the western DPS in the early 1980s (75 percent in the U.S. and 15 percent in Russia) and 10 percent in the eastern DPS. Loughlin et al. (1984) concluded that the total worldwide population size (both DPSs) was not significantly different from that estimated by Kenyon and Rice (1961) for the years 1959 and 1960, though the distribution of animals had changed. Steller sea lions collected in the Gulf of Alaska during the early 1980s showed evidence of reproductive failure and reduced rates of body growth that were consistent with nutritional limitation (Calkins et al. 1998; Calkins and Goodwin 1988; Pitcher et al. 1998). Between late 1970s and the mid-1990s, counts of the western population of sea lions fell from 109,880 animals to 22,167 animals, a decline of 80 percent (Hauser et al. 2007; NMFS 1995). Although data vary for the major rookeries, as a whole, the western DPS in Alaska has increased in size by an average of 1.45% y^{-1} of pups and 1.67% y^{-1} of non-pups (95 percent credible interval) from 2000 to 2012, and has been increasing annually since 2002 (Allen and Angliss 2014). In 2014, NMFS estimated the western DPS to be comprised of 55,422 individuals in Alaska (Allen and Angliss 2014).

Estimated annual mortality is 0.22 for ages 0 to 2, dropping to 0.07 at age 3, then increasing gradually to 0.15 by age 10 and 0.20 by age 20 (York 1994). Population modeling suggests decreased juvenile survival likely played a major role in the decline of sea lions in the central Gulf of Alaska during 1975-1985 (Holmes and York 2003; Pascual and Adkison 1994; York 1994).

4.2.7.9 *Natural Threats*

Killer whale predation, particularly on the western DPS under reduced population size, may cause significant reductions in the stock (NMFS 2008c). Sleeper sharks are also significant predators of Steller sea lions. Frid et al. (2009) suggested that risk of predation in nearshore waters by killer whales and offshore predation risk by sleeper sharks limited the use of Pacific herring in deep water and walleye Pollock in shallow water.

Steller sea lions have tested positive for several pathogens, but disease levels are unknown (FOC 2008). Similarly, parasites in this species are common, but mortality resulting from infestation is unknown. However, significant negative effects of these factors may occur in combination with stress, which reduces immune capability to resist infections and infestations. If other factors,

such as disturbance, injury, or difficulty feeding occur, it is more likely that disease and parasitism can play a greater role in population reduction.

4.2.7.10 *Anthropogenic Threats*

Steller sea lions were historically and recently subjected to substantial mortality by humans, primarily due to commercial exploitation and both sanctioned and unsanctioned predator control, (Atkinson et al. 2008; Bigg 1988; Bonnot 1928; Bonnot and Ripley 1948; NMFS 2008c; Pearson and Verts 1970; Rowley 1929; Scheffer 1945; Scheffer 1950). Several dozen individuals may become entangled and drown in commercial fishing gear (Atkinson et al. 2008; NMFS 2008c). Several hundred individuals are removed by subsistence hunters annually in controlled and authorized harvests. Occasional harvest occur in Canada (FOC 2008). Additional mortality (362 from 1990 to 2003) has occurred from shooting of sea lions interfering in aquaculture operations along British Columbia (FOC 2008).

Significant concern also exists regarding competition between commercial fisheries and Steller sea lions for the same resource: stocks of pollock, Pacific cod, and Atka mackerel. Significant evidence exists that supports the western DPS declining as a result of change in diet and resulting declines in growth, birth rates, and survival (Atkinson et al. 2008; Calkins et al. 1998; Calkins and Goodwin 1988; Pitcher et al. 1998; Trites and Donnelly 2003). As a result, limitations on fishing grounds, duration of fishing season, and monitoring have been established to prevent Steller sea lion nutritional deficiencies as a result of inadequate prey availability.

Contaminants are a considerable issue for Steller sea lions. Roughly 30 individuals died as a result of the Exxon *Valdez* oil spill and contained particularly high levels of PAH contaminants, presumably as a result of the spill. Blood testing confirmed hydrocarbon exposure. Subsequently, premature birth rates increased and pup survival decreased (Calkins et al. 1994; Loughlin et al. 1996). Organochlorines, including PCBs and DDT (and their metabolites), have been identified in Steller sea lions in greater concentrations than any other pinniped during the 1980s, although levels appear to be declining (Barron et al. 2003; Hoshino et al. 2006). The levels of PCBs have been found to have twice the burden in individuals from Russia than from western Alaska (4.3 ng/g wet weight versus 2.1 ng/g wet weight; (Myers et al. 2008). Levels of DDT in Russian pups were also on average twice that in western Alaska pups (3.3 ng/g wet weight blood versus 1.6 ng/g wet weight). PCB levels in the kidneys of some adult males are high enough that reproductive and immune function may have been compromised (Wang et al. 2011). The source of contamination is likely from pollack, which have been found to contain organochlorines throughout the Gulf of Alaska, but higher in regions occupied by the eastern DPS of Steller sea lions (NMFS 2008c). Heavy metals, including mercury, zinc, copper, metallothionien, and butyltin have been identified in Steller sea lion tissues, but are in concentrations lower than other pinnipeds (Beckmen et al. 2002; Castellini 1999; Kim et al. 1996; NMFS 2008c; Noda et al. 1995). Mercury may be of higher significance, with liver levels being measured at levels above

those necessary to impact fish (Holmes et al. 2008). However, contaminants leading to mortality in Steller sea lions have not been identified (NMFS 2008c). Contaminant burdens are lower in females than males, because contaminants are transferred to the fetus *in utero* as well as through lactation (Lee et al. 1996; Myers et al. 2008). However, this means that new generations tend to start with higher levels of contaminants than their parents originally had. Steller sea lion contaminants are of additional concern because contaminants in the body tend to be mobilized as fat reserves are used, such as when prey availability is low; a situation that is likely occurring for Steller sea lions today.

4.2.8 Coho Salmon

Coho salmon (*Oncorhynchus kisutch*) have dark metallic blue or greenish backs with silver sides and a light belly and there are small black spots on the back and upper lobe of the tail while in the ocean. The gumline in the lower jaw has lighter pigment than does the Chinook salmon. Spawning fish in inland rivers are dark with reddish-maroon coloration on the sides. Adult coho salmon may measure more than 2 feet (61 cm) in length and can weigh up to 36 pounds (16 kg). However, the average weight of adult coho is 8 pounds (3.6 kg).

4.2.8.1 ***Species Distribution***

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle et al. 1986).

4.2.8.1.1 Occurrence in the Gulf of Alaska TMAA action area

In general, only ESA-listed coho salmon ESUs from Oregon and Washington are expected to occur in the Gulf of Alaska TMAA action area. Further discussion of the occurrence of ESA-listed coho salmon in the action area is in section 6.7.2 of this opinion.

4.2.8.2 ***Reproduction***

Coho salmon adults migrate from a marine environment into freshwater streams and rivers of their birth in order to mate (called anadromy). They spawn only once and then die (called semelparity). Adults return to their stream of origin to spawn and die, usually at around three years old. Some precocious males known as "jacks" return as two-year-old spawners. Typically Coho salmon spawn from November to January, although there are many exceptions throughout their range. Spawning duration usually spans about three months in most basins, with individual fish actively spawning for several days to weeks. Spawning occurs in a few third-order streams, but most spawning activity occurs in fourth- and fifth-order streams.

Spawning males develop a strongly hooked snout and large teeth. Females prepare several redds (nests) where the eggs will remain for six to seven weeks until they hatch. As with other Pacific

salmon, Coho salmon fecundity varies with the size of the fish and latitudinally with Coho salmon in northern climes generally exhibiting higher rates of fecundity (Sandercock 1991).

4.2.8.3 ***Habitat***

The typical life history of Coho salmon is similar to most of the other large bodied Pacific salmonids, in so much as adult fish spawn in the fall and winter, young emerge in the spring, rear in freshwater and saltwater and return to spawn as adults. Sympatric in many river basins with Chinook, chum, sockeye, and pink salmon, partitioning occurs through the species' use of different areas of a river for reproduction and rearing, and the length of time they spend in these ecosystems. For instance, Chinook salmon spawn in fast flowing mainstem riverine reaches with large substrate; sockeye salmon spawn in rivers and lakes, and chum salmon spawn in mid- to lower reaches of rivers and have been observed spawning in areas of tidal influence. Coho salmon characteristically spawn in tributaries and slow-flowing shallow creeks in tributaries with gradients of 3 percent or less, which may be fed by cool groundwater sources, and are often widely dispersed within watersheds. Adult Coho salmon may remain in freshwater three or more months before spawning, with early migrants often moving farther upstream (Sandercock 1991).

Most Coho salmon enter rivers between September and February, but entry is influenced by discharge and other factors. In many river systems, Coho salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. First fall freshets combined with high tides trigger the upstream migration of Coho salmon in many basins. Until then, if river flows are low or warm summer temperatures persist, fish may congregate in pools near the mouth of the river or natal stream until conditions change.

Rates of incubation are largely temperature dependent: colder water temperatures will slow development. Generally, in optimal temperatures eggs incubate for about 35 to 50 days, and fry start emerging from the gravel two to three weeks after hatching. Incubation and emergence success are also influenced by dissolved oxygen levels, sediment loading, and scouring high flows. Following emergence, fry aggregate and move to shallow areas near the stream banks. Most Coho salmon rear in freshwater for about 15 to 18 months. As fry grow, they disperse up- and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of 3 percent or less, although they may move to streams with gradients of 4 to 5 percent. Juvenile Coho salmon are often found in small streams less than five feet wide, and may migrate considerable distances to rear in lakes and off-channel ponds. During the summer, fry prefer pools featuring adequate cover such as large woody debris, undercut banks, and overhanging vegetation. Overwintering tends to occur in larger pools, backwater areas, and off stream channels and ponds (e.g., wall-based channels that are groundwater fed).

At not quite 2 years of age, Coho salmon will migrate downstream where they undergo the physiological transition to salt water. The outmigration of smolts begins as early as February and

may continue through the summer and fall, with peak outmigration often between March and June, although this varies among basins and environmental conditions (Sandercock 1991). Several weeks are spent in coastal waters prior to northward migration (PFMC 2000). This is particularly true for Coho originating from Oregonian streams, whose northward movement is generally delayed by strong southerly currents which weaken in the winter months (PFMC 2000). Once in the ocean, Coho salmon generally migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile Coho salmon tend to occur in both coastal and offshore waters. During spring and summer, Coho salmon will forage in waters between 46º N, the Gulf of Alaska, and along Alaska's Aleutian Islands (PFMC 2000). Coho salmon are found in depths ranging from the surface to 250 m, but individuals in the open ocean generally stay within 30 m of the surface (Emmett et al. 1991b). Juveniles occur at even shallower depths (<10 m)(PFMC 2000). Juveniles are also found closer to shore; generally within 74 km (NMFS-NWR 2005; PFMC 2000). However, adults have been tracked well beyond the EEZ off Oregon (PFMC 2000). In years of weak upwelling, individuals tend to concentrate over submarine canyons and areas of more permanent upwelling, while strong upwelling years result in more dispersed stocks. Acceptable temperature regimes run from 4° to 15.2° C, but optimal range is between 8° and 12° C (Emmett et al. 1991b). Further discussion of coho salmon distribution in the Gulf of Alaska is in section 6.7.2.

4.2.8.4 ***Feeding***

Coho salmon are opportunistic feeders. While at sea, Coho salmon tend to eat fish, including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt (Emmett et al. 1991b). While in estuaries and in freshwater Coho salmon are significant predators of Chinook, pink, and chum salmon, as well as aquatic and terrestrial insects. Smaller fish, such as fry, eat chironomids, plecopterans, and other larval insects, and typically use visual cues to find their prey. Juveniles appear to prefer to feed in upwelled oceanic waters, although they are also present in eddy systems (Pool et al. 2008). Oceanic juveniles commonly feed upon euphausiids, chaetognaths, and decapod megalopae in these locations (Brodeur et al. 2010; Pool et al. 2008).

4.2.8.5 ***Hearing***

Although the data available on the hearing sensitivities of Pacific salmon is limited, that information suggests that the species in the family Salmonidae have similar auditory systems and hearing sensitivities (Popper 1977; Popper et al. 2007; Wysocki et al. 2007a). Most of the data available resulted from studies of the hearing capability of Atlantic salmon (*Salmo salar*), which is a "hearing generalist" with a relatively poor sensitivity to sound (Hawkins and Johnstone 1978). Based on the information available, we assume that the coho salmon considered in this consultation have hearing sensitivities ranging from less than 100 Hz to about 580 Hz (Hawkins and Johnstone 1978; Knudsen et al. 1992; Knudsen et al. 1994).

4.2.8.6 *Natural Threats*

Coho salmon, like other salmon, are exposed to high rates of natural predation at each life stage. Winter mortality may be significant for Coho salmon because they overwinter in freshwater, where they can be swept downstream from freshets or eaten by raccoon, cutthroat trout, or other small animals. Once Coho reach the ocean, survival is high (Sandercock 1991). In freshwater, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals; 10 percent of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011a).

4.2.8.7 *Anthropogenic Threats*

Coho salmon have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile Coho salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species range. Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011a).

4.2.8.8 *Coho Salmon ESUs*

Each Coho salmon ESU is treated as a separate species under the ESA (NMFS 2005b). There are currently seven ESUs of coho salmon in Washington, Oregon, and California (NMFS 2005b). Of these ESUs, one is endangered (Central California Coast), and three are threatened (Northern California-Southern Oregon Coasts, Lower Columbia River and Oregon Coast) (NMFS 2005b) (70 FR 37160). As described in sections 6.7.2 and 4.1.3, only individuals from the Lower Columbia River and Oregon Coast ESUs are expected to occur in the Gulf of Alaska TMAA.

4.2.8.8.1 Lower Columbia River Coho Salmon

NMFS listed Lower Columbia River (LCR) coho salmon as threatened in 2005 (70 FR 37160). The LCR coho salmon ESU includes all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Oregon and Washington, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers in Washington; and the Willamette River to Willamette Falls in Oregon. This ESU also includes 21 artificial propagation programs (79 FR 20802). Myers et al. (2006) identified three major population groups (Coastal, Cascade, and Gorge) in the LCR coho salmon ESU, containing a total of 24 independent populations. Two

major native life history types are recognized among LCR coho salmon populations: Type N or late returning, and Type S or early returning. The life history types differ according to run timing, spawn timing, ocean migration patterns and spawning habitat preference (Myers *et al.* 2006).

The vast majority of historical populations in the LCR coho ESU appear to be either extirpated or nearly so, and the two populations with any significant production (Sandy and Clackamas rivers) are at appreciable risk because of low abundance, declining trends, and failure to respond after a dramatic reduction in harvest. Spatial structure is rated "moderate" to "very high" for all populations within this ESU, except the North Fork Lewis River, which has a "low" rating for spatial structure. Out of the 24 populations that make up this ESU, 21 have a "very low" probability of persisting over the next 100 years, and none of them are viable (Beamesderfer 2010; Ford 2011; LCFRB 2010; NMFS 2013c). The large number of hatchery coho salmon in the ESU was also considered an important risk factor. All three status evaluations of LCR coho conducted between the 2005 and 2011 status reviews concluded that the ESU was at very high risk of extinction (Ford 2011).

A few populations with longer term data sets available show stable or slightly positive abundance trends in the last few years (NWFSC 2015). Some trap and haul programs appear to be operating at or near replacement, although other programs still are far from that threshold and require supplementation with additional hatchery-origin spawners. Improvements in the downstream juvenile facilities at Cowlitz Falls, Merwin, and North Fork Dam are likely to further improve the status of the associated upstream populations. While these and other recovery efforts have likely improved the status of a number of LCR coho populations, abundances are still at low levels and the majority of populations remain at moderate or high risk. Land development and increasing human population in this region will likely continue to degrade habitat, especially in lowland areas. Although populations in this ESU have generally improved, especially in the 2013-2014 and 2014-2015 return years, recent poor ocean conditions suggest that population declines might occur in the upcoming return years and the ESU is still at moderate risk of extinction (NWFSC 2015). The 2016 5-year review concluded that the LCR coho salmon ESU should remain listed as threatened (NMFS 2016d).

4.2.8.8.2 Oregon Coast Coho Salmon

NMFS listed the Oregon coast coho salmon as a threatened species in 2008 (73 FR 7816). This ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587), and one hatchery population. The geographic area is physically diverse, and includes numerous rocky headlands and an extensive area with sand dunes. Most rivers within the ESU drain the west slope of the Coast Range, with the exception the Umpqua River, which extends through the Coast Range to drain the Cascade Mountains (Weitkamp et al. 1995). While most coho salmon populations within the

ESU use stream and riverine habitats, there is extensive lake rearing by juvenile coho salmon in several large lake systems.

A thorough status review for Oregon Coast coho was conducted by Stout *et al.* (2011) in response to a delisting petition. The overall assessment of ESU extinction risk indicated considerable uncertainty about its status. The review team assessment was evenly split between moderate risk (47%) and low risk (47%), with a small minority (6%) at high risk. This uncertainty was due largely to the difficulty in balancing clear improvements in some aspects of the ESU's status over the past 15 years (increased abundance, lower harvest rates, reduced hatchery risks), against persistent threats (habitat degradation, climate change) potentially driving the longer term status of the ESU which probably had not changed over the same time frame and were predicted to degrade in the future (NWFSC 2015).

Noted areas of improvement for Oregon Coast coho salmon populations include positive trends in long-term abundance and escapement (NWFSC 2015). Increases in ESU scores for persistence and sustainability also clearly indicate the biological status of the ESU is improving, due in large part to management decisions (reduced harvest and hatchery releases) and favorable environmental variation (i.e., high marine survival) (NWFSC 2015). Significant improvements in hatchery and harvest practices have improved the current status of ESU diversity (Stout et al. 2011). In addition, recent efforts in several coastal estuaries to restore lost wetlands should be beneficial. However, diversity remains lower than it was historically because of the loss of both freshwater and tidal habitat combined with very low returns over the past 20 years(NWFSC 2015).

Despite recent improvements, the ability of the Oregon Coast coho salmon ESU to survive another prolonged period of poor marine survival remains in question (NWFSC 2015). Recent increases in adult escapement do not provide strong enough evidence that the century-long downward trend has changed. It is too early to determine if recent abundance trends can be attributed to stream restoration projects and other recovery actions or high marine survival (Lawson 1993; NWFSC 2015). With marine survival rates expected to decrease for Oregon Coast coho salmon entering the ocean between 2014-2016, it may be advisable to wait to observe how populations fare during this potential downturn before deciding to change the risk status of this ESU (NWFSC 2015). The 2016 5-year review concluded that the Oregon Coast Coho salmon ESU should remain listed as threatened (NMFS 2016f).

4.2.9 Chum Salmon

Second only to Chinook salmon in adult size, chum salmon (*Oncorhynchus keta*) individuals have been reported up to 3.6 feet (1.1 m) and 46 pounds (20.8 kg). However, average weight is around 8 to 15 pounds (3.6 to 6.8 kg). Chum salmon are best known for the enormous canine-like fangs and striking body color of spawning males (a calico pattern, with the front two-thirds

of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line). Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with black speckles. They closely resemble both sockeye and coho salmon at this stage. As chum salmon enter fresh water, their color and appearance changes dramatically. Both sexes develop a "tiger stripe" pattern of bold red and black stripes. Age at maturity appears to follow a latitudinal trend in which a greater number of fish mature at a later age in the northern portion of the species' range.

4.2.9.1 *Distribution*

Chum salmon are more widely distributed than other salmon and may have at one time made up nearly 50 percent of the Pacific salmon biomass in the Pacific Ocean (Salo 1991b). Historically, chum salmon were distributed throughout the coastal regions of western Canada and the United States, as far south as Monterey Bay, California, to the Arctic coast and east to the Mackenzie River, in the Beaufort Sea. They also ranged in Asia from Korea to the Arctic coast of Russia and west to the Lena River. Presently, major spawning populations on the west coast of the United States are found only as far south as Tillamook Bay on the northern Oregon coast.

4.2.9.1.1 Occurrence in the Gulf of Alaska TMAA action area

Both ESA-listed chum salmon ESUs (Hood Canal summer-run and Columbia River) are expected to occur in the Gulf of Alaska TMAA action area. Further discussion of the occurrence of ESA-listed chum salmon in the action area is in section 6.7.2 of this opinion.

4.2.9.2 *Reproduction*

Spawning migrations generally occur in the summer and fall; the precise spawn timing and migration varies across populations. Stream flows and water temperatures can influence stream entry. Sexual differences in the timing of returns to spawning grounds are apparent, with males generally arriving early and females later in the run. Once on the spawning grounds mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior. Generally, spawning runs consist of fish between 2 and 5 years of age, and like Chinook salmon, chum females will build large redds that consist of four or five egg pockets laid in succession. Chum salmon fecundity is highly variable, and is correlated with body size and region (latitudinal trends are evident with northern population having lower absolute and relative fecundities)(Salo 1991b).

Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior. Generally, spawning runs consist of fish between 2 and 5 years of age, and like Chinook salmon, chum females will build large redds that consist of 4 or 5 egg pockets (Salo 1991b). Chum salmon fecundity is highly variable, and is correlated with body size and

region (latitudinal trends are evident with northern population having lower absolute and relative fecundities) (Salo 1991b).

The time necessary for egg incubation until emergence of alevins in freshwater varies among basins and among years within a basin, and is closely correlated to water temperatures such that low temperatures prolong incubation. Egg and alevin survival, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawning time and density, and water temperatures.

Once they emerge from their gravel nests, chum salmon fry outmigrate to seawater almost immediately (Salo 1991b).

4.2.9.3 ***Habitat***

Chum salmon exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations), and like Chinook salmon, chum salmon are semelparous (die after one spawning event). Chum salmon, like pink salmon, usually spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. Juveniles outmigrate to seawater almost immediately after emerging from the gravel that covers their redds (Salo 1991a). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., coastal cutthroat trout, steelhead, coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions.

Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175° E longitude (Johnson et al. 1997b). North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer run chum, may not make extended migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean (Johnson et al. 1997b).

Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

4.2.9.4 *Feeding*

Generally, chum fry emigrate to estuaries between March and May where they forage on epibenthic and neritic food resources. As food resources decline and the fish grow, they move further out to forage on pelagic and nektonic organisms (Salo 1991b; Simenstad and Salo 1982). The timing of juvenile entry into seawater is commonly correlated with nearshore warming and associated plankton blooms (Groot and Margolis 1991). General migratory studies indicate that chum salmon in their first year of life will typically maintain a coastal migratory pattern although the pattern is variable as they mature at sea. At sea, chum salmon feed on pteropods, euphausiids, amphipods, fish, and squid larvae (Salo 1991b). Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids.

4.2.9.5 *Hearing*

Although the data available on the hearing sensitivities of Pacific salmon is limited, that information suggests that the species in the family Salmonidae have similar auditory systems and hearing sensitivities (Popper 1977; Popper et al. 2007; Wysocki et al. 2007a). Most of the data available resulted from studies of the hearing capability of Atlantic salmon (*Salmo salar*), which is a "hearing generalist" with a relatively poor sensitivity to sound (Hawkins and Johnstone 1978). Based on the information available, we assume that the chum salmon considered in this consultation have hearing sensitivities ranging from less than 100 Hz to about 580 Hz (Hawkins and Johnstone 1978; Knudsen et al. 1992; Knudsen et al. 1994; Popper 2008c).

4.2.9.6 *Natural Threats*

Chum salmon are exposed to high rates of natural predation at each life stage, particularly during migration. Mortality at or prior to emergence is significant because eggs develop in the interstitial spaces in the stream gravel; storm surges that redeposit gravel and wash out eggs or introduce silt to the interstitial spaces can reduce egg survival. Other factors that reduce egg survival and larvae development include low dissolved oxygen, poor percolation, and extreme cold or warm temperatures. In freshwater, fry fall prey to older salmon and other trout, as well as birds, sculpin, and various mammals; 10 percent of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011a).

4.2.9.7 *Anthropogenic Threats*

Chum salmon have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of

juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific northwest.

Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011a).

4.2.9.8 *Chum Salmon ESUs*

Each Chum salmon ESU is treated as a separate species under the ESA (NMFS 2005b). There are currently four ESUs of chum, two of which (Columbia River and the Hood Canal Summer-run) have been designated as threatened (70 FR 37161). The Puget Sound/Strait of Georgia and Pacific Coast ESUs have not yet warranted a designation of threatened or endangered (NMFS 2005b). The distribution, as well as the status and trends, of the Chum salmon ESUs considered in this opinion are discussed below.

4.2.9.8.1 Columbia River Chum Salmon

NMFS listed Columbia River chum salmon as threatened on March 25, 1999 (64 FR 14508) and reaffirmed their status on June 28, 2005 (71 FR 37160). The Columbia River chum salmon ESU includes all natural-origin chum salmon in the Columbia River and its tributaries in Washington and Oregon. The species consists of three populations: Grays River, Hardy Creek and Hamilton Creek in Washington State.

The majority of populations in this ESU remain at high to very high risk, with very low abundances (NWFSC 2015). Ford (2011) concluded that 14 out of 17 of chum populations in this ESU were either extirpated or nearly extirpated. The very low persistence probabilities or possible extirpations of most chum salmon populations are due to low abundance, productivity, spatial structure, and diversity. Although, hatchery production of Columbia River chum salmon has been limited and hatchery effects on diversity are thought to have been relatively small, diversity has been greatly reduced at the ESU level because of presumed extirpations and low abundance in the remaining populations (fewer than 100 spawners per year for most populations) (LCFRB 2010; NMFS 2013c). Only one population (Grays River) is at low risk, with spawner abundances in the thousands, and demonstrating a recent positive trend. Two other populations (Washougal River and Lower Gorge) maintain moderate numbers of spawners and appear to be relatively stable (NWFSC 2015).

The life history of chum salmon is such that ocean conditions have a strong influence on the survival of emigrating juveniles. The potential prospect of poor ocean conditions for the near future may put further pressure on Columbia River ESU chum salmon populations. Freshwater

habitat conditions may be negatively influencing spawning and early rearing success in some basins, and contributing to the overall low productivity of the ESU. Land development, especially in the low gradient reaches that chum salmon prefer, will continue to be a threat to most chum populations due to projected increases in the population of the greater Vancouver-Portland area and the Lower Columbia River overall. The viability of this ESU is relatively unchanged since the 2011 review (Ford 2011), and modest improvements in some populations do not warrant a change in risk category, especially given the uncertainty regarding climatic effects and ocean conditions in the near future (NWFSC 2015). The 2016 5-year review concluded that the Columbia River chum salmon ESU should remain listed as threatened (NMFS 2016d).

4.2.9.8.2 Hood Canal Summer-run Chum Salmon

NMFS listed Hood Canal summer-run chum salmon as threatened in 1999 (64 FR 14508), and reaffirmed their status in 2005 (70 FR 37160). The Hood Canal summer-run chum salmon ESU includes summer-run populations in Hood Canal in Puget Sound and in Discovery and Sequim Bays on the Strait of Juan de Fuca. This ESU may also include summer-run fish in the Dungeness River, but the existence of that run is uncertain. Of the sixteen populations of summer chum that are included in this species, seven are considered "functionally extinct" (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek and Chimicum). The remaining nine populations are well distributed throughout the range of the ESU except for the eastern side of Hood Canal (Johnson et al. 1997a). Two independent major population groups have been identified for this ESU: spawning aggregations from rivers and creeks draining into the Strait of Juan de Fuca; and spawning aggregations within Hood Canal proper (Sands 2009). Five hatchery populations are also considered part of this ESU.

The 2011 and 2015 status reviews indicate some positive signs for the Hood Canal summer-run chum ESU. Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; this was considered to be a good sign for viability in terms of spatial structure and diversity (Ford 2011). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NWFSC 2015). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (Ford 2011). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years (NWFSC 2015). Productivity rates, which were quite low at the time of the last review (Ford 2011), increased from 2011-2015 and have been greater than replacement rates in the past two years for both major population groups (NWFSC 2015). However, productivity of individual spawning aggregates still shows only two of eight aggregates have viable performance. Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer

chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time (NWFSC 2015).

4.2.10 Steelhead

Steelhead trout (*Oncorhynchus mykiss*) are usually dark-olive in color, shading to silvery-white on the underside with a heavily speckled body and a pink to red stripe running along their sides. Steelhead trout can reach up to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though average size is much smaller. The life history of this species varies considerably throughout its range. While all *O. mykiss* hatch in gravel-bottomed, fast-flowing, well-oxygenated rivers and streams, some stay in fresh water all their lives. These fish are called rainbow trout. The steelhead that migrate to the ocean develop a much more pointed head, become more silvery in color, and typically grow much larger than the rainbow trout that remain in fresh water. Maximum age is about 11 years. Males mature generally at two years and females at three.

Generally, steelhead occur in two races: the stream-maturing type, summer steelhead, enters freshwater in a sexually immature condition and requires several months in freshwater to mature and spawn; and the ocean-maturing type, winter steelhead, enters freshwater with well-developed gonads and spawns shortly after river entry. Variations in migration timing exist between populations, and some river basins have both summer and winter steelhead, while others only have one race.

Survival at smoltification is higher for larger fish than smaller ones; this is particularly true for individuals that grew larger earlier in life (Beakes et al. 2010).

4.2.10.1 *Species Distribution*

Steelhead are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Good et al. 2005b; Good et al. 2005a; Moyle 1976; NMFS 1997b; Stolz and Schnell 1991).

4.2.10.1.1 Occurrence in the Gulf of Alaska TMAA action area

In general, only ESA-listed steelhead DPSs from Oregon and Washington are expected to occur in the Gulf of Alaska TMAA action area. Further discussion of the occurrence of ESA-listed steelhead in the action area is in section 6.7.2 of this opinion.

4.2.10.2 *Reproduction*

Adults migrate from a marine environment into the freshwater streams and rivers of their birth in order to mate. Unlike other Pacific salmonids, they can spawn more than one time (called iteroparity). Steelhead can be divided into two basic reproductive types, stream-maturing or ocean-maturing, based on the state of sexual maturity at the time of river entry and duration of

spawning migration. The stream-maturing type (summer-run steelhead in the Pacific Northwest and northern California) enters freshwater in a sexually immature condition between May and October and requires several months to mature and spawn. The ocean-maturing type (winter-run steelhead in the Pacific Northwest and northern California) enters freshwater between November and April, with well-developed gonads, and spawns shortly thereafter. Coastal streams are dominated by winter-run steelhead, whereas inland steelhead of the Columbia River basin are almost exclusively summer-run steelhead.

Adult female steelhead will prepare a redd (or nest) in a stream area with suitable gravel type composition, water depth, and velocity. The adult female may deposit eggs in 4 to 5 "nesting pockets" within a single redd. The eggs hatch in 3 to 4 weeks. Steelhead mortality is high early in life and decreases with age. For example, Puget Sound steelhead leaving freshwater and estuarine habitats experience 55 to 86 percent survival to the point of reaching Hood Canal and 0 to 49 percent from Hood Canal to the Strait of Juan de Fuca, with survival increasing greatly upon entering the Pacific Ocean (Moore et al. 2010).

There is a high degree of overlap in spawning timing between populations regardless of run type (Busby et al. 1996b). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996b; Nickelson et al. 1992). Second-time spawners often make up about 70 to 85 percent of repeat spawners, with third time spawners make up 10 to 25 percent of repeats (Stolz and Schnell 1991). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996b).

4.2.10.3 *Habitat*

Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity. Intermittent streams may also be used for spawning (Barnhart 1986b; Everest 1972). Summer steelhead enter freshwater between May and October in the Pacific northwest (Busby et al. 1996b; Nickelson et al. 1992). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). Summer steelhead migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn in January and February (Barnhart 1986a; Meehan and Bjornn 1991; Nickelson et al. 1992). Winter steelhead enter freshwater between November and April in the Pacific northwest (Busby et al. 1996b; Nickelson et al. 1992), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986a). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

As with other salmonids, the larger the fish, the more eggs produced. Egg and hatching success are related to the conditions within the redd and time to hatching is temperature dependent.

Fertilization to hatching is generally less than a month, after which newly hatched fish will remain in the redd for another 2 to 3 weeks. In late spring and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. After emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams. Fry occupy stream margins (Nickelson et al. 1992). Summer rearing takes place primarily in the faster parts of pools, although young-of-the-year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al. 1992).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson et al. 1988). Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in freshwater from 1 to 4 years, then smolt and migrate to the ocean in March and April (Barnhart 1986a). Winter steelhead juveniles generally smolt after 2 years in freshwater (Busby et al. 1996b). Juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal belt as salmon do. Steelhead typically reside in marine waters for 2 or 3 years prior to returning to their natal stream to spawn as 4- or 5-year olds (August 9, 1996, 61 FR 41542); fish in the northern portion of the range may spend more time rearing in marine waters (Stolz and Schnell 1991). Populations in Oregon and California have higher frequencies of age-1-ocean steelhead than populations to the north, but age-2-ocean steelhead generally remain dominant (Busby et al. 1996a).

4.2.10.4 *Feeding*

Juveniles feed primarily on insects (chironomids, baetid mayflies, and hydropsychid caddisflies (Merz 1994). Adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including greenling and other trout; (Chapman and Bjornn 1969; Stolz and Schnell 1991)).

4.2.10.5 *Hearing*

Although the data available on the hearing sensitivities of Pacific salmon is limited, that information suggests that the species in the family Salmonidae have similar auditory systems and hearing sensitivities (Popper 1977; Popper et al. 2007; Wysocki et al. 2007a). Most of the data available resulted from studies of the hearing capability of Atlantic salmon (*Salmo salar*), which is a "hearing generalist" with a relatively poor sensitivity to sound (Hawkins and Johnstone 1978). Based on the information available, we assume that the steelhead considered in this consultation have hearing sensitivities ranging from less than 100 Hz to about 580 Hz (Hawkins and Johnstone 1978; Knudsen et al. 1992; Knudsen et al. 1994).

4.2.10.6 *Natural Threats*

Steelhead are exposed to high rates of natural predation each life stage. The highest mortality occurs between the egg stage and smolt outmigration, and is highest in the first few months following emergence from the redd (Stolz and Schnell 1991). In freshwater, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals; 10 percent of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011a). In the ocean, marine mammals and other fish prey on steelhead, but the extent of such predation is not well known.

4.2.10.7 *Anthropogenic Threats*

Steelhead have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile steelhead; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species range.

Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011a).

4.2.10.8 *Steelhead DPSs*

Each steelhead DPS is treated as a separate species under the ESA (NMFS 2005b). Of the eleven listed steelhead DPSs, one is endangered (Southern California) and ten are threatened (Puget Sound, Central California coast, Snake River basin, Upper Columbia River, Middle Columbia River, Lower Columbia River, Upper Willamette River, Northern California, South-Central California coast, California Central Valley). The distribution, as well as the status and trends, of the steelhead DPSs considered in this opinion are discussed below.

4.2.10.8.1 Lower Columbia River Steelhead

The LCR steelhead DPS is currently listed as threatened under the ESA (originally listed in 1998, reaffirmed in 2006 and 2012). The DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in streams and tributaries to the Columbia River between the Cowlitz and Wind Rivers, Washington, and the Willamette and Hood Rivers, Oregon. The DPS also includes the progeny of seven artificial propagation

programs (79 FR 20802). Four strata and 23 historical populations of LCR steelhead occur within the DPS: 17 winter-run populations and six summer-run populations, within the Cascade and Gorge ecological subregions. Summer steelhead spawning areas in the LCR are found above waterfalls and other features that create seasonal barriers to migration. Where no temporal barriers exist, the winter-run life history dominates. Recent genetic studies indicate that winter run steelhead in the Clackamas River are genetically more similar to native winter run steelhead in the Upper Willamette River than to steelhead in the Lower Columbia River (NWFSC 2015). Further review is necessary before there can be any consideration of redefining the DPS; therefore, the present status evaluation is being conducted based on existing DPS boundaries.

Baseline persistence probabilities (100 year survival) were estimated to be "low" or "very low" for three out of the six summer-run LCR steelhead populations and 13 out of 17 winter-run populations (Beamesderfer 2010; LCFRB 2010; NMFS 2013c). The "low" to "very low" baseline persistence probabilities of most Lower Columbia River steelhead populations reflects low abundance and productivity (NMFS 2013c). All of the populations increased in abundance during the early 2000s but have generally leveled off since 2005 (Ford 2011; NWFSC 2015).

The 2015 status review report concluded that the LCR steelhead DPS continues to be at moderate risk of extinction (NWFSC 2015). The majority of steelhead populations in this DPS (both winter-run and summer-run) continue to persist at low abundances. It is likely that genetic and life history diversity has been reduced as a result of pervasive hatchery effects and population bottlenecks. Hatchery interactions remain a concern in select basins, although the overall situation has improved somewhat in recent years. Passage programs in the Cowlitz and Lewis basins have the potential to provide considerable improvements in abundance and spatial structure, but have not produced self-sustaining populations to date. Habitat degradation continues to be a concern for most populations. Spatial structure remains relatively high for most populations. Even with modest improvements in the status of several winter-run populations, none of the populations appear to be at fully viable status (NWFSC 2015). The 2016 5-year review concluded that the LCR steelhead DPS should remain listed as threatened (NMFS 2016d).

4.2.10.8.2 Middle Columbia River Steelhead

The Middle Columbia River (MCR) steelhead DPS is currently listed as threatened under the ESA (originally listed in 1999, reaffirmed in 2006 and 2012). This DPS includes all naturally spawning populations of steelhead (*Oncorhynchus mykiss*) using tributaries upstream and exclusive of the Wind River (Washington) and the Hood River (Oregon), excluding the Upper Columbia River tributaries (upstream of Priest Rapids Dam) and the Snake River. Also included are steelhead from seven artificial propagation programs (79 FR 20802). Seventeen extant populations have been identified in this DPS plus two extirpated populations (ICTRT 2003; McClure 2005). These populations fall into four major population groups: the Yakima River

Basin (four extant populations), the Umatilla/Walla-Walla drainages (three extant populations); the John Day River drainage (five extant populations) and the Eastern Cascades group (five extant and two extirpated populations that are being reestablished).

Natural origin returns to the majority of populations in two of the four population groups (Yakima River Basin John Day River) in this DPS showed modest increases from 2010 to 2015 (NWFSC 2015). Abundance estimates for 2 of 3 populations with sufficient data in the remaining two population groups (Eastside Cascades and Umatilla/Walla-Walla) were marginally lower. Three of the four population groups in this DPS include at least one population rated at low risk for abundance and productivity. While there have been improvements in the viability ratings for some of the component populations, the MCR steelhead DPS is still not currently meeting the viability criteria described in the recovery plan. Several of the risk factors cited in the 2005 status review remained areas of concern (or key uncertainties) in the 2015 review (NWFSC 2015). The 2016 5-year review concluded that the MCR steelhead DPS remain classified as threatened (NMFS 2016e).

4.2.10.8.3 Upper Columbia River Steelhead

The UCR steelhead DPS is currently listed as threatened under the ESA. This DPS was originally listed in 1997 as endangered, changed to threatened in 2006, reinstated to endangered status per U.S. District Court decision in 2007, and reclassified as threatened per U.S. District Court order in 2009. The UCR steelhead DPS includes all naturally-spawned steelhead populations below natural and manmade impassable barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border, and progeny of six artificial propagation programs. Four independent populations of UCR steelhead have been identified in the same upriver tributaries as for UCR spring-run Chinook salmon (*i.e.*, Wenatchee, Entiat, Methow, and Okanogan).

UCR steelhead populations have increased relative to the low levels observed in the 1990s, but natural origin abundance and productivity remain well below viability thresholds for three out of the four populations (NWFSC 2015). The status of the Wenatchee River steelhead population has continued to improve and the current abundance and productivity viability rating for this population exceeds the minimum threshold for 5% extinction risk. However, the overall DPS remains at high risk (NWFSC 2015). The proportions of hatchery-origin returns in natural spawning areas remain high across the DPS, especially in the Methow and Okanogan river populations. The improvements in natural returns in recent years largely reflect several years of relatively good natural survival in the ocean and tributary habitats. The 2016 5-year review concluded that the UCR steelhead DPS should remain listed as threatened (NMFS 2016g).

4.2.10.8.4 Upper Willamette River Steelhead

The UWR steelhead DPS is currently listed as threatened under the ESA (originally listed in 1999, reaffirmed in 2006 and 2012). This species includes all naturally-spawned steelhead populations below natural and manmade impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to and including the Calapooia River (79 FR 20802). There are four extant populations of UWR steelhead, all within the Western Cascade Range ecological subregion. Historical observations, hatchery records, and genetics suggest that the presence of UWR steelhead in many tributaries on the west side of the upper basin is the result of recent introductions. Nevertheless, although west-side UWR steelhead does not represent a historical population, those tributaries may provide juvenile rearing habitat or may be temporarily (for one or more generations) colonized during periods of high abundance. Recent genetic studies indicate that winter run steelhead in the Clackamas River are genetically more similar to native winter run steelhead in the Upper Willamette River than to steelhead in the Lower Columbia River (NWFSC 2015). Further review is necessary before there can be any consideration of redefining this DPS; therefore, the present status evaluation is being conducted based on existing DPS boundaries.

Populations in this DPS have experienced long-term declines in spawner abundance. Ford (2011) noted UWR steelhead abundance initially increased for a few years starting in 2005 but by 2010 had subsequently declined to levels observed in the mid-1990s. Although the recent magnitude of these declines is relatively moderate, continued declines would be a cause for concern (NWFSC 2015). There is considerable uncertainty in many of the abundance estimates, except for perhaps the tributary dam counts. The elimination of winter-run hatchery release in the basin reduces hatchery threats, but non-native summer steelhead hatchery releases are still a concern for species diversity. Genetic analysis suggests that there is some level introgression among native late-winter steelhead and summer-run steelhead (Van Doornik et al. 2015). Accessibility to historical spawning habitat is still limited, especially in the North Santiam River, and much of the accessible habitat is degraded and under continued development pressure. The 2016 5-year review concluded that the UWR steelhead DPS should remain listed as threatened (NMFS 2016a).

4.2.10.8.5 Snake River Basin Steelhead

The Snake River steelhead DPS is currently listed as threatened under the ESA (originally listed in 1997, reaffirmed in 2006 and 2012). This species includes all naturally-spawned steelhead populations below natural and manmade impassable barriers in streams in the Snake River Basin of southeast Washington, northeast Oregon, and Idaho, and progeny of six artificial propagation programs. Twenty-four historical populations have been identified within five major groups or ecological subregions: Lower Snake River; Grande Ronde River; Clearwater River; Salmon River; and Imnaha River (Ford 2011). Snake River steelhead are classified as summer run based

on their adult run timing patterns. Much of the freshwater habitat used by Snake River steelhead for spawning and rearing is warmer and drier than that associated with other steelhead DPSs.

The 2010-2014 five year geometric mean abundance estimates for the two long term data series of direct population estimates (Joseph Creek and Upper Grande Ronde Mainstem) both increased compared to the 2005-2009 means (NWFSC 2015). These populations have increased an average of 2% per year over the past 15 years and both populations are approaching the peak abundance estimates observed in the mid-1980s. Hatchery origin spawner estimates for both populations continued to be low. Spatial structure ratings for all but one of the Snake Basin steelhead populations were at low or very low risk given the evidence for distribution of natural production within populations. The exception was Panther Creek, which was given a high risk rating for spatial structure based on the lack of spawning in the upper sections.

The 2015 status update concluded that four out of the five ecological subregions within this DPS are not meeting the specific objectives in the draft recovery and the status of many individual populations remains uncertain (NWFSC 2015). A great deal of uncertainty still remains regarding the relative proportion of hatchery fish in natural spawning areas near major hatchery release sites within individual populations. The Grande Ronde subregion is tentatively rated as viable, but more specific data on spawning abundance and the relative contribution of hatchery spawners for the Lower Grande Ronde and Wallowa populations would improve future assessments. Overall, the information analyzed in the 2015 status review did not indicate a change in biological risk status for the Snake River Basin steelhead DPS. The 2016 5-year review concluded that the Snake River Basin steelhead DPS should remain listed as threatened (NMFS 2016c).

4.2.10.8.6 Puget Sound Steelhead

The Puget Sound steelhead DPS was listed as threatened in 2007 (72 FR 26722). This DPS includes all naturally spawned anadromous winter-run and summer-run steelhead in the river basins of Strait of Juan de Fuca, Puget Sound and Hood Canal, Washington. The DPS is bounded to the west by the Elwha River and to the north by the Nooksack River and Dakota Creek. Thirty-two demographically independent populations and 3 major population groups have been identified within the Puget Sound steelhead DPS (Myers 2015). Six artificial propagation programs are also included in the DPS (79 FR 20802).

Following the initial status review of this DPS in 1996, NMFS issued a determination that listing of Puget Sound steelhead was not warranted (61 FR 41451). The 2007 Biological Review Team concluded that this DPS was likely to become at risk of extinction in the foreseeable future due to the following major risk factors: widespread declines in abundance and productivity for most natural populations in the DPS (including those in Skagit and Snohomish rivers, previously considered strongholds for steelhead in Puget Sound); low abundance of all summer-run

populations; and continued releases of out-of-DPS hatchery fish from Skamania River-derived summer-run and highly domesticated Chambers Creek-derived winter-run stocks. Most of the populations in the DPS are small, and recent declines in abundance of natural fish have persisted despite widespread reductions in harvest of natural steelhead in the DPS since the mid-1990s. (Ford 2011; NWFSC 2015). Low population viability is widespread throughout the DPS based on evidence of diminished abundance, productivity, diversity, and spatial structure. The DPS's current status, particularly with respect to abundance and productivity, is considered to be well below the targets needed to achieve delisting and recovery (NWFSC 2015). Particular aspects of diversity and spatial structure, including limited use of suitable habitat, are still likely to be limiting viability of most Puget Sound steelhead populations.

4.3 Summary of NMFS' Final Effects Determinations for ESA-listed Species and Designated Critical Habitat

This section provides a summary of NMFS's final determinations on effects to ESA-listed species, species proposed for listing as threatened or endangered, and designated and proposed critical habitat as assessed in this biological opinion.

Table 13. Summary of Species Effect Determinations for Training Activities in the Action Area

Common Name		NMFS Overall ESA Determination
North Pacific right whale		**LAA**
Humpback whale – Western North Pacific DPS		**LAA**
Humpback whale – Mexico DPS		**LAA**
Blue whale		**LAA**
Fin whale		**LAA**
Sei whale		**LAA**
Western North Pacific gray whale DPS		NLAA
Sperm whale		**LAA**
Steller sea lion – Western DPS		**LAA**
Critical Habitat - Steller sea lion – Western DPS		NLAA
Critical Habitat - North Pacific right whale		NLAA
Leatherback sea turtle		NLAA
Green sea turtle – Central North Pacific and Eastern Pacific DPSs		NLAA
Loggerhead sea turtle – North Pacific Ocean DPS		NLAA
Olive ridley sea turtle		NLAA
Chinook Salmon	Puget Sound ESU	NLAA
	Upper Columbia River Spring-run ESU	NLAA
	Lower Columbia River ESU	NLAA
	Upper Willamette River ESU	NLAA
	Snake River Spring/Summer-run ESU	NLAA
	Snake River Fall-run ESU	NLAA
	California Coastal ESU	NLAA
	Central Valley spring-run ESU	NLAA
	Sacramento River winter-run ESU	NLAA
Coho Salmon	Lower Columbia ESU	**LAA**
	Oregon Coast ESU	**LAA**
	Southern Oregon/Northern California Coast ESU	NLAA
	Central California Coast ESU	NLAA
Chum Salmon	Hood Canal Summer-run ESU	**LAA**
	Columbia River ESU	**LAA**

Common Name		NMFS Overall ESA Determination
Sockeye Salmon	Ozette Lake ESU	NLAA
	Snake River ESU	NLAA
Steelhead Trout	Upper Columbia River DPS	**LAA**
	Middle Columbia River DPS	**LAA**
	Lower Columbia River DPS	**LAA**
	Upper Willamette River DPS	**LAA**
	Snake River Basin DPS	**LAA**
	Puget Sound DPS	**LAA**
	Northern California DPS	NLAA
	California Central Valley DPS	NLAA
	Central California Coast DPS	NLAA
	South-Central California Coast DPS	NLAA
	Southern California DPS	NLAA

5 ENVIRONMENTAL BASELINE

The "Environmental Baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this opinion includes the effects of several activities that affect the survival and recovery of ESA-listed resources in the action area.

The following information summarizes the principal natural and human-caused phenomena in the action area believed to affect the survival and recovery of ESA-listed species in the wild.

5.1 Climate Change

The Fifth Assessment Synthesis Reports from the Working Groups on the Intergovernmental Panel on Climate Change (IPCC) conclude that climate change is unequivocal (IPCC 2013; IPCC 2014). The Report concludes oceans have warmed, with ocean warming the greatest near the surface (e.g., the upper 75 m have warmed by 0.11°C per decade over the period 1971 to

2010) (IPCC 2013; IPCC 2014). Global mean sea level rose by 0.19 m between 1901 and 2010, and the rate of sea level rise since the mid-nineteenth century has been greater than the mean rate during the previous 2 millennia (IPCC 2013). The IPCC projects a rise of the world's oceans from 0.26 to 0.98 meters by the end of the century, depending on the level of greenhouse gas emissions (Doney et al. 2012a). Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (IPCC 2013; IPCC 2014). Further, ocean acidity has increased by 26% since the beginning of the industrial era (IPCC 2013) and this rise has been linked to climate change (Andersson et al. 2015; Foreman and Yamanaka 2011; GAO 2014; Murray et al. 2014; Okey et al. 2014; Secretariat of the Convention on Biological Diversity 2014). Climate change is also expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, heat waves, and droughts (IPCC 2014). Climate change has the potential to impact species abundance, geographic distribution, migration patterns, timing of seasonal activities (IPCC 2014), and species viability into the future. Climate change is also expected to result in the expansion of low oxygen zones in the marine environment (Gilly et al. 2013). Though predicting the precise consequences of climate change on highly mobile marine species, such as many of those considered in this opinion, is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring.

Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012a). Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. He predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. MacLeod (2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans would be affected by climate change, with 47 percent likely to be negatively affected.

Similarly, climate-mediated changes in important prey species populations are likely to affect predator populations. For example, blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990b). Pecl and Jackson (2008) predicted climate change will likely result in squid that hatch out smaller and earlier, undergo faster growth over shorter life-spans, and mature younger at a smaller size. This could have significant negative consequences for species such as sperm whales, whose diets can be dominated by cephalopods. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009).

Previous warming events (e.g., El Niño, the 1977 through 1998 warm phase of the Pacific Decadal Oscillation) may illustrate the potential consequences of climate change. Off the U.S. west coast, past warming events have reduced nutrient input and primary productivity in the California Current, which also reduced productivity of zooplankton through upper-trophic level consumers (Doney et al. 2012a; Sydeman et al. 2009; Veit et al. 1996). In the past, warming events have resulted in reduced food supplies for marine mammals along the U.S. west coast (Feldkamp et al. 1991; Hayward 2000; Le Boeuf and Crocker 2005). Some marine mammal distributions may have shifted northward in response to persistent prey occurrence in more northerly waters during El Niño events (Benson et al. 2002; Danil and Chivers 2005; Lusseau et al. 2004; Norman et al. 2004b; Shane 1994; Shane 1995). Low reproductive success and body condition in humpback whales may have resulted from the 1997/1998 El Niño (Cerchio et al. 2005).

There is now widespread consensus within the scientific community that average atmospheric temperatures on earth are increasing (warming) and that this will continue for at least the next several decades (IPCC 2001; IPCC 2014; Oreskes 2004; Poloczanska et al. 2013). There is also consensus within the scientific community that this warming trend will alter current weather patterns and patterns associated with climatic phenomena, including the timing and intensity of extreme events such as heat-waves, floods, storms, and wet-dry cycles. The threats posed by the direct and indirect effects of global climate change are, or will be, common to many of the species we discuss in this opinion (Doney et al. 2012b; Hazen et al. 2012; Poloczanska et al. 2013).

Climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the reasonably foreseeable future (Houghton 2001; IPCC 2001; IPCC 2002; Parry et al. 2007) (Alter et al. 2010; Cheung et al. 2015; Ramp et al. 2015). The direct effects of climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet, although the magnitude of these changes remain unknown. Species that are shorter-lived, of larger body size, or generalist in nature are likely to be better able to adapt to climate change over the long term versus those that are longer-lived, smaller-sized, or rely upon specialized habitats (Brashares 2003; Cardillo 2003; Cardillo et al. 2005; Issac 2009; Purvis et al. 2000). Climate change is most likely to have its most pronounced effects on species whose populations are already in tenuous positions (Issac 2009). As such, we expect the risk of extinction to listed species to rise with the degree of climate shift associated with global warming.

The IPCC (2014) estimated that by the mid-21st century, the spatial shifts of marine species will cause species richness to increase at mid and high latitudes (high confidence) and to decrease at tropical latitudes (medium confidence), resulting in global redistribution of catch potential for fishes and invertebrates, with implications for food security. Animal displacements are projected to lead to high-latitude invasions and high local-extinction rates in the tropics and semi-enclosed seas. This will cause a 30 to 70 percent increase in the fisheries yield of some high-latitude regions by 2055 (relative to 2005), a redistribution at mid latitudes, but a drop of 40 to 60 percent in the tropics and the Antarctic, based on 2 °C warming above pre-industrial values (medium confidence in the direction of trends in fisheries yields, low confidence in the magnitude of change). If a decrease in global net primary production (NPP) or a shift towards smaller primary producers occurs, the overall fisheries catch potential may also decrease.

The limits to acclimatization or adaptation capacity are presently unknown. However, mass extinctions occurring during much slower rates of climate change in Earth history suggest that evolutionary rates in some organisms may not be fast enough to cope (IPCC 2014).

The IPCC also estimated that average global land and sea surface temperature has increased by 0.6 °C (±0.2) since the mid-1800s, with most of the change occurring since 1976. Eleven of the 12 warmest years on record since 1850 have occurred since 1995 (Poloczanska et al. 2009). Furthermore, the Northern Hemisphere (where a greater proportion of ESA-listed species occur) is warming faster than the Southern Hemisphere, although land temperatures are rising more rapidly than over the oceans (Poloczanska et al. 2009). This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley 2000). The IPCC reviewed computer simulations of the effect of greenhouse gas emissions on observed climate variations that have been recorded in the past and evaluated the influence of natural phenomena such as solar and volcanic activity. Based on their review, the IPCC concluded that natural phenomena are insufficient to explain the increasing trend in land and sea surface temperature, and that most of the warming observed over the last 50 years is likely to be attributable to human activities (IPCC 2001). Climatic models estimate that global temperatures would increase between 1.4 to 5.8 °C from 1990 to 2100 if humans do nothing to reduce greenhouse gas emissions (IPCC 2001). Fiedler et al. (2013) for the 50-year period from 1958 to 2008 concluded that climatic variability has led to documented changes in the pycnocline in the eastern tropical and North Pacific. In particular, "in the eastern equatorial Pacific the pycnocline shoaled by 10 m and weakened by 5 percent over the 50 years, while in the California Current the pycnocline deepened by ~5 m but showed little net change in stratification (which weakened by 5 percent to the mid-1970s, strengthened by 8 percent to the mid-1990s, and then weakened by 4 percent to 2008)." These projections identify a suite of changes in global climate conditions that are relevant to the future status and trend of endangered and threatened species (Table 14).

Table 14. Phenomena associated with projections of global climate change including levels of confidence associated with projections (adapted from (IPCC 2001) and (Patz et al. 2008)).

Phenomenon	Confidence in Observed Changes (observed in the latter 20th Century)	Confidence in Projected Changes (during the 21st Century)
Higher maximum temperatures and a greater number of hot days over almost all land areas	Likely	Very likely
Higher minimum temperatures with fewer cold days and frost days over almost all land areas	Very likely	Very likely
Reduced diurnal temperature range over most land areas	Very likely	Very likely
Increased heat index over most land areas	Likely over many areas	Very likely over most areas
More intense precipitation events	Likely over many mid- to high-latitude areas in Northern Hemisphere	Very likely over many areas
Increased summer continental drying and associated probability of drought	Likely in a few areas	Likely over most mid-latitude continental interiors (projections are inconsistent for other areas)
Increase in peak wind intensities in tropical cyclones	Not observed	Likely over some areas
Increase in mean and peak precipitation intensities in tropical cyclones	Insufficient data	Likely over some areas

The indirect effects of climate change would result from changes in the distribution of temperatures suitable for calving and rearing calves, the distribution and abundance of prey, and the distribution and abundance of competitors or predators. For example, variations in the recruitment of krill (*Euphausia superba*) and the reproductive success of krill predators have been linked to variations in sea-surface temperatures and the extent of sea-ice cover during the winter months. The 2001 IPCC (2001) did not detect significant changes in the extent of Antarctic sea-ice using satellite measurements, Curran (2003) analyzed ice-core samples from 1841 to 1995 and concluded Antarctic sea ice cover had declined by about 20 percent since the 1950s. The Fifth Assessment Report by the IPCC discusses how over the last two decades, the Greenland and Antarctic ice sheets have been losing mass, glaciers have continued to shrink almost worldwide, and Arctic sea ice and Northern Hemisphere spring snow cover have continued to decrease in extent (http://ar5-syr.ipcc.ch/index.php).

The indirect effects of climate change would result from changes in the distribution of temperatures suitable for reproduction, the distribution and abundance of prey and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009). Seagrass habitats have declined by 29 percent in the last 130 years and 19 percent of coral reefs have been lost due to human degradation, reducing

lower latitude habitat for some species (Poloczanska et al. 2009). Primary production is estimated to have declined by 6 percent between the early 1980s and 2010, making foraging more difficult for marine species (Hoegh-Guldberg and Bruno 2010).

The Antarctic Peninsula, which is the northern extension of the Antarctic continent, contains the richest areas of krill in the Southern Ocean. The extent of sea ice cover around this Peninsula has the highest degree of variability relative to other areas within the distribution of krill. Relatively small changes in climate conditions are likely to exert a strong influence on the seasonal pack-ice zone in the Peninsula area, which is likely to affect densities of krill in this region. Because krill are important prey for baleen whales or form a critical component of the food chains on which baleen whales depend, increasing the variability of krill densities or causing those densities to decline dramatically is likely to have adverse effect on populations of baleen whales in the Southern Ocean.

Reid and Croxall (2001) analyzed a 23-year time series of the reproductive performance of predators that depend on krill for prey—Antarctic fur seals (*Arctocephalus gazella*), gentoo penguins (*Pygoscelis papua*), macaroni penguins (*Eudyptes chrysolophus*), and black-browed albatrosses (*Thalassarche melanophrys*)—at South Georgia Island and concluded that these populations experienced increases in the 1980s followed by significant declines in the 1990s accompanied by an increase in the frequency of years with reduced reproductive success. The authors concluded that macaroni penguins and black-browed albatrosses had declined by as much as 50 percent in the 1990s, although incidental mortalities in longline fisheries probably contributed to the decline of the albatross. These authors concluded, however, that these declines result, at least in part, from changes in the structure of the krill population, particularly reduced recruitment into older age classes, which lowers the number of predators this prey species can sustain. The authors concluded that the biomass of krill within the largest size class was sufficient to support predator demand in the 1980s but not in the 1990s.

Similarly, a study of relationships between climate and sea-temperature changes and the arrival of squid off southwestern England over a 20-year period concluded that veined squid (*Loligo forbesi*) migrate eastwards in the English Channel earlier when water in the preceding months is warmer, and that higher temperatures and early arrival correspond with warm phases of the North Atlantic oscillation (Sims et al. 2001). The timing of squid peak abundance advanced by 120 to 150 days in the warmest years compared with the coldest. Seabottom temperatures were closely linked to the extent of squid movement and temperature increases over the five months prior to and during the month of peak squid abundance did not differ between early and late years. These authors concluded that the temporal variation in peak abundance of squid seen off Plymouth represents temperature-dependent movement, which is in turn mediated by climatic changes associated with the North Atlantic Oscillation. Changes in oxygen concentrations and positon within the California Current have the potential to impact the prey of sperm whales.

Hazen et al. (2012) predicted up to 35 percent change in core habitat for some key Pacific species based on climate change scenarios predicated on the rise in average sea surface temperature by 2100. Climate-mediated changes in the distribution and abundance of keystone prey species like krill and climate-mediated changes in the distribution of cephalopod populations worldwide is likely to affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. Blue whales, as predators that specialize in eating krill, seem likely to change their distribution in response to changes in the distribution of krill (for example, see Payne et al. 1990a; Payne 1986); if they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations seem likely to experience declines similar to those observed in other krill predators, which would cause dramatic declines in their population sizes or would increase the year-to-year variation in population size; either of these outcomes would dramatically increase the extinction probabilities of these whales.

Sperm whales, whose diets can be dominated by cephalopods, would have to re-distribute following changes in the distribution and abundance of their prey. This statement assumes that projected changes in global climate would only affect the distribution of cephalopod populations, but would not reduce the number or density of cephalopod populations. If, however, cephalopod populations collapse or decline dramatically, sperm whale populations are likely to collapse or decline dramatically as well.

Periodic weather patterns such as El Niño, La Niña, the Pacific decadal oscillation, and North Pacific Gyre Oscillation can fundamentally change oceanographic conditions in the northeastern Pacific and the biology that is based upon it (Chenillat et al. 2013; Chenillat et al. 2012; Doney et al. 2012b; Kudela et al. 2008; Litzow and Mueter 2013; Mundy and Cooney 2005; Mundy and Olsson 2005; Stabeno et al. 2004; Sydeman et al. 2013). Roughly every 3 to 7 years, El Niño can influence the northeastern Pacific (JOI/USSSP 2003; Stabeno et al. 2004). Typical changes include increased winter air temperature, precipitation, sea level, and down welling favorable conditions (Royer and Weingartner 1999; Whitney et al. 1999). La Niña events tend to swing these conditions in the negative direction (Stabeno et al. 2004). However, sea surface temperatures (SSTs) can take 1 year to change following an El Niño event or change to varying degrees (Bailey et al. 1995; Brodeur et al. 1996a; Freeland 1990; Royer 2005). The 1982/1983 El Niño and other down welling events are generally regarded to have reduced food supplies for marine mammals along the U.S. west coast (Feldkamp et al. 1991; Hayward 2000; Le Boeuf and Crocker 2005). During La Niña conditions in the Gulf of California, Bryde's whales were found to be more abundant, possibly due to increased availability of their prey under La Niña conditions (Salvadeo et al. 2011). Marine mammal distribution and social organization (group size) is also believed to have shifted northward in response to persistent or extralimital prey occurrence in more northerly waters during El Niño events (Benson et al. 2002; Danil and Chivers 2005; Lusseau et al. 2004; Norman et al. 2004b; Shane 1994; Shane 1995). Low

reproductive success and body condition in humpback whales have also been suggested to have resulted from the 1997/1998 El Niño (Cerchio et al. 2005). El Niño events in the winters of 1952 to 1953, 1957 to 1958, 1965 to 1966, and 1982 to 1983 were associated with strong down welling anomalies, which reduces nutrient availability for plankton (Bailey et al. 1995; Thomas and Strub 2001; Wheeler and Hill 1999). Plankton diversity also shifts, as smaller plankton are better able to cope with reduced nutrient availability (Corwith and Wheeler 2002; Sherr et al. 2005).

The Pacific decadal oscillation is the leading mode of variability in the North Pacific and operates over longer periods than either El Niño or La Niña and is capable of altering sea surface temperature, surface winds, and sea level pressure (Mantua 2002; Mantua and Hare 2002; Stabeno et al. 2004). Unlike El Niño and La Niña events, Pacific decadal oscillation events can persist for 20 to 30 years, are more prominent outside the tropics, and mechanisms controlling them are relatively unknown (Hare and Mantua 2000; Mantua and Hare 2002; Minobe 1997; Minobe 1999). During positive Pacific decadal oscillations, the northeastern Pacific experiences above-average sea surface temperatures while the central and western Pacific Ocean undergoes below-normal sea surface temperatures (Mundy and Olsson 2005; Royer 2005). Warm Pacific decadal oscillation regimes, as with El Niño events, tends to decrease productivity along the U.S. west coast (Childers et al. 2005; Hare et al. 1999). However, during the 1977 warm phase of the Pacific decadal oscillation, euphausiid biomass remained the same and copepod abundance actually increased in the Pacific northwest; zooplankton biomass doubled in offshore waters of the Gulf of Alaska (Brodeur et al. 1996b; Brodeur and Ware 1992; Francis and Hare 1997; MacCall et al. 2005; McFarlane and Beamish 1992). Opposite sea surface temperature regimes occur during negative Pacific decadal oscillations (Mundy and Olsson 2005). Positive Pacific decadal oscillations occurred from 1925 to 1946 and 1977 to 1999. Negative Pacific decadal oscillations occurred from 1890 to 1924, 1947 to 1976, and 1999 to present (Childers et al. 2005; Mantua et al. 1997; Minobe 1997).

Recently, additional research has shown that the North Pacific Gyre Oscillation as impacted by the Pacific Decadal Oscillation and El Niño or La Niña events may have a dominant influence on California Current oceanography and associated biological productively (Chenillat et al. 2013; Di Lorenzo et al. 2008; Litzow and Mueter 2013; Patara et al. 2012; Sydeman et al. 2013). While fluctuations in the North Pacific Gyre Oscillation are strongly influenced by the Pacific Decadal Oscillation, the North Pacific Gyre Oscillation in turn has a more dramatic impact and is better correlated with North Pacific variability in salinity, nutrients, chlorophyll, and a variety of zooplankton taxa (Di Lorenzo et al. 2008). Chenillat et al. (2013) found that within the California Current System, changes in the North Pacific Gyre Oscillation impacted timing of spring time favorable winds responsible for the wind driven upwelling and associated nutrient and biological productivity. Sydeman et al. (2013) showed how variation in the North Pacific Gyre Oscillation could account for North Pacific krill productively (primarily *Thysanoessa spinifera*).

Thysanoessa spinifera is a key prey species for blue whales off Central and Southern California (Fiedler et al. 1998; Schoenherr 1991).

Foraging is not the only aspect that climate change could influence. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence. Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott. 2009). It has also been suggested that increases in harmful algal blooms could be a result from increases in sea surface temperature (Simmonds and Eliott. 2009).

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent by increasing sea levels and the intensity, if not the frequency, of hurricanes and tropical storms (Wilkinson and Souter 2008). A half degree Celsius increase in temperatures during hurricane season from 1965 to 2005 correlated with a 40 percent increase in cyclone activity in the Atlantic. Sea levels have risen an average of 1.7 mm/year over the 20th century due to glacial melting and thermal expansion of ocean water; this rate will likely increase. Based on computer models, these phenomena would inundate nesting beaches of sea turtles, change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and would increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson and Souter 2008).

5.2 Whaling

Large whale population numbers in the action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. Prior to current prohibitions on whaling, such as the International Whaling Commission's 1966 moratorium, most large whale species had been depleted to the extent it was necessary to list them as endangered under the Endangered Species Act of 1966. For example, from 1900 to 1965 nearly 30,000 humpback whales were captured and killed in the Pacific Ocean with an unknown number of additional animals captured and killed before 1900 (Perry et al. 1999b). In addition, 9,500 blue whales were reported killed by commercial whalers in the North Pacific between 1910 and 1965 (Ohsumi and Wada. 1972); 61,500 sei whales between 1947 and 1987 (C. Allison, IWC, pers. comm. as cited in Carretta et al. 2014); 46,000 fin whales between 1947 and 1987 (Rice 1984), and 261,148 sperm whales between 1912 and 2006, of which 259,120 individuals were taken between 1946 and 1987 (International Whaling Commission, BIWS catch data, February 2008 version, unpublished; as cited in Allen and Angliss 2014). North Pacific right whales were heavily exploited in the 19th century, affecting an estimated of 26,500 to 37,000 between 1839 and 1909 (Scarff 2001).

These whaling numbers represent minimum catches, as illegal or underreported catches are not included. For example, recently uncovered Union of Soviet Socialist Republics catch records

indicate extensive illegal whaling activity between 1948 and 1979, with a harvest totalling 157,680 sperm whales in the North Pacific Ocean (Ivashchenko et al. 2014). Of these, only 132,505 were reported by the USSR to the Bureau of International Whaling Statistics. Additionally, despite the moratorium on large-scale commercial whaling, catch of some of these species still occurs in the Pacific Ocean whether it be under objection of the IWC, for aboriginal subsistence purposes, or under International Whaling Commission special permit. From 1985 through 2013, 1089 sei whales and 444 sperm whales were harvested. Although these fisheries operate outside of the action area, some of the whales killed in these fisheries are possibly part of the same populations of whales occurring within the action area for this consultation.

Historically, commercial whaling caused all of the large whale species to decline to the point where they faced extinction risks high enough to list them as endangered species. Since the end of large-scale commercial whaling, the primary threat to these species has been eliminated. However, as described in greater detail in the *Status of the Species* section of this opinion, all whale species have not recovered from those historic declines. Scientists cannot determine if those initial declines continue to influence current populations of most large whale species in the North Pacific. For example, the North Pacific right and western North Pacific gray whales have not recovered from the effects of commercial whaling and continue to face very high risks of extinction because of their small population sizes and low population growth rates. In contrast, species such as humpback and blue whale have increased substantially from post-whaling population levels and appear to be recovering despite the impacts of ship strikes, interactions with fishing gear, and increased levels of ambient sound in the Pacific Ocean.

Aboriginal subsistence whaling catch limits by the International Whaling Commission exist in various places around the world. For the year 2014, a total of 12 fin (from Greenland), 9 humpback (from Greenland and the West Indies), 157 minke (from Greenland), 124 gray (from Chukotka), and 53 bowhead (from Alaska) whales were reported for aboriginal catch (IWC 2015). The native people of Chukotka and Washington State are permitted 744 eastern North Pacific gray whales for the years 2013 to 2018 (maximum of 140 in any one year) (IWC 2015). Due to migration patterns and similariy of appearance to eastern North Pacific gray whales, the catch of a western North Pacific gray whale during an aboriginal subsistence whaling event is possible but not likely due to their low abundance. The estimated number of western North Pacific Gray whales to be struck in a single year during proposed aboriginal hunts in Washington State (by the Makah Indian Tribe) is 0.01 to 0.04 individuals (Moore and Weller 2013).

5.3 Fisheries and Bycatch

In addition to being subject to capture in fisheries closer to their natal rivers, ESA-listed salmon are caught in several fisheries that operate in Gulf of Alaska waters. These fisheries include the following: groundfish fisheries managed by NMFS under the Fishery Management Plan for Groundfish of the Gulf of Alaska; salmon fisheries under the Fishery Management Plan for the

Salmon Fisheries in the EEZ off Alaska; Pacific salmon fisheries that operate under the Pacific Salmon Treaty between the U.S. and Canada; 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 Gulf of Alaska TMAA, so are not considered further as part of the Environmental Baseline.

The salmon fisheries in the EEZ off Alaska are managed by NMFS under the Fishery Management Plan (FMP) for the Salmon Fisheries in the EEZ off the Coast of Alaska. This FMP is unique in that it closes a majority of Alaska EEZ waters to commercial salmon fishing, and facilitates State management of the few salmon fisheries in the EEZ. A commercial troll fishery is authorized in the EEZ off Southeast Alaska, but the majority of the remaining EEZ off Central and Western Alaska is closed to commercial salmon fishing (inclusive of the portion of the Gulf of Alaska TMAA inside the EEZ). Groundfish fisheries do occur in the action area and are known to incidentally capture ESA-listed salmonids. Annual prohibited species catch (PSC) limits in groundfish fisheries have been established by the North Pacific Fishery Management Council for Chinook salmon in the central and western Gulf of Alaska. The annual PSC limit for Chinook salmon in the directed central and western Gulf of Alaska pollock trawl fisheries is 18,316 and 6,684 individuals, respectively. Additionally in the central and western Gulf of Alaska non-pollock fisheries, 3,600 Chinook salmon are permitted for the catcher/processor sector, and 3,900 Chinook salmon in the catcher vessel sector (NPFMC 2015). It is important to note that only a small percentage of these fish would be expected to be from ESA-listed populations.

Marine mammals may be impacted by fisheries through entrapment or entanglement in actively fished gear, or may be impacted through entanglement in, or ingestion of, derelict fishing gear. Additionally, some marine mammals considered in this opinion have the potential to be impacted indirectly if a fishery reduces the available prey base for higher trophic level organisms. Due to their highly migratory nature, many species considered in this opinion have the potential to interact with fisheries both in and outside of the action area. Assessing the impact of fisheries on such species is difficult, due to the large number of fisheries that may interact with the animals. For a comprehensive list of U.S. commercial fisheries that may interact with marine mammals in the North Pacific Ocean see NMFS (2013e, Appendix 3).

The vast majority of documented cases of baleen whale entanglements with fishing gear are from actively fished gear (NOAA 2014). Entanglement in fishing gear can result in serious injury and mortality to cetaceans. In 2013, one humpback whale entanglement in the ground tackle of a groundfish (cod jigger) fishery was reported to the NMFS Alaska Region (Helker et al. 2015), and one incidental serious injury and mortality in the Bering Sea/Aleutian Islands flatfish trawl fishery and two in the Bering Sea/Aleutian Islands pollock trawl fishery (Breiwick 2013). From 2009 to 2013, an average of 0.8 humpback whales per year in waters off of Alaska were

seriously injured or killed from the western North Pacific stock and an average of 7.3 individuals from the central North Pacific stock due to entanglements with commercial fishing gear (Muto and Angliss 2015). Mortality and serious injury numbers are minimum estimates as some interactions go unobserved. For example, whales may swim away with portions of the net, not allowing fishery observers or fishers to document the interaction (Carretta et al. 2014). Additionally, since cetaceans occurring in the action area are migratory, these populations are likely to interact with fisheries and derelict gear from outside the action area. For example, many of the humpback whales that occur in the action area migrate to and from Hawaii or the U.S. West coast. In the previous years between 2007 and 2011, 16 documented humpback whale interactions occurred with pot and trap fisheries off the U.S. west coast, and in all instances, the whale either died or was seriously injured. During the same time period and in the same area, gill nets and unidentified fisheries accounted for 10 documented interactions with humpback whales, with one mortality and nine serious injuries (Carretta et al. 2013a). From November 2009 through April 2010, the Hawaii Whale Entanglement Response Network received 32 reports of entangled humpback whales from fishing gear including longline, monofilament (hook-and-line), and local crab pot (trap) gear (Navy 2013). Reports of fin whale entanglement are less common than for humpbacks off the U.S. west coast, with one fin whale death reported off of Alaska in 2012, which was entangled in the ground tackle of a commericial mechanical jig fishing vessel (Helker et al. 2015). In Hawaii, the two longline fisheries that may interact with large marine mammals (the deep-set longline fishery and the shallow-set longline fishery) did not document a fin whale interaction between 2007 and 2011 (Bradford and Forney 2013; McCracken 2013).

5.4 Vessel Strike

In 2012, 28 cruise ships were scheduled to make 450 voyages through Southeast Alaska. Cruise ships comprise 19 percent of the vessel activity in Southeast Alaska and typically only operate in the area about 5 months out of the year. Ferries, passenger vessels with overnight accommodations, and cruise ships comprise 67 percent of the vessel activity, although cruise ships only operate during the 5-month period from May through September. Dry freight cargo barges, tank barges, and freight ships (log and ore carriers) comprise another 30 percent of the vessel activity (Conservation 2012).

The Alaska Marine Highway is a ferry service operated by the State of Alaska, headquartered in Ketchikan, Alaska. The Highway is composed of 3,500 mi. of routes that go as far south as Bellingham, Washington and as far west as Unalaska/Dutch Harbor, Alaska. The highway system operates along the south-central coast of the state, the eastern Aleutian islands, and the inside passage of Alaska and British Columbia. There are 32 terminals located in Washington, British Columbia, and Alaska. Primary concerns for the cumulative impacts analysis include vessels striking marine mammals, introduction of non-native species through hull fouling and ballast water, and underwater sound from ships and other vessels.

Figure 9 depicts the commercial vessel density provided by the automated identification system data for the area from Alaska to the Pacific Northwest in 2011. As evident from the graphic, commercial vessel use is highest in the U.S. EEZ, at straits and passages, and along least-distance line routes between ports. As is evident from the figure, some of those commercial vessel routes pass through the TMAA. Navy vessels used during a Carrier Strike Group exercise are a small, infrequent, and short duration component of overall vessel traffic in the Gulf of Alaska and would only be present during one 21 day (max length) exercise conducted between April and October.

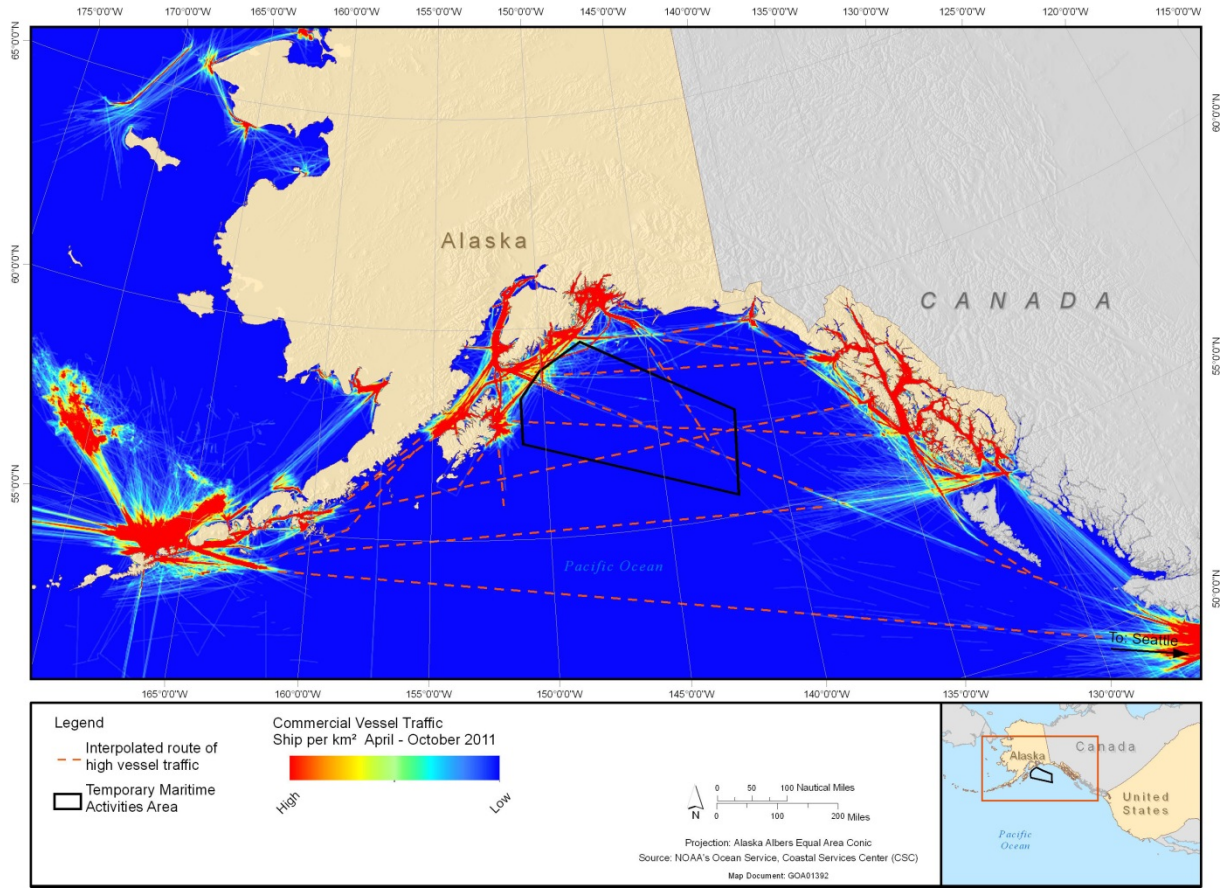

Figure 9. Commercial Ship Automated Identification System Position Data for the Action Area in 2011

Collisions with commercial ships are an increasing threat to many large whale species, particularly as shipping lanes cross important large whale breeding and feeding habitats or migratory routes. From 2009 to 2013, the mean annual mortality rate due to serious injury or mortality from ship strikes reported to the NMFS Alaska Region for the western North Pacific stock of humpback whales was 0.2 individuals and 1.9 individuals for the central North Pacific stock (Muto and Angliss 2015). During this time period, none of these humpback whale vessel strikes occurred in the Gulf of Alaska. According to Helker et al. (2015), only one vessel strike of a marine mammal occurred in the Gulf of Alaska from 2009 to 2013, an unidentified whale

struck in 2012 that was assumed to not be injured by the incident due to the slow speed of the vessel at the time of the collision.

5.5 Ocean Noise

The marine mammals that occur in the action area are regularly exposed to several sources of natural and anthropogenic sounds. Anthropogenic noises that could affect ambient noise arise from the following general types of activities in and near the sea, any combination of which can contribute to the total noise at any one place and time. These noises include transportation, geophysical (seismic) surveys; sonars; explosions; and ocean research activities (Richardson et al. 1995d).

A wide variety of anthropogenic and natural sources contribute to ocean noise throughout the world's oceans (Hatch and Wright 2007b). Anthropogenic sources of noise that are most likely to contribute to increases in ocean noise are vessel noise from commercial shipping and general vessel traffic, oceanographic research, oil and gas exploration, underwater construction, and naval and other use of sound navigation and ranging.

Any potential for cumulative impact should be put into the context of recent changes to ambient sound levels in the world's oceans as a result of anthropogenic activities. However, there is a large and variable natural component to the ambient noise level as a result of events such as earthquakes, rainfall, waves breaking, and lightning hitting the ocean as well as biological noises such as those from snapping shrimp, other crustaceans, fishes, and the vocalizations of marine mammals (Crawford and Huang 1999; Hildebrand 2004; Patek 2002).

Seismic surveys are typically conducted by towing a sound source behind a research vessel, such as an airgun array that emits acoustic energy in timed intervals. The transmitted acoustic energy is reflected and received by an array of hydrophones. This acoustic information is processed to provide information about geological structure below the seafloor. Research geologists have conducted seismic surveys to study plate tectonics as well as other topics in marine geology in the Gulf of Alaska. The oil and gas industry conduct seismic surveys to search for new hydrocarbon deposits, however these are limited to the Cook Inlet and Alaska North Slope regions. The underwater sound produced by seismic surveys could affect marine life, including ESA-listed marine species. All seismic surveys conducted by U.S. vessels are subject to the MMPA authorization process administered by the NMFS, as well as the NEPA process associated with issuing MMPA authorizations.

Noise is of particular concern to marine mammals because many species use sound as a primary sense for navigating, finding prey, avoiding predators, and communicating with other individuals. As described in greater detail later in this opinion, noise may cause marine mammals to leave a habitat, impair their ability to communicate, or to cause stress. Noise can cause

behavioral disturbances, mask other sounds including their own vocalizations, may result in injury and, in some cases, may result in behaviors that ultimately lead to death. The severity of these impacts can vary greatly between minor impacts that have no real cost to the animal, to more severe impacts that may have lasting consequences. A comprehensive discussion of the potential impacts of ocean noise on listed species is included in the *Effects of the Action* section of this opinion.

Noise in the marine environment has received a lot of attention in recent years and is likely to continue to receive attention in the foreseeable future. Several investigators have argued that anthropogenic sources of noise have increased ambient noise levels in the ocean over the last 50 years (Jasny et al. 2005; NRC 1994a; NRC 2000; NRC 2003c; NRC 2005; Richardson et al. 1995d). There can be regional and temporal variations including reductions in anthropogenic noise, especially from commercial shipping volume as it is affected by economic drivers (McKenna et al. 2012a). As discussed in the preceding section, much of this increase is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003c). Commercial fishing vessels, cruise ships, transport boats, airplanes, helicopters and recreational boats all contribute sound into the ocean (NRC 2003c). Anthropogenic noise is also produced during military training and testing activities (e.g., vessels, sonar, explosives). In some areas where oil and gas production takes place, noise originates from the drilling and production platforms, tankers, vessel and aircraft support, seismic surveys, and the explosive removal of platforms (NRC 2003c).

Kipple and Gabriele (2007) measured sounds emitted from 38 vessels ranging in size from 14 to 962 feet at speeds of 10 knots and at a distance of 500 yards from the hydrophone in Glacier Bay, Alaska. Sound levels ranged from a minimum of 157 to a maximum of 182 dB re 1 µPa@1 m, with sound levels showing an increasing trend with both increasing vessel size and with increasing vessel speed. Vessel sound levels also showed dependence on propulsion type and horsepower. Vessel noise can result from several sources including propeller cavitation, vibration of machinery, flow noise, structural radiation, and auxiliary sources such as pumps, fans and other mechanical power sources. McKenna et al. (2012b) measured radiated noise from several types of commercial ships, combining acoustic measurements with ship passage information from AIS. On average, container ships and bulk carriers had the highest estimated broadband source levels (186 dB re 1 lPa^2 20 to 1,000 Hz), despite major differences in size and speed. Differences in the dominant frequency of radiated noise were found to be related to ship type, with bulk carrier noise predominantly near 100 Hz while container ship and tanker noise was predominantly below 40 Hz. The tanker had less acoustic energy in frequencies above 300 Hz, unlike the container and bulk carrier.

Sound emitted from large vessels, such as shipping and cruise ships, is the principal source of low frequency noise in the ocean today, and marine mammals are known to react to or be

affected by that noise (Anderwald et al. 2013; Erbe et al. 2014; Foote et al. 2004; Guerra et al. 2014; Hatch and Wright 2007a; Hildebrand 2005b; Holt et al. 2008a; Kerosky et al. 2013; May-Collado and Quinones-Lebron 2014; Melcon et al. 2012; Richardson et al. 1995c; Williams et al. 2014b). In the Inland Waters of Puget Sound, Erbe et al. (2012) estimated the maximum annual underwater sound exposure level from vessel traffic near Seattle was 215 dB re 1 μPa2-s and Bassett et al. (2010) measured mean sound pressure levels at Admiralty Inlet from commercial shipping at 117 dB re 1 μPa with a maximum exceeded 135 dB re 1 μPa on some occasions. In contrast, Navy combatant vessels have been designed to generate minimal noise and use ship quieting technology to elude detection by enemy passive acoustic devices (Mintz and Filadelfo 2011).

Many researchers have described behavioral responses of marine mammals to the sounds produced by helicopters and fixed-wing aircraft, boats and ships, as well as dredging, construction, geological explorations, etc. (Richardson et al. 1995d). Most observations have been limited to short-term behavioral responses, which included cessation of feeding, resting, or social interactions. Smultea et al. (2008) documented a recognized "stress behavioral reaction" by a group of sperm whales in response to small aircraft fly-bys. The group ceased forward movement, moved closer together in a parallel flank-to-flank formation, and formed a fan-shaped semi-circle with the lone calf remaining near the middle of the group. Several studies have demonstrated short-term effects of disturbance on humpback whale behavior (Baker et al. 1983; Bauer and Herman 1986; Hall 1982; Krieger and Wing 1984), but the long-term effects, if any, are unclear or not detectable. Carretta et al. (2001) and Jasny et al. (2005) identified the increasing levels of anthropogenic noise as a habitat concern for whales and other cetaceans because of its potential effect on their ability to communicate. Significant changes in odontocete behavior attributed to vessel noise have been documented up to at least 5.2 kilometers away from the vessel (Pirotta et al. 2012).

Galli et al. (2003) measured ambient noise levels and source levels of whale-watch boats in Haro Strait. They measured ambient noise levels of 91 dB (at frequencies between 50 and 20,000 Hz) on extremely calm days (corresponding to sea states of zero) and 116 dB on the roughest day on which they took measures (corresponding to a sea state of ~5). Mean sound spectra from acoustic moorings set off Cape Flattery, Washington, showed that close ships dominated the sound field below 10 kHz while rain and drizzle were the dominant sound sources above 20 kHz. At these sites, shipping noise dominated the sound field about 10 to 30 percent of the time but the amount of shipping noise declined as weather conditions deteriorated. The large ships they measured produced source levels that averaged 184 dB at 1 m ± 4 dB, which was similar to the 187 dB at 1 m reported by Greene (1995).

Commercial shipping traffic is a major source of low frequency (5 to 500 Hz) human generated sound in the world's oceans (NRC 2003c; Simmonds and Hutchinson 1996). The radiated noise

spectrum of merchant ships ranges from 20 to 500 Hz and peaks at approximately 60 Hz. Ross (Ross 1976) estimated that between 1950 and 1975 shipping had caused a rise in ambient ocean noise levels of 10 dB; based on his estimates, Ross predicted a continuously increasing trend in ocean ambient noise of 0.55 dB per year. Chapman and Price (2011) recorded low frequency deep ocean ambient noise in the Northeast Pacific Ocean from 1976 to 1986 and reported that the trend of 0.55 dB per year predicted by Ross (1976) persisted until at least around 1980; afterward, the increase per year was significantly less, about 0.2 dB per year.

Williams et al. (2014a) measured ocean noise levels at 12 sites in the Canadian Pacific Ocean, including Haro Strait, and reported that noise levels were high enough to reduce the communication spaces for fin, humpback and killer whales under typical (median) conditions by 1, 52 and 62 percent, respectively, and 30, 94 and 97 percent under noisy conditions.

Bassett et al. (2012) paired one year of AIS data with hydrophone recordings in Puget Sound's Admiralty Inlet to assess ambient noise levels and the contribution of vessel noise to these levels. Results suggested ambient noise levels between 20 Hz and 30 kHz were largely driven by vessel activity and that the increases associated with vessel traffic were biologically significant. Throughout the year, at least one AIS-transmitting vessel was within the study area 90 percent of the time and multiple vessels were present 68 percent of the time. A vessel noise budget showed cargo vessels accounted for 79 percent of acoustic energy, while passenger ferries and tugs had lower source levels but spent substantially more time in the study site and contributed 18 percent of the energy in the budget. All vessels generated acoustic energy at frequencies relevant to all marine mammal functional hearing groups.

Urick (1983) provided a discussion of the ambient noise spectrum expected in the deep ocean. Shipping, seismic activity, and weather are primary causes of deep-water ambient noise. Noise levels between 20 and 500 Hz appear to be dominated by distant shipping noise that usually exceeds wind-related noise. Above 300 Hz, the level of wind-related noise might exceed shipping noise. Wind, wave, and precipitation noise originating close to the point of measurement dominate frequencies from 500 to 50,000 Hz. The ambient noise frequency spectrum and level can be predicted fairly accurately for most deep-water areas based primarily on known shipping traffic density and wind state (wind speed, Beaufort wind force, or sea state) (Urick 1983). For frequencies between 100 and 500 Hz, Urick (1983) has estimated the average deep water ambient noise spectra to be 73 to 80 dB for areas of heavy shipping traffic and high sea states, and 46 to 58 dB for light shipping and calm seas.

In contrast to deep water, ambient noise levels in shallow waters (i.e., coastal areas, bays, harbors, etc.) are subject to wide variations in level and frequency depending on time and location. The primary sources of noise include distant shipping and industrial activities, wind and waves, and marine animals (Urick 1983). At any given time and place, the ambient noise level is a mixture of these noise types. In addition, sound propagation is also affected by the variable

shallow water conditions, including the depth, bottom slope, and type of bottom. Where the bottom is reflective, the sound levels tend to be higher than when the bottom is absorptive.

McDonald et al. (2006a) reported that wind-driven wave noise was an important contributor to ocean ambient noise in the 200 to 500 Hz band. Ross (1976) and Wenz (1962) compared wind data for five northeast Pacific sites and concluded wind was the primary cause for differences in average ambient noise levels above 200 Hz. Assuming the observed increases in ambient noise these authors reported are representative of the larger coast, McDonald et al. (2006a) concluded that the breakpoint between shipping and wind dominated noise has probably now moved well above 200 Hz.

5.6 Commercial and Private Marine Mammal Watching

Vessels (both commercial and private) engaged in marine mammal watching also have the potential to impact whales in the action area. A study of whale watch activities worldwide found that the business of viewing whales and dolphins in their natural habitat has grown rapidly over the past decade into a billion dollar ($US) industry involving over 80 countries and territories and over 9 million participants (Hoyt 2001). In Alaska, the number of whale watchers increased from 76,700 in 1998 to 519,000 in 2008 (O'Connor et al. 2009). In 1988, the Center for Marine Conservation and the NMFS sponsored a workshop to review and evaluate whale watching programs and management needs (CMC and NMFS 1988). That workshop produced several recommendations for addressing potential harassment of marine mammals during wildlife viewing activities that include developing regulations to restrict operating thrill craft near cetaceans, swimming and diving with the animals, and feeding cetaceans in the wild.

Since 1998, NMFS has promulgated regulations at 50 CFR §224.103 that specifically prohibit: (1) the negligent or intentional operation of an aircraft or vessel, or the doing of any other negligent or intentional act which results in disturbing or molesting a marine mammal; (2) feeding or attempting to feed a marine mammal in the wild; and (3) approaching humpback whales in Hawaii and Alaska waters closer than 100 yards (91.4 m). On September 9, 2016, NMFS recodified the approach regulations that appeared in 50 CFR §224.103 for the protection of humpback whales occurring in waters surrounding Alaska. In addition, NMFS launched an education and outreach campaign to provide commercial operators and the general public with responsible marine mammal viewing guidelines which in part state that viewers should: (1) remain at least 50 yards from dolphins, porpoise, seals, sea lions and sea turtles and 100 yards from large whales; (2) limit observation time to 30 minutes; (3) never encircle, chase or entrap animals with boats; (4) place boat engine in neutral if approached by a wild marine mammal; (5) leave the water if approached while swimming; and (6) never feed wild marine mammals.

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, marine mammal watching is not without

potential negative impacts. One concern is that animals may become more vulnerable to vessel strikes once they habituate to vessel traffic. Another concern is that preferred habitats may be abandoned if disturbance levels are too high.

Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Cotton 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Several investigators have studied the effects of whale watch vessels on marine mammals (Amaral and Carlson 2005a; Au and Green 2000a; Christiansen et al. 2013; Christiansen et al. 2011; Corkeron 1995; Erbe 2002b; Felix 2001; Magalhaes et al. 2002; May-Collado and Quinones-Lebron 2014; Richter et al. 2006; Scheidat et al. 2004; Simmonds 2005a; Watkins 1986; Williams et al. 2002a). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. Responses changed with these different variables and, in some circumstances, the whales or dolphins did not respond to the vessels, but in other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

5.7 Large Whale Unusual Mortality Event 2015-2016

Since May 2015, elevated large whale mortalities have occurred in the western portion of the Gulf of Alaska, particularly Kodiak Island (Figure 10). As of September 12, 2016, at least 12 fin, 39 humpback, 2 gray, and 8 unidentified cetaceans have stranded in the area. As of the signing of this opinion, no definitive cause has been determined for this event. Lefebvre et al. (2016) indicate that toxic algal blooms are suspected as a causative agent for Alaska wide marine mammal strandings.

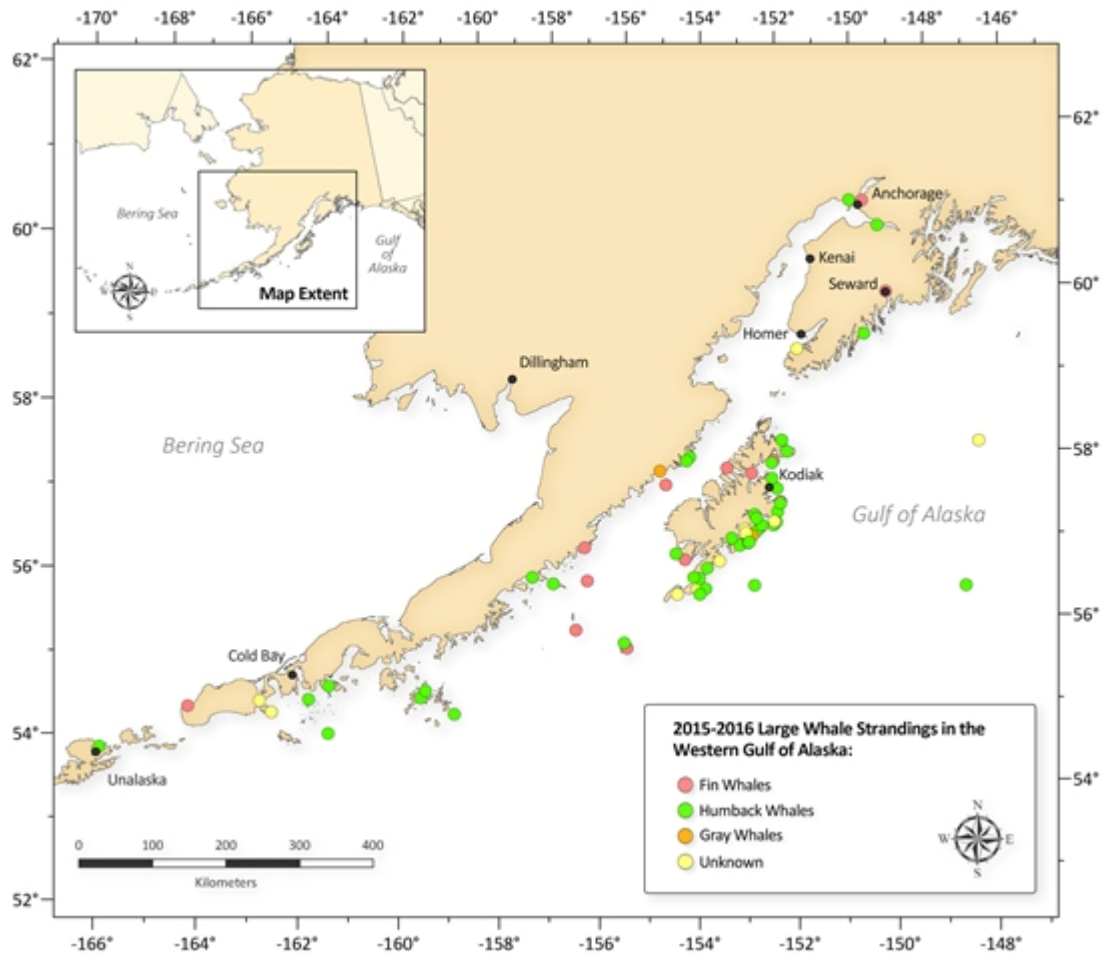

Figure 10. Large whale stranding locations in the Western Gulf of Alaska through September 9, 2016. Map sourced from http://www.nmfs.noaa.gov/pr/health/mmume/large_whales_2015.html (accessed March 10, 2017).

5.8 Marine Debris

Anthropogenic marine debris is prevalent throughout the action area, originating from a variety of oceanic and land-based sources. Debris can be introduced into the marine environment by its improper disposal, accidental loss, or natural disasters (Watters et al. 2010), and can include plastics, glass, derelict fishing gear, derelict vessels, or military expendable materials. Marine debris affects marine habitats and marine life worldwide, primarily by entangling or choking individuals that encounter it. Despite debris removal and outreach to heighten public awareness, marine debris in the environment has not been reduced (Academies 2008).

As noted above in the fisheries interactions section of the Environmental Baseline, entanglement or entrapment in derelict fishing gear can pose a threat to many of the species considered in this opinion. The vast majority of reported cases of entangled baleen whales in the U.S. are humpbacks, with most of these interactions likely involving actively fished, rather than derelict,

gear (Program 2014). In Alaska, only 24 percent of documented entanglements were from unknown sources, possibly including marine debris, with the rest of the cases being from a known fishery and likely being actively fished (Jensen et al. 2009). As noted previously, it is likely that some animals interact with fishing gear outside of the action area, become entangled, and bring that gear with them when they migrate to the action area. For example, 10 humpbacks with entangled gear observed in Hawaii have also been sighted with entangled gear in Alaska, with one animal traveling over 2,450 nautical miles with gear attached (Lyman 2012).

Anthropogenic marine debris can also be accidentally consumed while foraging. Recently weaned juveniles, who are investigating multiple types of prey items, may be particularly vulnerable to ingesting non-food items (Baird and Hooker 2000) (Schuyler et al. 2013). This can have significant implications for an animal's survival, potentially leading to starvation or malnutrition, or internal injuries from consumption. In 2008, two sperm whales stranded along the California coast, with an assortment of fishing related debris (e.g., net scraps, rope) and other plastics inside their stomachs (Jacobsen et al. 2010). One whale was emaciated, and the other had a ruptured stomach. It was suspected that gastric impaction was the cause of both deaths. Jacobsen et al. (2010) speculated that the debris likely accumulated over many years, possibly in the North Pacific gyre that would carry derelict Asian fishing gear into eastern Pacific waters.

5.9 Navy Activities in the Action Area

Since the 1990s, the Navy has participated in major training exercises in the Gulf of Alaska. As described in Section 5.9.2, the Navy also invests in marine mammal research and monitoring in the Gulf of Alaska.

5.9.1 U.S. Navy Training

U.S. Navy training and testing activities are ongoing in nature, though in the Gulf of Alaska, Navy participation in training activities typically occurs on a biennial basis (i.e., every other year). One or many Navy programs may be operating in and around the action area during a 21 day period between April and October within a year. The discussion here will focus on the most recent Navy training activities conducted in the Gulf of Alaska (i.e., activities conducted from 2013 through 2016), the potential effects of which were analyzed in a biological opinion issued on May 14, 2013 (NMFS 2013a).

There were no Navy at-sea training activities or vessel use within the Gulf of Alaska TMAA from 2013 to 2014, or in 2016. In 2015, approximately 6,000 U.S. military personnel participated in exercise Northern Edge, a joint training exercise which has both land and sea-based components hosted by Alaskan Command. The exercise took place in June of 2015 on and above central Alaska ranges and the Gulf of Alaska. Northern Edge 2015 was one in a series of U.S. Pacific Command exercises in 2015 that prepared joint forces to respond to crises in the Indo-

Asia-Pacific region. The exercise was designed to sharpen tactical combat skills, improve command, control and communication relationships, and to develop interoperable plans and programs across the joint force. Personnel from U.S. military units stationed in the continental United States and from U.S. installations in the Pacific participated with approximately 200 aircraft from all services, as well as three U.S. Navy destroyers and one U.S. Navy submarine operating in the Gulf of Alaska. For the 2015 exercise, most personnel and units deployed to and operate from Joint Base Elmendorf-Richardson and Eielson Air Force Base. Participants served as part of a joint task force practicing tasks associated with joint operations. The Navy's training activities are conducted with an extensive set of range clearance and mitigation measures designed to minimize the potential risk to marine life. U.S. Navy vessels also conduct range clearance and mitigation measures designed to avoid interaction and potential for any damage to participating and non-participating vessels and aircraft.

During Northern Edge 2015, there were 33 reported sightings of 70 marine mammals and 4 mitigation events (i.e., two instances where sonar was shut down and two instances where sonar was off, but the vessel maneuvered). This information is summarized in Table 15.

Table 15. Marine mammal sighting information and mitigation events during Northern Edge 2015.

Marine animal type	Number of sightings	Number of individuals	Sonar source in use?	Mitigation implemented?
Whale	24	45	N	N
Whale	2	9	Y	Y
Whale	2	2	N	Y
Sea otter	1	1	N	N
Pinniped	2	2	N	N
Dolphins	1	8	N	N
Unknown marine mammal	1	3	N	N

5.9.2 Ongoing Monitoring

The effort described below from the Navy's 2014 and 2015 annual GOA TMAA monitoring reports represents a 2013-2016 investment by the Navy of ~$2M in marine mammal science for the Gulf of Alaska.

Project Description	Intermediate Scientific Objective*	Status
Title: Gulf of Alaska Line-Transect Survey (GOALS) II: Marine Mammal Occurrence in the Temporary Maritime Activities Area Location: Gulf of Alaska Temporary Maritime Activities Area Objectives: Fill knowledge gaps on the distribution, movements, and densities of marine mammals Methods: Vessel-based line-transect surveys, passive acoustic monitoring, photo-identification, tagging Performing Organizations: National Marine Fisheries Service Alaska Fisheries Science Center; Cascadia Research Collective; Bio-Waves, Inc., HDR, Inc. Timeline: 2013 Funding: $1.1M	Determine what species and populations of marine mammals and ESA-listed species are present in Navy range complexes and testing ranges Determine what species and populations of marine mammals and ESA-listed species are exposed to Navy training and testing activities Estimate the distribution, abundance, and density of marine mammals and sea turtles in Navy range complexes, testing ranges, and in specific training and testing areas	Complete.
Title: Passive Acoustic Monitoring of Marine Mammals in the Gulf of Alaska Temporary Maritime Activities Area using Autonomous Gliders Location: Gulf of Alaska Objectives: Determine spatial distribution and occurrence of beaked whales, other odontocetes, and baleen whales in offshore areas using deep-diving autonomous gliders Methods: Passive acoustic monitoring Performing Organizations: HDR, Inc., Oregon State University, University of Washington Timeline: 2015 Funding: $198k	Determine what species and populations of marine mammals and ESA-listed species are present in Navy range complexes and testing ranges Development and validation of techniques and tools for detecting, classifying, and tracking marine mammals Establish the regional baseline vocalization behavior, including seasonality and acoustic characteristics) of marine mammals where Navy training and testing activities occur	Field work June-July 2015 Final analysis and reporting Spring 2016

Project Description	Intermediate Scientific Objective*	Status
Title: Passive Acoustic Monitoring of Marine Mammals in the Gulf of Alaska Temporary Maritime Activities Area using Bottom-Mounted Passive Acoustic Devices Location: Gulf of Alaska Temporary Maritime Activities Area Objectives: Determine spatial distribution and occurrence of beaked whales, other odontocetes, and baleen whales in offshore areas using deep-diving autonomous gliders Methods: Passive acoustic monitoring Performing Organizations: : Scripps Institution of Oceanography, University of California San Diego Timeline: 2011-2015 Funding: $200k	Determine what species and populations of marine mammals and ESA-listed species are present in Navy range complexes and testing ranges Development and validation of techniques and tools for detecting, classifying, and tracking marine mammals Establish the regional baseline vocalization behavior, including seasonality and acoustic characteristics) of marine mammals where Navy training and testing activities occur	Field work 2011-15 Final analysis and reporting by December 2015

5.10 Recovery Actions in the Action Area

Recovery is the process by which species' ecosystems are restored and threats to the species are minimized such that ESA-listed species can be self-sustaining. This section addresses ongoing recovery actions that may compensate for effects from stressors in the *Environmental Baseline* and the action assessed in this opinion. Ongoing conservation actions for ESA-listed cetaceans include, but are not limited to, the following:

- NOAA Fisheries Alaska Protected Resources Division large whale disentanglement efforts (https://alaskafisheries.noaa.gov/protectedresources/entanglement/whale_entanglement_factsheet.pdf).
- Marine Mammal Viewing Guidelines and Regulations (http://alaskafisheries.noaa.gov/protectedresources/mmv/guide.htm);
- Research humpback population structure and abundance including the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) project.

Additionally, recovery actions for fish can be found in proposed ESA Recovery Plans (e.g., Proposed ESA Recovery Plan for Snake River Fall Chinook Salmon, NMFS 2015).

5.11 Scientific Research

Scientific research permits issued by the NMFS currently authorize studies on listed species in the North Pacific Ocean, some of which extend into portions of the Gulf of Alaska. Authorized research on ESA-listed whales and sea lions may include close vessel and aerial approaches, biopsy sampling, tagging, ultrasound, and exposure to acoustic activities, and breath sampling. Research activities involve non-lethal "takes" of these marine mammals by harassment. From 2009 to 2016, 72 scientific research permits for the Pacific Ocean were authorized for marine mammal species considered in this opinion, with 29 of these permits active today. Additionally, 55 of these permits are specific to the North Pacific Alaska region, with 22 currently active. None of these permits authorized intentional lethal takes of the these species, and three permits authorized unintentional lethal take of western DPS Steller sea lion Alaska.

5.12 The Impact of the Baseline on Listed Resources

This section summarizes the effects of past and present, Federal, state, or private actions and other human activities in the action area (Figure 2). Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed species considered in this opinion. Some of these stressors result in mortality or serious injury to individual animals (e.g., vessel strike, whaling, fisheries and bycatch), whereas others result in more indirect (e.g., a fishery that impacts prey availability, marine debris) or non-lethal (e.g., whale watching, anthropogenic sound, scientific research, climate change) impacts. There is also concern about recent mortalities in the population, a reduction in food (prey) availability and increasing stress from whale watchers and boaters. Assessing the aggregate impacts of these stressors on the species considered in this opinion is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that most of the species in this opinion are wide ranging and subject to stressors in locations well beyond the action area.

Based on information provided in Navy exercise reports, under the previous MMPA rule (2011 to 2016), the Navy's training activities in the Gulf of Alaska have resulted in estimated take that are well below the five-year levels evaluated in the previous biological opinion for the Gulf of Alaska TMAA. There have not been any vessel strikes of any species during training activities in the Gulf of Alaska during the five-year period.

We consider the best indicator of the aggregate impact of the *Environmental Baseline* on ESA-listed resources to be the status and trends of those species. As noted in Table 12, some of the species considered in this opinion are seeing increases in population abundance, some are declining, and for some, the status remains unknown. Taken together, this indicates that the

Environmental Baseline is impacting species in different ways. For the species that are increasing in population abundance, they are doing so in light of potential impacts of aspects of the *Environmental Baseline*. Therefore, while aspects of the *Environmental Baseline* described previously may slow their recovery, recovery is not being prevented. For the species that may be declining in abundance, it is possible that, cumulatively, the conditions described in the *Environmental Baseline* are preventing their recovery. However, it is also possible that their populations are at such low levels (e.g., due to historic commercial whaling) that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, they may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself. A thorough review of the status and trends of each species is presented in the *Status of Listed Resources* section of this opinion.

6 EFFECTS OF THE ACTION ON ESA-LISTED SPECIES AND CRITICAL HABITAT

Section 7 regulations define "effects of the action" as the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur. This effects analyses section is organized following the stressor, exposure, response, risk assessment framework.

The ESA defines "take" as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct." 16 U.S.C. § 1532(19). Harm is further defined by regulation to include "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering." 50 C.F.R. 222.102. NMFS has not yet defined "harass" under the ESA in regulation. However, on December 21, 2016, NMFS issued interim guidance on the term "harass," defining it as an action that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering." NMFS relied on this definition of "harass" to evaluate whether the proposed activities are likely to harass fish species considered in this opinion.

For marine mammal species, NMFS' consultations with the Navy and NMFS (Permits and Conservation Division) regarding the effects of the Navy's testing and training activities and NMFS' issuance of regulations and LOAs pursuant to the MMPA have long relied on outputs from NAEMO modeling to quantify instances of harassment (see, *e.g.*, NMFS' biological opinions for Hawaii and Southern California training and testing activities, Mariana Islands

training and testing activities, and Northwest training and testing activities). The NAEMO model uses acoustic criteria to estimate the number of responses that could qualify as Level B harassment under the MMPA.[5] Therefore, NMFS has relied on the MMPA definition of Level B harassment in estimating the number of instances of harassment of ESA-listed marine mammals in prior consultations. The Navy requested initiation of formal consultation for its Phase II GOA training activities in February of 2015, almost two years prior to issuance of the interim guidance. Further, data and information gathering for GOA modeling began in November 2011 and modeling occurred over a 20 month period from October 2012 to June 2014. Given how far into this consultation the interim guidance was issued and the complexity associated with modeling take estimates of marine mammals, consistent with prior consultations for Navy testing and training activities, NMFS continues to rely on the MMPA definition of Level B harassment and the NAEMO model outputs to evaluate whether the proposed activities are likely to harass ESA-listed species and to estimate the number of instances of harassment of ESA-listed marine mammals considered in this opinion.

We note that as the definition of Level B harassment is currently applied, including in this Opinion, a wide range of behavioral reactions may qualify as harassment, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, temporary disruption of feeding, migrating, or reproductive behaviors. The modeled estimates of Level B harassment calculated using the behavioral response function do not differentiate between the different types of potential behavioral reactions. Nor do the estimates provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. We therefore consider the available scientific evidence to determine the likely nature of the modeled behavioral responses and the potential fitness consequences for affected individuals.

For all species considered in this opinion, we rely upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

[5] For military readiness activities, Level B harassment under the MMPA means: "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered." 16 U.S.C. 1362(18)(B).

6.1 Stressors Associated with the Proposed Action

The potential stressors (risks) to ESA-listed species that we analyzed based on the training activities the Navy proposes to conduct in the Gulf of Alaska TMAA are summarized in Table 16.

Table 16. Navy Stressor Categories Analyzed in this Opinion

Stressor	Description of Stressor
Acoustic (sonar and other active acoustic sources, underwater explosives, weapons firing, launch and impact noise, aircraft noise, and vessel noise)	Effects on species from acoustic sources are dependent on a number of factors, including the type of sound received, the proximity of the animal to the sound source, and the duration, frequency, and intensity of the sound. Underwater sound propagation is highly dependent upon environmental characteristics such as bathymetry, bottom type, water depth, temperature, and salinity. The sound received at a particular location will be different than near the source due to the interaction of many factors, including propagation loss; how the sound is reflected, refracted, or scattered; the potential for reverberation; and interference due to multi-path propagation. Sonar and other active acoustic sources emit sound waves into the water to detect objects, safely navigate, and communicate. Most systems operate within specific frequencies (although some harmonic frequencies may be emitted at lower sound pressure levels). Most sonar use is associated with anti-submarine warfare (ASW) activities. Sonar use associated with mine warfare (MIW) would also contribute a notable portion of overall acoustic sound. Explosives used during training activities include explosive ordnance, such as bombs, missiles, and naval gun shells; torpedoes; demolition charges; and explosive sonobuoys. Depending on the activity, detonations would occur in the air, near the water's surface, or underwater (some torpedoes and sonobuoys). Demolition charges could occur near the surface, in the water column, or on the seafloor. Most detonations would occur in waters greater than 200 ft. (61 m) in depth, and greater than 3 nm from shore, although MIW, demolition, and some training detonations could occur in shallow water closer to shore. Detonations associated with ASW would typically occur in waters greater than 600 ft. (183 m) depth. Noise associated with weapons firing and the impact of non-explosive practice munitions (NEPM) could happen at any location within the action area but generally would occur at locations greater than 12 nm from shore for safety reasons. These training events would occur in areas designated for anti-surface warfare and similar activities. The firing of a weapon may have several components of associated noise. Firing of guns could include sound generated by firing the gun (muzzle blast), vibration from the blast propagating through a ship's hull, and sonic booms generated by the projectile flying through the air. Missiles and targets would also produce noise during launch. In addition, the impact of NEPM at the water surface can introduce noise into the water. Fixed- and rotary-wing aircraft are used for a variety of training activities throughout the action area, contributing both airborne and underwater sound to the ocean environment. Aircraft used in training generally have reciprocating, turboprop, or jet engines. Motors, propellers, and rotors produce the most noise, with some noise contributed by aerodynamic turbulence. Aircraft sounds have more energy at lower frequencies. Takeoffs and landings occur at established airfields as well as on vessels at sea throughout the action area. Most aircraft noise would be produced around air fields in the range complex. Military activities involving aircraft generally are dispersed over large expanses of open ocean but can be highly concentrated in time and location. Vessels (including ships, small boats, and submarines) would produce low-frequency, broadband underwater sound. Overall, naval traffic is often a minor component of total vessel traffic (Mintz and Filadelfo 2011) (Mintz and Parker 2006). Commercial vessel traffic, which includes cargo vessels, bulk carriers, passenger vessels, and oil tankers (all over 65 ft. [20 m] in length), is heaviest near and between the major shipping ports.

Stressor	Description of Stressor
Energy (electromagnetic devices)	Electromagnetic devices are used in towed or unmanned MIW systems that mimic the electromagnetic signature of a vessel passing through the water. None of the devices include any type of electromagnetic "pulse." The devices work by emitting an electromagnetic field and mechanically generated underwater sound to simulate the presence of a ship. The sound and electromagnetic signature cause nearby mines to detonate. The static magnetic field generated by the electromagnetic devices is of relatively minute strength. Typically, the maximum magnetic field generated would be approximately 23 gauss (G). By comparison, magnetic field generated by a refrigerator magnet is between 150 and 200 G. The strength of an electromagnetic field decreases quickly with distance from the device. The magnetic field generated at a distance of 4 m from the source is comparable to the earth's magnetic field, which is approximately 0.5 G.

Stressor	Description of Stressor
Physical disturbance and strike (vessels, in water devices, military expended materials)	Physical disturbances, including direct strikes on marine animals, may occur in association with vessel movements, the use of in-water devices, and materials expended from vessels and aircraft. Vessels used as part of the Action include ships (e.g., aircraft carriers, surface combatants, protection vessels), support craft, small boats, and submarines, ranging in size from 5 to over 300 m. Large Navy ships generally operate at speeds in the range of 10 to 15 knots, and submarines generally operate at speeds in the range of 8 to 13 knots. Small boats (for purposes of this discussion, less than 65 ft. [12 m] in length), which are all support craft, have variable speeds. Locations of vessel use in the action area varies with the type of activity taking place. In-water devices as discussed in this analysis are unmanned vehicles, such as remotely operated vehicles, unmanned surface vehicles and unmanned undersea vehicles, and towed devices. These devices are self-propelled and unmanned or towed through the water from a variety of platforms, including helicopters and surface ships. In-water devices are generally smaller than most participating vessels ranging from several inches to about 15 m. These devices can operate anywhere from the water surface to the benthic zone. Certain devices do not have a realistic potential to strike marine animals because they either move slowly through the water column (e.g., most unmanned undersea vehicles) or are closely monitored by observers manning the towing platform (e.g., most towed devices). Military expended materials include: (1) all sizes of NEPM; (2) fragments from explosive munitions; and (3) expended materials other than munitions, such as sonobuoys, and expendable targets. Activities using NEPM (e.g., small-, medium-, and large-caliber gun ammunitions, missiles, rockets, bombs, torpedoes, and neutralizers), explosive munitions (generating munitions fragments), and materials other than munitions (e.g., flares, chaff, sonobuoys, decelerators/parachutes, aircraft stores and ballast, and targets) have the potential to contribute to the physical disturbance and strike stressor either in-air or in-water or both, depending on how the device is used. Aircraft and aerial targets used in Navy training activities are separated into four categories: (1) fixed-wing aircraft, (2) rotary-wing aircraft, (3) unmanned aircraft systems, and (4) aerial targets. Fixed-wing aircraft include, but are not limited to, planes such as P-3, P-8, E/A-6B, E/A-18G, and F-35. Rotary-wing aircraft are generally helicopters, such as MH-60. Unmanned aircraft systems include a variety of platforms, including but not limited to, the Small Tactical Unmanned Aircraft System—Tier II, Broad Area Maritime Surveillance unmanned aircraft, Fire Scout Vertical Take-off and Landing Unmanned Aerial Vehicle, and the Unmanned Combat Air System. Aerial targets include remotely operated airborne devices, most of which are designed to be recovered for reuse. However, if they are used during activities that utilize explosives, then they may result in fragments. Expendable aerial targets that may result in fragments include air-launched decoys. Aircraft and aerial target strikes are only applicable to birds. Seafloor devices include moored mine shapes, anchors, bottom placed instruments, and robotic vehicles referred to as "crawlers." Seafloor devices are either stationary or move very slowly along the bottom and do not pose a threat to highly mobile organisms.

Stressor	Description of Stressor
Entanglement (fiber optic cables and guidance wires, and decelerators/parachutes)	The only type of cable expended during training are fiber optic cables. Fiber optic cables are flexible, durable, and abrasion or chemical-resistant. The physical characteristics of the fiber optic material render the cable brittle and easily broken when kinked, twisted, or bent sharply (i.e., to a radius greater than 360 degrees). The fiber optic cable would be suspended within the water column during the activity, and then be expended to sink to the sea floor. The only types of wires expended during training activities are guidance wires from heavy-weight torpedoes. Guidance wires are used to help the firing platform control and steer the torpedo. They trail behind the torpedo as it moves through the water or air. Finally, the guidance wire is released from both the firing platform and the torpedo then sinks to the ocean floor. The torpedo guidance wire is a single-strand, thin gauge, coated copper alloy. The tensile breaking strength of the wire is a maximum of 42 pounds (lb.) (19 kilograms [kg]) and can be broken by hand (Group 2005). The length of wire dispensed would generally be equal to the distance the torpedo travels to impact the target and any undispensed wire would be contained in the dispensers upon impact. Degradation rates for the wire may vary because of changing environmental conditions in seawater, but are likely to take between 12 and 45 months Aircraft-launched sonobuoys, lightweight torpedoes (such as the MK 46 and MK 54), illumination flares, and targets use nylon parachutes or decelerators ranging in size from 18 to 48 in. (46 to 122 cm) in diameter. Decelerators are made of cloth and nylon, and many have weights attached to the lines for rapid sinking. At water impact, the decelerator assembly is expended, and it sinks away from the unit. The decelerator assembly may remain at the surface for 5–15 seconds before the decelerator and its housing sink to the seafloor, where it becomes flattened (Group 2005). Some decelerators are weighted with metal clips that facilitate their descent to the seafloor. Once settled on the bottom the canopy may temporarily billow if bottom currents are present.

Stressor	Description of Stressor
Ingestion (munitions and military expended materials other than munitions)	The only munitions small enough for a marine mammal or fish to ingest are small- and medium-caliber projectiles. These projectiles include all sizes up to and including 2.25 in. (57 mm) in diameter. Projectiles are composed of solid metal materials and would quickly move through the water column and settle on the seafloor where they are most likely to be encountered by bottom foraging animals. Sinking projectiles are unlikely to be encountered in the water column by marine mammals or fish. Many different types of explosive munitions can result in fragments that are expended at sea during training activities. Types of explosive munitions that can result in fragments include demolition charges, grenades, projectiles, missiles, and bombs. Fragments would result from fractures in the munitions casing and would vary in size depending on the size of the net explosive weight and munition type; however, typical sizes of fragments are unknown. These solid metal materials would quickly sink through the water column and settle to the seafloor. Military expended materials other than munitions include target fragments, chaff, and flares. At-sea targets are usually remotely operated airborne, surface, or subsurface traveling units, most of which, but not all, are designed to be recovered for re-use. However, if they are used during activities that utilize explosives then they may result in fragments. Expendable targets that may result in fragments would include air-launched decoys, surface targets (such as marine markers, paraflares, cardboard boxes, and 10 ft. [3.05 m] diameter red balloons), and mine shapes. Most target fragments would sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time Chaff consists of reflective, aluminum-coated glass fibers used to obscure ships and aircraft from radar-guided systems. Chaff, which is stored in canisters, is either dispensed from aircraft or fired into the air from the decks of surface ships when an attack is imminent. The glass fibers create a radar cloud that mask the position of the ship or aircraft. Chaff is composed of an aluminum alloy coating on glass fibers of silicon dioxide (Force 1997). Chaff is released or dispensed in cartridges or projectiles that contain millions of fibers. When deployed, a diffuse cloud of fibers is formed that is undetectable to the human eye. Chaff is a very light material, similar to fine human hair. It can remain suspended in air anywhere from 10 minutes to 10 hours and can travel considerable distances from its release point, depending on prevailing atmospheric conditions (Arfsten et al. 2002; Force 1997). Chaff cartridge plastic end caps and pistons would also be released into the marine environment, where they would persist for long periods and could be ingested by marine animals. Chaff end caps and pistons sink in saltwater (Spargo 2007). Flares are pyrotechnic devices used to defend against heat-seeking missiles, where the missile seeks out the heat signature from the flare rather than the aircraft's engines. Similar to chaff, flares are also dispensed from aircraft and fired from ships. The flare device consists of a cylindrical cartridge approximately 1.4 in. (3.6 cm) in diameter and 5.8 in. (14.7 cm) in length. Flares are designed to burn completely. The only material that would enter the water would be a small, round, plastic end cap (approximately 1.4 in. [3.6 cm] in diameter).

Stressor	Description of Stressor
Secondary (explosion byproducts, metals, chemicals, sedimentation, and transmission of marine mammal diseases and parasites)	Secondary stressors associated with some training activities could pose indirect impacts to ESA-listed marine species through habitat degradation or alteration or an effect on prey availability. Effects to habitat and prey availability may result from: (1) explosives, (2) explosion byproducts and unexploded ordnance, (3) metals, (4) chemicals, and (5) transmission of marine mammal diseases and parasites. In addition to directly impacting marine species, underwater explosions could impact other species in the food web, including prey species that ESA-listed marine species feed upon. The impacts of explosions would differ depending upon the type of prey species in the area of the detonation. Indirect impacts of explosives and unexploded ordnance to marine species via degradation of sediment or water quality is possible in the immediate vicinity of the ordnance. Explosion byproducts are not toxic to marine organisms at realistic exposure levels (Rosen and Lotufo 2010) . Relatively low solubility of most explosives and their degradation products means that concentrations of these contaminants in the marine environment are relatively low and readily diluted. Metals are introduced into seawater and sediments as a result of training activities involving ship hulks, targets, ordnance, munitions, and other military expended materials. Several training activities introduce potentially harmful chemicals into the marine environment; principally, flares and propellants for rockets, missiles, and torpedoes. Properly functioning flares missiles, rockets, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures allow propellants and their degradation products to be released into the marine environment. The greatest risk to marine species would be from perchlorate released from flares, missile, and rockets that operationally fail. Perchlorate is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals.

6.1.1 Level of Annual Activities Introducing Stressors into the Action Area

Table 2 provides an overview of training activity levels by stressor.

6.1.2 Summary of Effect Determinations By Stressor

Table 17 below summarizes our final determinations of effect by stressor category. Previously in section 4.3 of this opinion, we summarized our determination of effects by species and designated critical habitat. We provided the status of those resources that were considered further in our Risk Analysis. Here we further summarize which stressors are likely to adversely affect the species as carried forward in the Effects Section of this opinion. We also summarize those that are not likely to adversely affect ESA-listed species. We concurred with Navy's NLAA determinations for all stressors except its determinations for explosives and some ESUs/DPSs of salmonids. The stressors that were determined to be LAA and the contributing training activities are likely to result in take of one or more ESA-listed species.

Table 17. Summary of NMFS' Effect Determinations by Stressor - Training Activities

Common Name	Overall ESA Determination	Effect Determinations by Stressor (TRAINING ACTIVITIES)															
		Acoustic						Energy	Physical				Entanglement		Ingestion		Secondary
		Sonar and other non-impulsive sources	Explosives and other impulse sources	Weapons firing, launch and non-explosive impact noise	Vessel Noise	Aircraft Noise	Acoustic sources not analyzed in NAEMO	Electromagnetic devices	Vessels	In-water devices	Military expended materials	Seafloor Devices	Fiber optic cables and guidance wires	Decelerator/parachutes	Munitions	Military expended materials other than munitions	
MARINE MAMMALS																	
North Pacific right whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale – Western North Pacific DPS	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale – Mexico DPS	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Blue whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Fin whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sei whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Sperm whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Stellar Sea Lion – Western DPS	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
FISH																	
Coho salmon – Lower Columbia ESU	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Coho salmon – Oregon Coast ESU	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chum salmon – Hood Canal summer-run ESU	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chum salmon – Columbia River ESU	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Upper Columbia River DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Middle Columbia River DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Lower Columbia River DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Upper Willamette River DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Snake River Basin DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead – Puget Sound DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

6.2 Stressors Not Likely to Adversely Affect ESA-listed Species

The following section discusses stressors that are not likely to adversely affect ESA-listed species. If a stressor is likely to adversely affect any of the ESA-listed species in the action area, it is carried forward in our effects analysis.

6.2.1 Vessel Strike

Vessel strikes from commercial, recreational, and Navy vessels are known to affect large whales and have resulted in serious injury and occasional fatalities to cetaceans (Berman-Kowalewski et al. 2010; Calambokidis 2012; Douglas et al. 2008a; Laggner 2009; Lammers et al. 2003). Reviews of the literature on ship strikes mainly involve collisions between commercial vessels and whales (e.g., Laist et al. (2001), Jensen and Silber (2004b)). The ability of any ship to detect a marine mammal and avoid a collision depends on a variety of factors, including environmental conditions, ship design, size, speed, and manning, as well as the behavior of the animal. Records of collisions date back to the early 17th century, and the worldwide number of collisions appears to have increased steadily during recent decades (Laist et al. 2001; Ritter 2012).

Vessel speed, size and mass are all important factors in determining potential impacts of a vessel strike to marine mammals. For large vessels, speed and angle of approach can influence the severity of a strike. Based on modeling, Silber et al. (2010) found that whales at the surface experienced impacts that increased in magnitude with the ship's increasing speed. Results of the study also indicated that potential impacts were not dependent on the whale's orientation to the path of the ship, but that vessel speed may be an important factor. At ship speeds of 15 knots or higher (7.7 m/second), there was a marked increase in intensity of centerline impacts to whales. Results also indicated that when the whale was below the surface (about one to two times the vessel draft), there was a pronounced propeller suction effect. This suction effect may draw the whale into the hull of the ship, increasing the probability of propeller strikes (Silber et al. 2010).

There has never been a vessel strike to a whale or pinniped during any of the training activities in the Gulf of Alaska TMAA. There have been Navy strikes of large whales in areas outside the TMAA, such as Hawaii and Southern California. However, these areas differ significantly from the TMAA given that both Hawaii and Southern California have a much higher number of Navy vessel activities and much higher densities of large whales.

Key points in discussion of participating vessels in relationship to potential ship strike include:

- Many military ships have their bridges positioned closer to the bow, offering better visibility ahead of the ship (compared to a commercial merchant vessel).
- There are often aircraft associated with the training or training activity, which can more readily detect cetaceans in the vicinity of a vessel or ahead of a vessel's present course before crew on the vessel would be able to detect them.

- Military ships are generally more maneuverable than commercial merchant vessels, and if cetaceans are spotted in the path of the ship, would be capable of changing course more quickly. Military ships generally operate at the slowest speed possible consistent with either transit needs or training or training needs. While minimum speed is intended as a fuel conservation measure particular to a certain ship class, secondary benefits include better ability to spot and avoid objects in the water including marine mammals. In addition, a standard operating procedure for Navy vessels is to maneuver the vessel to maintain a distance of at least 500 yd. (457 m) from any observed whale and to avoid approaching whales head-on, as long as safety of navigation is not imperiled.
- The crew size on military vessels is generally larger than merchant ships, allowing for the possibility of stationing more trained Lookouts on the bridge. At all times when vessels are underway, trained Lookouts and bridge navigation teams are used to detect objects on the surface of the water ahead of the ship, including cetaceans. Additional Lookouts, beyond those already stationed on the bridge and on navigation teams, are positioned as Lookouts during some training events.
- Lookouts receive extensive training including Marine Species Awareness Training, which instructs Lookouts to recognize marine species detection cues (e.g., floating vegetation or flocks of seabirds) as well as provides additional information to aid in the detection of cetaceans.

While it is possible for a vessel to strike a cetacean or pinniped during the course of training activities in Gulf of Alaska TMAA, we do not believe that a vessel strike of a cetacean or pinniped is reasonably likely to occur. As stated previously, the Navy has been training in the action area for years and no such incident has occurred. Additionally, the Navy and other vessels participating in training exercises employ minimization measures to reduce the likelihood for a surface vessel to strike a large whale or pinniped (i.e., lookouts, minimium approach distances as discussed in section 2.3 of this opinion). The location of stellar sea lions (i.e., more likely to be found in habitats closer to shore) and their mobility reduce the potential for vessel strike. Consequently, NMFS has determined that the likelihood of vessel strike during training over the five-year period of the MMPA rule and continuing into the reasonably foreseeable future is sufficiently low so as to be discountable. Because the likelihood of vessel strike is so low as to be discountable, vessel strike is not likely to adversely affect ESA-listed marine mammals and will not be considered further in this opinion.

As noted in the *Status of Listed Resources* section of this opinion, some juvenile salmonids from the Columbia River may be subject to mortality from ship wakes. This was only identified as a threat to these species in the Columbia River estuary, primarily for juvenile fry that are less than 60 mm long and that rear inches from the shore (NMFS 2011b). The ESA-listed salmonids that will occur in the action area will be larger than 60 mm and the action area is well offshore and not in close proximity to estuaries where juvenile salmonids would be rearing. By the time any ESA-listed salmonids reach the action area, they would be highly mobile, and would be expected to actively avoid any oncoming vessels associated with Navy activities. Therefore, the likelihood of a vessel strike to an ESA-listed fish species is so low as to be discountable and is not likely to

adversely affect the ESA-listed fish species considered in this opinion. Vessel strike of ESA-listed fish species is not considered further in this opinion.

6.2.2 Acoustic Sources Excluded from the Navy's Quantiative Analysis

As mentioned in section 2.2.4, the Navy did not include a number of acoustic sources in their acoustic effects analysis. These sources included Doppler sonar/speed logs, fathometers, hand-held sonar, imaging sonar, acoustic modems and tracking pingers, acoustic releases, side scan sonar, and small impulsive sources (i.e., < 0.25 lb NEW). A full description of each of these sources is included in the 2016 Gulf of Alaska FSEIS/OEIS, along with additional rationale as to why these sources were not included in the Navy's acoustic effects analysis. In general, these sources are of low source level (i.e., < 160 dB), narrow beam width, and many are of a frequency outside of the hearing range of ESA-listed marine mammals (i.e., > 200 kHz). Even in the off chance marine mammals were exposed and the source was within the animal's hearing range, the source is expected to result in no more than a brief behavioral response that is inconsequential to the animal due to the acoustic characteristics of the sources considered (e.g., short pulse length, narrow beamwidth, downward directed beam) and manner of system operation. For the small impulsive sources, quantitative modeling in multiple locations has validated that these low level impulsive sources are expected to cause no more than short-term and inconsequential responses in marine species due to the low explosive weight and corresponding very small zone of influence associated with these types of sources (Navy 2016a). Injury or mortality would not be expected. In summary, the Navy did not include these sources in their acoustic effects modeling analysis because they determined exposures to those sources would be unlikely, and if an exposure did occur, the response would be inconsequential to the animal. The Navy also did not request authorization from NMFS Permits and Conservation Division to take marine mammals incidental to the use of these sources and no take of marine mammals was included in the proposed MMPA rule. No injury or mortality is expected to result from the use of these sources. We agree that the types of behavioral responses that could occur from exposure to these sources would not rise to the level of take under the ESA. For these reasons, the potential effects of these other acoustic sources on ESA-listed species are insignificant, and not likely to adversely affect them.

6.2.3 Effects of Weapons Firing, Launch and Impact Noise

Ship fired munitions can create blast waves from the gun muzzle and along the trajectory of the shell but it is expected the noise will only be detectable to marine mammals and fish within a very small footprint along the trajectory. Aircraft fired munitions are not expected to have sound waves emanating from the firing source that would be of sufficient intensity to propagate a sound wave into the water. Non-explosive ordnance can also impact the water with substantial force and produce loud noises. Marine mammals and ESA-listed fish within the audible range of activities involving weapons firing, launch, and impact noise may exhibit a behavioral startle

response but are expected to quickly return to normal behavior. Activities involving weapons firing, launch, and impact noise are sporadic events of short duration reducing the likelihood of subjecting individual marine mammals or ESA-listed fish to prolonged or repeated exposures. Due to the short duration and sporadic nature of munition firing, the low likelihood that an ESA-listed animal would be in close enough proximity to detect sound from munition firing above water, and the high likelihood that an ESA-listed animal able to detect noise from weapons firing would react very briefly and quickly resume normal activities, any behavioral responses would be insignificant and would not rise to the level of take. Because the potential effects of weapons firing, launch, and impact noise are insignificant, these potential stressors are not likely to adversely affect ESA-listed species or designated critical habitat and will not be considered further in this opinion.

6.2.4 Effects of Aircraft Noise

Aircraft noise in the TMAA would either be from fixed-wing aircraft overflights or helicopters. Fixed-wing aircraft flights generally have altitudes ranging from 6,000 feet to 30,000 feet. Typical airspeeds range from very low (less than 100 knots [185.2 km/hr]) to high subsonic (less than 600 knots [1,111.2 km/hr]). Some training in the TMAA will also involve supersonic flight which produces sonic booms, but this would not occur below 15,000 ft (4,572 m). Helicopter overflights occur at lower altitudes; for some training activities as low as 75 feet above the water's surface, though this is relatively infrequent, and would not occur near Western DPS Steller sea lion rookeries or haulouts. Most helicopter overflights occur above 500 feet.

6.2.4.1 *Cetaceans*

Studies have shown that aircraft presence and operation can result in changes in behavior of cetaceans (Arcangeli and Crosti 2009; Holt et al. 2009; Luksenburg and Parsons 2009b; Noren et al. 2009; Patenaude et al. 2002; Richter et al. 2006; Richter et al. 2003b; Smultea et al. 2008). Many of the activities the U.S. Navy conducts in the Gulf of Alaska TMAA involve some level of activity from aircraft that include helicopters, maritime patrols, and fighter jets. Low-flying aircraft produce sounds that marine mammals can hear when they occur at or near the ocean's surface. Helicopters generally tend to produce sounds that can be heard at or below the ocean's surface more than fixed-wing aircraft of similar size and larger aircraft tend to be louder than smaller aircraft. Underwater sounds from aircraft are strongest just below the surface and directly under the aircraft. Sounds from aircraft would not have physical effects on marine mammals but represent acoustic stimuli (primarily low-frequency sounds from engines and rotors) that have been reported to affect the behavior of some marine mammals.

Rather than estimating the number of endangered or threatened species that are likely to be exposed to noise from aircraft overflight, take-offs and landings from carriers, or other fixed or rotary-wing aircraft operations at altitudes low enough for the sounds to be salient at or

immediately below the ocean's surface, we conservatively assumed that any ESA-listed species that occur in the action area during training activities that involve aircraft could be exposed to minor acoustic stimuli associated with aircraft traffic during one training event 21 days in duration in the April to October timeframe.

Several authors have reported that sperm whales did not react to fixed-wing aircraft or helicopters in some circumstances (Au and Perryman 1982b; Clarke 1956b; Gambell 1968; Green et al. 1992a) and reacted in others (Clarke 1956b; Fritts et al. 1983; Mullin et al. 1991; Patenaude et al. 2002; Richter et al. 2006; Richter et al. 2003a; Smultea et al. 2008; Wursig et al. 1998). Richardson et al. (1985b) reported that bowhead whales responded behaviorally to fixed-wing aircraft that were used in their surveys and research studies when the aircraft were less than 457 m above sea level; their reactions were uncommon at 457 m, and were undetectable above 610 meters. They also reported that bowhead whales did not respond behaviorally to helicopter overflights at about 153 m above sea level.

Smultea et al. (2008) studied the response of sperm whales to low-altitude (233 to 269 m) flights by a small fixed-wing airplane near Kauai and reviewed data available from other studies. They concluded that sperm whales responded behaviorally to aircraft passes in about 12 percent of encounters. All of the reactions consisted of sudden dives and occurred when the aircraft was less than 360 m from the whales (lateral distance). They concluded that the sperm whales had perceived the aircraft as a predatory stimulus and responded with defensive behavior. In at least one case, Smultea et al. (2008) reported that the sperm whales formed a semi-circular "fan" formation that was similar to defensive formations reported by other investigators.

In a review of aircraft noise effects on marine mammals, Luksenburg and Parsons (2009a) determined that the sensitivity of whales and dolphins to aircraft noise may depend on the animals' behavioral state at the time of exposure (e.g. resting, socializing, foraging or travelling) as well as the altitude and lateral distance of the aircraft to the animals. While resting animals seemed to be disturbed the most, low flying aircraft with close lateral distances over shallow water elicited stronger disturbance responses than higher flying aircraft with greater lateral distances over deeper water ((Patenaude et al. 2002; Smultea et al. 2008) in Luksenburg and Parsons (2009a)).

Thorough reviews on the behavioral reactions of marine mammals to aircraft and missile overflight are presented in Richardson et al. (1995d), Efroymson et al. (2000), Luksenburg and Parsons (2009b), and Holst et al. (2011). The most common responses of cetaceans to aircraft overflights were short surfacing durations, abrupt dives, and percussive behavior (breaching and tail slapping) (Nowacek et al. 2007). Other behavioral responses such as flushing and fleeing the area of the source of the noise have also been observed (Holst et al. 2011; Manci et al. 1988). Richardson et al. (1995d) noted that marine mammal reactions to aircraft overflight largely consisted of opportunistic and anecdotal observations. These observations lack a clear distinction

between reactions potentially caused by the noise of the aircraft and the visual cue an aircraft presents. In addition, it was suggested that variations in the responses noted were due to other undocumented factors associated with overflight (Richardson et al. 1995d). These factors could include aircraft type (single engine, multi-engine, jet turbine), flight path (centered on the animal, off to one side, circling, level and slow), environmental factors such as wind speed, sea state, cloud cover, and locations where native subsistence hunting continues.

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Efroymson et al. 2000; Koski et al. 1998). Richardson et al. (1995d) reported that while data on the reactions of mysticetes is meager and largely anecdotal, there is no evidence that single or occasional aircraft flying above mysticetes causes long-term displacement of these mammals. In general, overflights above 1,000 ft. (305 m) do not cause a reaction.

Bowhead whales in the Beaufort Sea exhibited a transient behavioral response to fixed-wing aircraft and vessels. Reactions were frequently observed at less than 1,000 ft. (305 m) above sea level, infrequently observed at 1,500 ft. (457 m), and not observed at 2,000 ft. (610 m) above sea level (Richardson et al. 1995d). Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 492 ft. (150 m) or higher. It should be noted that bowhead whales may have more acute responses to anthropogenic activity than many other marine mammals since these animals are often presented with limited egress due to limited open water between ice floes. Additionally, many of these animals may be hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

Variable responses to aircraft have been observed in toothed whales, though overall little change in behavior has been observed during flyovers. Toothed whale responses to aircrafts include diving, slapping the water with their flukes or flippers, swimming away from the direction of the aircraft, or not visibly reacting (Richardson et al. 1995d).

During standard marine mammal surveys at an altitude of 750 ft. (229 m), some sperm whales remained on or near the surface the entire time the aircraft was in the vicinity, while others dove immediately or a few minutes after being sighted. Other authors have corroborated the variability in sperm whales' reactions to fixed-wing aircraft or helicopters (Green et al. 1992b; Richter et al. 2006; Richter et al. 2003b; Smultea et al. 2008; Wursig et al. 1998). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al. 1995d). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 ft. [244 to 335 m]) by moving closer together and forming a defensive fan-shaped semicircle, with their heads facing outward. Several individuals in the group turned on their sides, apparently to look up toward the aircraft (Smultea et al. 2008). Whale-watching aircraft apparently caused sperm whales to turn more sharply but did not affect blow interval, surface

time, time to first click, or the frequency of aerial behavior (Richter et al. 2003b). Navy aircraft do not fly at low altitude, hover over, or follow whales and so are not expected to evoke this type of response.

Smaller delphinids generally react to overflights either neutrally or with a startle response (Wursig et al. 1998). The same species that show strong avoidance behavior to vessel traffic (*Kogia* species and beaked whales) also react to aircraft (Wursig et al. 1998). Beluga whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns to a greater extent than mysticetes in the same area (Patenaude et al. 2002). These reactions increased in frequency as the altitude of the helicopter dropped below 492 ft. (150 m).

Exposure to fixed-wing aircraft noise would be brief (seconds) as an aircraft quickly passes overhead at relatively high speeds. Exposure to helicopter overflights may last longer and would have a higher likelihood of causing a behavioral response from a cetacean due to the lower flight altitudes and longer duration the helicopter could be in proximity to an animal. However, exposures to both sorts of aircraft would be infrequent based on the transitory and dispersed nature of the overflights; repeated exposure to individual animals over a short period of time (hours or days) is extremely unlikely. Furthermore, the sound exposure levels would be relatively low to marine mammals that spend the majority of their time underwater. Based on the literature described above (e.g., Nowacek et al. 2007) and the nature of Navy aircraft use in the TMAA (i.e., Navy aircraft do not fly at low altitude, hover over, or follow whales), cetaceans exposed to aircraft noise could exhibit a short-term behavioral response (if they respond at all), but we do not expect these short term behavioral reactions to rise to the level of take. For these reasons, the effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed cetaceans considered in this opinion. The potential effect of aircraft noise on ESA-listed cetaceans will not be considered further in this opinion.

6.2.4.2 ***Pinnipeds***

As with whales, we assume Stellar sea lions that occur in the action area during training activities that involve aircraft are likely to be exposed to minor acoustic stimuli associated with aircraft traffic. However, there are few studies of the responses of marine animals to air traffic and the few that are available have produced mixed results. Some investigators report responses while others report no responses.

Richardson et al. (1995) noted that data on pinniped reactions to aircraft overflight largely consisted of opportunistic and anecdotal observations. Richardson et al.'s (1995) summary of this variable data note that responsiveness generally was dependent on the altitude of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). Hauled out pinnipeds exposed to aircraft sight or sound often react by becoming alert and in

many cases rushing into the water. Stampedes resulting in mortality to pups (by separation or crushing) have been noted in some cases although it is rare. However, there are no haul outs or rookeries in the TMAA and none of the overflight activities in the TMAA would take place near a haul out or rookery location.

Similar to cetaceans, exposure of pinnipeds to fixed-wing aircraft noise would be brief (seconds) as an aircraft quickly passes overhead at relatively high speeds. Exposure to helicopter overflights may last longer and have a higher likelihood of causing a behavioral response from a pinniped due to the lower flight altitudes and longer duration the helicopter could be in proximity to an animal. However, exposure to both sorts of aircraft would be infrequent based on the transitory and dispersed nature of the overflights; repeated exposure to individual animals over a short period of time (hours or days) is extremely unlikely. Furthermore, the sound exposure levels would be relatively low to marine mammals that spend the majority of their time underwater when in the open water environment of the TMAA without haulouts or rookeries. In addition, for areas outside of the TMAA, the Navy complies with restrictions prohibiting fixed wing aircraft or helicopter flights or surface training activities within 3,000 ft (914 m) of Steller sea lion critical habitat, rookeries, or pinniped haulout areas (DoN 2011). Based on the literature described above and the nature of Navy aircraft use in the TMAA, pinnipeds exposed to aircraft noise could exhibit a short-term behavioral response (if they respond at all), but we do not expect these short term behavioral reactions to rise to the level of take. For these reasons, the effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed pinnipeds considered in this opinion. The potential effect of aircraft noise on ESA-listed pinnipeds will not be considered further in this opinion.

6.2.4.3 *Fish*

The lack of substantial sound propagation into the water column from aircraft indicates there is a low probability of exposing fish to aircraft noise at perceivable levels. In the event an ESA-listed fish species was exposed to aircraft noise, it would likely result in very minor temporary behavioral responses such as a startle response. These types of behavioral responses would not rise to the level of take. For these reasons, the effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed fish considered in this opinion. The potential effect of aircraft noise on ESA-listed fish will not be considered further in this opinion.

6.2.5 Effects of Vessel Noise

Sounds emitted by large vessels can be characterized as low-frequency, continuous, or tonal, and sound pressure levels at a source will vary according to speed, burden, capacity and length (Kipple and Gabriele 2007; McKenna et al. 2012b; Richardson et al. 1995d). Vessels ranging from 135 to 337 m (Nimitz-class aircraft carriers, for example, have lengths of about 332 m)

generate peak source sound levels from 169 to 200 dB between 8 Hz and 430 Hz. Source levels for 593 container ship transits were estimated from long-term acoustic recording received levels in the Santa Barbara shipping channel, and a simple transmission loss model using Automatic Identification System data for source-receiver range (McKenna et al. 2013). Ship noise levels could vary 5 to 10 dB depending on transit conditions. Given the sound propagation of low frequency sounds, a large vessel in this sound range can be heard 139 to 463 km away (Polefka 2004). Hatch et al. (2008) measured commercial ship underwater noise levels and reported average source level estimates (71 to 141 Hz, root-mean-square pressure re 1 uPa ± standard error) for individual vessels ranged from 158 ± 2 dB (research vessel) to 186 ± 2 dB (oil tanker). McKenna et al (2012b) in a study off Southern California documented different acoustic levels and spectral shapes observed from different modern ship-types.

Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Amaral and Carlson 2005b; Au and Green 2000b; Bain et al. 2006; Bauer 1986; Bejder et al. 1999; Bejder and Lusseau. 2008; Bejder et al. 2009; Bryant et al. 1984; Corkeron 1995; Erbe 2002b; Félix 2001; Goodwin and Cotton 2004; Lemon et al. 2006; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2003b; Scheidat et al. 2004; Simmonds 2005b; Watkins 1986; Williams et al. 2002b; Wursig et al. 1998). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Based on the suite of studies of cetacean behavior to vessel approaches (Acevedo 1991b; Aguilar Soto et al. 2006; Arcangeli and Crosti 2009; Au and Perryman 1982a; Au and Green 2000b; Bain et al. 2006; Bauer and Herman 1986; Bejder et al. 1999; Bejder et al. 2006a; Bejder et al. 2006b; Bryant et al. 1984; Christiansen et al. 2010; Corkeron 1995; David 2002; Erbe 2002b; Felix 2001; Goodwin and Cotton 2004; Hewitt 1985; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Noren et al. 2009; Nowacek et al. 2001; Richter et al. 2006; Richter et al. 2003b; Scheidat et al. 2004; Simmonds 2005b; Stensland and Berggren 2007; Stockin et al. 2008; Watkins 1986; Williams and Ashe 2007; Williams et al. 2009; Williams et al. 2002b; Wursig et al. 1998), the set of variables that help determine whether marine mammals are likely to be disturbed by surface vessels include:

- *Number of vessels*. The behavioral repertoire marine mammals have used to avoid interactions with surface vessels appears to depend on the number of vessels in their perceptual field (the area within which animals detect acoustic, visual, or other cues) and

the animal's assessment of the risks associated with those vessels (the primary index of risk is probably vessel proximity relative to the animal's flight initiation distance) (Sims et al. 2012). Below a threshold number of vessels (which probably varies from one species to another, although groups of marine mammals probably share sets of patterns), studies have shown that whales will attempt to avoid an interaction using horizontal avoidance behavior. Above that threshold, studies have shown that marine mammals will tend to avoid interactions using vertical avoidance behavior, although some marine mammals will combine horizontal avoidance behavior with vertical avoidance behavior (Bryant et al. 1984; David 2002; Kruse 1991; Lusseau 2003; Nowacek et al. 2001; Stensland and Berggren 2007; Williams and Ashe 2007);

- *The distance between vessel and marine mammals* when the animal perceives that an approach has started and during the course of the interaction (Au and Perryman 1982a; David 2002; Hewitt 1985; Kruse 1991; Lundquist et al. 2012; Lusseau 2003; Tseng et al. 2011);
- *The vessel's speed and vector* (David 2002);
- *The predictability of the vessel's path*. That is, cetaceans are more likely to respond to approaching vessels when vessels stay on a single or predictable path (Acevedo 1991a; Angradi et al. 1993; Browning and Harland. 1999; Lusseau 2003; Lusseau 2006; Williams et al. 2002a) than when it engages in frequent course changes (Evans et al. 1994; Lusseau 2006; Williams et al. 2002a);
- *Noise associated with the vessel* (particularly engine noise) and the rate at which the engine noise increases (which the animal may treat as evidence of the vessel's speed) (David 2002; Lusseau 2003; Lusseau 2006; Polagye et al. 2011);
- *The type of vessel* (displacement versus planing), which marine mammals may be interpret as evidence of a vessel's maneuverability (Goodwin and Cotton 2004);
- *The behavioral state of the marine mammals* (David 2002; Lusseau 2003; Lusseau 2006; Wursig et al. 1998). For example, Würsig et al. (Wursig et al. 1998) concluded that whales were more likely to engage in avoidance responses when the whales were milling or resting than during other behavioral states (e.g., feeding).

Most of the investigations reported that animals tended to reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies (Corkeron 1995; Lundquist et al. 2012; Lusseau 2003; Lusseau 2004; Nowacek et al. 2001; Van Parijs and Corkeron 2001; Williams et al. 2002a; Williams et al. 2002b). In the process, their dive times increased, vocalizations and jumping were reduced (with the exception of beaked whales), individuals in groups move closer together, swimming speeds increased, and their direction of travel took them away from the source of disturbance (Baker and Herman 1989; Edds and Macfarlane 1987; Evans et al. 1992; Kruse 1991). Some individuals also dove and remained motionless, waiting until the vessel moved past their location. Most animals

finding themselves in confined spaces, such as shallow bays, during vessel approaches tended to move towards more open, deeper waters (Kruse 1991). We assume that this movement would give them greater opportunities to avoid or evade vessels as conditions warranted.

Although most of these studies focused on small cetaceans (for example, bottlenose dolphins, spinner dolphins, spotted dolphins, harbor porpoises, beluga whales, and killer whales), studies of large whales have reported similar results for fin and sperm whales (David 2002). Baker et al. (1983) reported that humpbacks in Hawaii responded to vessels at distances of 2 to 4 km. Richardson et al. (1985a) reported that bowhead whales (*Balaena mysticetus*) swam in the opposite direction of approaching seismic vessels at distances between 1 and 4 km and engage in evasive behavior at distances under 1 km. Fin whales also responded to vessels at a distance of about 1 km (Edds and Macfarlane 1987). A study by Lundquist (2012) on dusky dolphins concluded that repeated disturbance from tour vessel traffic may interrupt social interactions, and postulated that those repeated disturbances may carry energetic costs, or otherwise affect individual fitness. However, they were unable to determine if such disturbances were likely to cause long-term harm. For behavioral responses to result in energetic costs that result in long-term harm, such disturbances would likely need to be sustained for a significant duration or extent where individuals exposed would not be able to select alternate habitat to recover and feed. Navy training activities in the Gulf of Alaska would not likely result in such prolonged exposures and preclusion of individuals from feeding, breeding, or sheltering habitat.

Würsig et al. (1998) studied the behavior of cetaceans in the northern Gulf of Mexico in response to survey vessels and aircraft. They reported that false killer whales either did not respond or approached the ship (most commonly to ride the bow). Four of 15 sperm whales avoided the ship while the remainder appeared to ignore its approach.

Because of the number of vessels involved in U.S. Navy training activities, the vessel speed, and the use of course changes as a tactical measure with the associated sounds, the available evidence leads us to expect marine mammals to treat Navy vessels as stressors. Further, without considering differences in sound fields associated with any active sonar that is used during these exercises, the available evidence suggests that major training activities, unit- and intermediate-level exercises, and training activities would represent different stress regimes because of differences in the number of vessels involved, vessel maneuvers, and vessel speeds.

Sound emitted from large vessels, such as shipping and cruise ships, is the principal source of low-frequency noise in the ocean today, and marine mammals are known to react to or be affected by that noise (Foote et al. 2004; Hatch and Wright 2007b; Hildebrand 2005a; Holt et al. 2008b; Kerosky et al. 2013; Melcon et al. 2012; Richardson et al. 1995d).

In short-term studies, researchers have noted changes in resting and surface behavior states of cetaceans to whale watching vessels (Acevedo 1991b; Aguilar Soto et al. 2006; Arcangeli and

Crosti 2009; Au and Green 2000b; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams and Noren 2009). Noren et al. (2009) conducted research in the San Juan Islands in 2005 and 2006 and their findings suggested that close approaches by vessels impacted the whales' behavior and that the whale-watching guideline minimum approach distance of 100 m may be insufficient in preventing behavioral responses. Most studies of this type are opportunistic and have only examined the short-term response to vessel sound and vessel traffic (Magalhaes et al. 2002; Noren et al. 2009; Richardson and Wursig 1995; Watkins 1981d).

Long-term and cumulative implications of vessel sound on marine mammals remains largely unknown. Clark et al. (2009) provided a discussion on calculating the cumulative impacts of anthropogenic noise on baleen whales and estimated the noise from the passage of two vessels could reduce the optimal communication space for North Atlantic right whales by 84 percent (see also (Hatch et al. 2012).

Navy combatant vessels have been designed to generate minimal noise and use ship quieting technology to elude detection by enemy passive acoustic devices (Mintz and Filadelfo 2011; Southall et al. 2005). Given this, and that they are much fewer in number than their commercial counterparts, Navy vessels are a small component of overall vessel traffic and vessel noise in most areas where they operate (see Mintz and Filadelfo (2011) for a general summary for the U.S. Exclusive Economic Zone).

6.2.5.1 *Mysticetes*

Fin whales may alter their swimming patterns by increasing speed and heading away from a vessel, as well as changing their breathing patterns in response to a vessel approach (Jahoda et al. 2003). Vessels that remain 328 ft. (100 m) or farther from fin and humpback whales were largely ignored in one study where whale watching activities are common (Watkins 1981a). Only when vessels approached more closely did the fin whales in this study alter their behavior by increasing time at the surface and exhibiting avoidance behaviors. Other studies have shown when vessels are near, some but not all fin whales change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au and Green 2000b; Castellote et al. 2012b; Richter et al. 2003b; Williams et al. 2002b).

Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012b) demonstrated that fin whales' songs had shortened duration and decreased bandwidth, center frequency, and peak frequency in the presence of high shipping noise levels. It is not known if these changes in vocal behavior corresponded to other behaviors.

In the Watkins (1981a) study, humpback whales did not exhibit any avoidance behavior but did exhibit minor behavioral reactions to vessel presence. In a study of regional vessel traffic, Baker et al. (1983) found that when vessels were in the area, the respiration patterns of the humpback whales changed. The whales also exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 1.24 and 2.48 mi. (2,000 and 4,000 m) away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were within approximately 1.2 mi. (2,000 m; (Baker and Herman 1983)). Similar findings were documented for humpback whales when approached by whale watch vessels in Hawaii (Au and Green 2000b).

Gende et al. (2011) reported on observations of humpback whales in inland waters of Southeast Alaska subjected to frequent cruise ship transits (i.e., in excess of 400 transits in a 4-month season in 2009). The study was focused on determining if close encounter distance was a function of vessel speed. The reported observations, however, seem in conflict with other reports of avoidance at much greater distance so it may be that humpback whales in those waters are more tolerant of vessels (given their frequency) or are engaged in behaviors, such as feeding, that they are less willing to abandon. This example again highlights that context is critical for predicting and understanding behavioral reactions as concluded by Southall et al. (2007b) and Ellison et al. (2012).

Sei whales have been observed ignoring the presence of vessels and passing close to them (National Marine Fisheries Service 1993). In the presence of approaching vessels, blue whales perform shallower dives accompanied by more frequent surfacing, but otherwise do not exhibit strong reactions (Calambokidis et al. 2009a). Minke whales in the Antarctic did not show any apparent response to a survey vessel moving at normal cruising speeds (about 12 knots) at a distance of 5.5 nm; however, when the vessel drifted or moved at very slow speeds (about 1 knot), many whales approached it (Leatherwood et al. 1982a).

Although not expected to be in the Gulf of Alaska TMAA, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004a) and therefore might provide insight to behavioral responses of other baleen whales. North Atlantic right whales continue to use habitats in high vessel traffic areas (Nowacek et al. 2004a). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Nowacek et al. 2004a; Terhune and Verboom 1999). Although this may minimize potential disturbance from passing ships, it does increase the whales' vulnerability to potential ship strike.

Using historical records, Watkins (1986) showed that the reactions of four species of mysticetes to vessel traffic and whale watching activities in Cape Cod had changed over the 25-year period examined (1957 through 1982). Reactions of minke whales changed from initially more positive reactions, such as coming towards the boat or research equipment to investigate, to more

'uninterested' reactions towards the end of the study. Finback [fin] whales, the most numerous species in the area, showed a trend from initially more negative reactions, such as swimming away from the boat with limited surfacing, to more uninterested (ignoring) reactions allowing boats to approach within 98.4 ft. (30 m). Right whales showed little change over the study period, with a roughly equal number of reactions judged to be negative and uninterested; no right whales were noted as having positive reactions to vessels. Humpback whales showed a trend from negative to positive reactions with vessels during the study period. The author concluded that the whales had habituated to the human activities over (Watkins 1986).

Mysticetes have been shown to both increase and decrease calling behavior in the presence of vessel noise. An increase in feeding call rates and repetition by humpback whales in Alaskan waters was associated with vessel noise (Doyle et al. 2008). Melcon et al. (2012) also recently documented that blue whales increased the proportion of time spent producing certain types of calls when vessels were present. Conversely, decreases in singing activity by humpback whales have been noted near Brazil due to boat traffic (Sousa-Lima and Clark 2008). The Central North Pacific stock of humpback whales is the focus of whale-watching activities in both its feeding grounds (Alaska) and breeding grounds (Hawaii). Regulations addressing minimum approach distances and vessel operating procedures are in place in Hawaii, however, there is still concern that whales may abandon preferred habitats if the disturbance is too high (Allen and Angliss 2010b).

The available information suggests that ESA-listed mysticetes are either not likely to respond to vessel noise (e.g., Nowacek et al. 2004a, Watkins 1981a) or are expected to respond only briefly if exposed to noise from Navy vessels. Expected behavioral responses include startle responses, brief avoidance behavior (e.g., Jahoda et al. 2003), or changes in respiration rate (e.g., Baker et al. 1983). Most avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect mysticete reactions to vessel noise to have any measurable effects on any individual's fitness and any such responses are not expected to rise to the level of a take. Therefore, the effects of vessel noise on ESA-listed mysticetes is insignificant and not likely to adversely affect them.

6.2.5.2 *Odontocetes*

Sperm whales generally react only to vessels approaching within several hundred meters; however, some individuals may display avoidance behavior, such as quick diving (Magalhaes et al. 2002; Wursig et al. 1998). One study showed that after diving, sperm whales showed a reduced timeframe from when they emitted the first click than before vessel interaction (Richter et al. 2006). Small whale-watching and research vessels generate more noise in higher frequency bands and are more likely to approach odontocetes directly, and to spend more time near the individual whale. Reactions to Navy vessels are not well documented, but smaller whale-watching and research boats have been shown to cause these species to alter their breathing intervals and echolocation patterns.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity (Holt et al. 2008a) as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado and Wartzok 2008). Likewise, modification of multiple vocalization parameters has been shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al. 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al. 2005). Killer whales off the northwestern coast of the United States have been observed to increase the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which has been suggested as a response to increased masking noise produced by the vessels (Foote et al. 2004). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. For example, the source level of killer whale vocalizations has been shown to increase with higher background noise levels associated with vessel traffic (Hotchkin and Parks 2013). In addition, calls with a high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al. 2008a).

Similar to mysticetes, the available information suggests that ESA-listed odontocetes in the TMAA are either not likely to respond to vessel noise or are expected to respond only briefly if exposed to noise from Navy vessels. Expected behavioral responses include brief avoidance behavior (e.g., Magalhaes et al. 2002; Wursig et al. 1998) or changes in vocal patterns (e.g., Holt et al. 2008a, Lesage et al. 1999). However, we expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect odontocete reactions to vessel noise to have any measurable effects on any individual's fitness and any such responses are not expected rise to the

level of a take. Therefore, the effects of vessel noise on ESA-listed odontocetes is insignificant and not likely to adversely affect them.

6.2.5.3 ***Pinnipeds***

Little is known about pinniped reactions to underwater non-impulsive sounds (Southall et al. 2007a) including vessel noise. In a review of reports on reactions of pinnipeds to small craft and ships, Richardson et al. (1995c) note that information on pinniped reactions is limited and most reports are based on anecdotal observations. Specific case reports in Richardson et al. (1995c) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007a) pinniped responses to vessels are affected by the context of the situation and by the animal's experience. In summary, pinniped's reactions to vessels are variable and reports include a wide entire spectrum of possibilities from avoidance and alert to cases where animals in the water are attracted and cases on land where there is lack of significant reaction suggesting "habituation" or "tolerance" of vessels (Richardson et al. 1995c).

A study of reactions of harbor seals hauled out on ice to cruise ship approaches in Disenchantment Bay, Alaska revealed that animals are more likely to flush and enter the water when cruise ships approach within 1,640 ft. (500 m) and four times more likely when the cruise ship approaches within 328 ft. (100 m) (Jansen et al. 2010). Navy vessels in the TMAA would not operate in vicinity of nearshore natural areas that are pinniped haul-out or rookery locations.

Information specific to the likely responses of pinnipeds to vessel noise is limited, but based on the information presented above, as well as the information presented previously on the likely responses of other marine mammals to vessel noise, ESA-listed pinnipeds are either not likely to respond to vessel noise (e.g., Richardson et al. 1995c) or are only expected to respond briefly. As with the cetacean species, we would expect Stellar sea lions to engage in avoidance behavior when surface vessels move toward them. Sea lions would likely reduce their visibility at the water's surface and move horizontally away from the source of disturbance. However, we expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect pinniped reactions to vessel noise to have any measurable effects on any individual's fitness or to rise to the level of a take. Therefore, the effects of vessel noise on ESA-listed pinnipeds is insignificant and not likely to adversely affect them.

6.2.5.4 ***Fish***

According to Popper et al. (2014a), there is no direct evidence of mortality or injury to fish from vessel noise. Further, TTS from continuous sound sources (e.g., vessel noise) has only been documented in fish species that have specializations for enhanced sensitivity to sound. None of the ESA-listed salmonids considered in this opinion are known to have these specializations.

Data for species which do not have these specializations have shown no TTS in response to long term exposure to continuous noise sources (Popper et al. 2014a). This includes a study of rainbow trout (*Oncorhynchus mykiss*) exposed to increased noise for nine months in an aquaculture facility. The study also did not document any negative effects on the health of the fish from this increased exposure to noise (Popper et al. 2014a; Wysocki et al. 2007b).

Popper et al. (2014a) suggest that low frequency vessel noise (primarily from shipping traffic) may mask sounds of biological importance. As described previously in this opinion, none of the ESA-listed salmonids considered in this opinion have hearing specializations (which would indicate they may rely heavily on hearing for essential life functions) and they are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, and orient in the water column (Popper et al. 2014a). Further, hearing is not thought to play a role in salmon migration (e.g., Putnam et al. 2013). Additionally, any potential masking would be temporary as both the fish and vessel would be transiting the action area (likely at different speeds and in different directions). For these reasons, we do not expect any short-term instances of masking to have any fitness consequences for any individual fish.

Vessel activity may result in changes in fish behavior (Popper et al. 2014a). Simpson et al. (2016) used laboratory and field experiments to study the potential effects of vessel noise on anti-predator behavior for the settlement life stage of the Ambon damselfish (*pomacentrus amboinensis*), a demersal, site-attached, coral reef fish. The authors found that motorboat noise can have detrimental effects on anti-predator behavior of Ambon damselfish, potentially as a result of increased stress. These results are not broadly applicable though, as the winners and losers in other predator-prey interactions will depend on the hearing sensitivities and noise tolerance levels of the species being considered, and the noise source (Simpson et al. 2016). It would be particularly difficult to generalize these results to the salmonid species considered in this opinion and the potential effect of Navy vessel noise on these species. First, we do not know the relative hearing sensitivities and noise tolerance levels between the salmon considered in this opinion and their predators in the action area. Additionally, the predator avoidance behavior of the Ambon damselfish (i.e., the ability to use reef structure to hide from predators) is quite different from the avoidance behavior of salmonids in pelagic offshore marine waters. Finally, it's worth noting that any potential effects of Navy vessel noise on predator-prey dynamics would be brief as both the fish and vessel would be transiting the action area (likely at different speeds and in different directions). Any behavioral responses to vessel noise are expected to be temporary (e.g., a startle response, brief avoidance behavior) and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. We do not expect these short term behavioral reactions to rise to the level of take. For these reasons, the effect of vessel noise that may result in behavioral reactions

is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

6.2.5.5 *Conclusion*

We recognize that Navy vessels incorporate quieting technologies that reduce their acoustic signature (relative to the acoustic signature of similarly sized vessels) in order to reduce their vulnerability to detection by enemy vessels (Southall 2005). Veirs et al. (2016) analyzed noise from vessel traffic in the Salish Sea and found that military vessels emitted some of the lowest source sound levels of all vessel types (with the exception of pleasure boats) analyzed. Nevertheless, we do not assume that any quieting technology would be sufficient to prevent marine mammals from detecting sounds produced by approaching Navy vessels and perceiving those sounds as predatory stimuli. We also consider evidence that factors other than received sound level, including the activity state of animals exposed to different sounds, the nature and novelty of a sound, and spatial relations between sound source and receiving animals (i.e., the exposure context) strongly affect the probability of a behavioral response (Ellison et al. 2012).

Behavioral disruptions of ESA-listed species resulting from the vessel noise are expected to be insignificant and temporary. Given the short duration of vessel noise stressors, the infrequency of this stressor, and the temporary nature of biological responses of marine mammals and ESA-listed fish to this stressor, ESA-listed species evaluated in this consultation are either not likely to respond or are not likely to respond in ways that could be of biological significance (the responses might represent an approach or attentive movement, a small change in orientation in the waters, etc.). Most avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives in marine mammals. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior. For these reasons, the effects of noise from Navy vessels on ESA-listed species would not rise to the level of take and are considered insignificant. The potential effects of vessel noise will not be considered further in this opinion.

6.2.6 Effects of Fiber Optic Cables, Guidance Wires, and Decelerators/Parachutes from Entanglement

Expended materials from U.S. Navy training may pose a risk of entanglement or ingestion to marine mammals and ESA-listed fish in the Gulf of Alaska TMAA. These interactions could occur at the sea surface, in the water column, or on the seafloor. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with military expended materials have the potential to result in negative sub-lethal effects, mortality, or result in no impact.

Expended materials from Navy training may include the following: fiber optic cables, guidance wires, parachutes, and targets. At-sea targets are usually remotely-operated airborne, surface, or subsurface traveling units, most of which are designed to be recovered for reuse. However, if they are used during activities that utilize high-explosives, they may fragment. Expendable targets that may fragment include air-launched decoys, surface targets (such as marine markers, paraflares, cardboard boxes, and 10 ft. diameter red balloons), and mine shapes. Most expended materials and target fragments are expected to sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time.

There has never been a reported or recorded instance of marine mammals or ESA-listed salmonids entangled in military expended materials. Though there is a potential for ESA-listed species to encounter military expended material, given the large geographic area involved, and the presumably low densities of threatened or endangered species in the Gulf of Alaska TMAA, we do not believe such interactions are likely to occur. Additionally, most expended materials are expected to sink upon release, and relatively few ESA-listed animals feed in the deepwater benthic habitats where most expended materials are likely to settle. While disturbance or strike from any expended material as it falls through the water column is possible, it is not likely because the objects will slow in velocity as they sink toward the bottom (e.g., guidance wires sink at an estimated rate of 0.7 ft. [0.2 m] per second), and can be avoided by highly mobile organisms such as marine mammals and fish. Bottom feeding animals have an increased likelihood of encountering expended materials because they may find them during feeding long after the training or training event has occurred.

If encountered, it is unlikely that an animal would get entangled in a fiber optic cable, guidewire, parachute, or other expended material while it was sinking or upon settling to the seafloor. An animal would have to swim through loops or become twisted within the cable or wire to become entangled, and given the properties of the expended cables and wires (low breaking strength and sinking rates) the probability of injury or mortality from expended materials is low. Specifically, fiber optic cable is brittle and would be expected to break if kinked, twisted or sharply bent. Thus, the physical properties of the fiber optic cable would not allow the cable to loop, greatly reducing the likelihood of entanglement of ESA-listed species. Based on degradation times, guidance wires would break down within one to two years and no longer pose an entanglement risk. The length of the guidance wires vary, but greater lengths increase the likelihood that a marine mammal could become entangled. Parachutes used by the Navy range in size from 18 to 48 in. (46 to 122 cm), but the vast majority of expended decelerator/parachutes are small (18 in.) cruciform shaped decelerators used with sonobuoys. They have short attachment lines and, upon water impact, may remain at the surface for 5 to 15 seconds before the decelerator/parachute and its housing sink to the seafloor. Entanglement of an animal in a parachute assembly at the surface or within the water column would be unlikely, since the parachute would have to land directly on an animal, or an animal would have to swim into it before it sinks. It is possible that a bottom

feeding animal (e.g., sperm whale) could become entangled when they are foraging in areas where parachutes have settled on the seafloor. For example, if bottom currents are present, the canopy may temporarily billow and pose a greater entanglement threat. However, the likelihood of currents causing a billowing of a parachute and being encountered by an ESA-listed species is so unlikely as to be considered discountable.

In conclusion, based on their deep-water location of use, their sinking rate, their degradation rate, and the comparably low density of ESA-listed species, the likelihood of ESA-listed species becoming entangled in expended materials is so low as to be discountable. Therefore, entanglement is not likely to adversely affect ESA-listed species and will not be considered further in this opinion.

6.2.7 Effects of Munitions and Other Military Expended Materials from Ingestion

The only munitions and other materials small enough to be ingested by ESA-listed marine mammals and ESA-listed fish are small- and medium-caliber projectiles, broken pieces of firing targets, chaff, flare caps, decelerators/parachutes, and shrapnel fragments from explosive ordnance. Most expendable materials will be used over deep water and these items will sink quickly and settle on the seafloor with the exception of chaff and some firing target materials. Given the limited time most items will spend in the water column it is not reasonably likely that these items will be accidentally ingested by ESA-listed species not accustomed to foraging on the sea floor. The ESA-listed species potentially exposed to expended munitions and shrapnel fragments while foraging on the sea floor in the deep water environment of the TMAA is limited to sperm whales (i.e., salmonids in the open water environment of the TMAA do not feed on the sea floor). Sperm whales are capable of foraging along the sea floor in deep water. However, the relatively low density of both sperm whales and expended materials along the vast sea floor suggests ingestion would be rare. In cases where sperm whales do accidentally ingest expended materials small enough to be eaten it is likely they will pass through the digestive tract and not result in an injury or an increased likelihood of injury from significant disruption of normal behavioral patterns such as breeding, feeding, or sheltering.

Chaff is composed of fine fibers of silicon dioxide coated with aluminum alloy. Due to their light weight and small size they float and can be carried great distances in both air and water currents. Their dispersal in wind and water results in chaff fibers likely occurring in low densities on the ocean surface. Given the small size, low densities, and low toxicity of chaff, any accidental ingestion by ESA-listed species feeding at the ocean surface is not expected to result in an injury or an increased likelihood of injury from significant disruption of normal behavioral patterns such as breeding, feeding, or sheltering. Firing target materials are normally retrieved before sinking and it is not reasonable to expect ingestion of these items to occur.

In conclusion, ESA-listed species are either so unlikely to ingest expended material as to be discountable, or in the case of bottom-feeding species, instances of ingestion would be rare and any ingested materials are likely to pass through the digestive tract without causing injury or any effects rising to the level of take. Therefore, the effects of ingested expended materials on ESA-listed species is either discountable, or insignificant, and not likely to adversely affect them. Potential effects of ingested expended materials will not be considered further in this opinion.

6.2.8 Effects of Electromagnetic Devices

The static magnetic field generated by the electromagnetic devices used in Navy training in the Gulf of Alaska TMAA is of relatively minute strength. Typically, the maximum magnetic field generated would be approximately 23 gauss (G). By comparison, magnetic field generated by a refrigerator magnet is between 150 and 200 G. The strength of an electromagnetic field decreases quickly with distance from the device. The magnetic field generated at a distance of 4 m from the source is comparable to the earth's magnetic field, which is approximately 0.5 G.

Normandeau et al. (2011) concluded there was behavioral, anatomical, and theoretical evidence indicating cetaceans sense magnetic fields. Most of the evidence in this regard is indirect evidence from correlation of sighting and stranding locations suggesting that cetaceans may be influenced by local variation in the earth's magnetic field (Kirschvink 1990; Klinowska 1985; Walker et al. 1992). Results from one study in particular showed that long-finned and short-finned pilot whales, striped dolphin, Atlantic spotted dolphin, Atlantic white-sided dolphin, fin whale, common dolphin, harbor porpoise, sperm whale, and pygmy sperm whale were found to strand in areas where the earth's magnetic field was locally weaker than surrounding areas (negative magnetic anomaly) (Kirschvink 1990). Results also indicated that certain species may be able to detect total intensity changes of only 0.05 microtesla (Kirschvink et al. 1986). This gives insight into what changes in intensity levels some species are capable of detecting, but does not provide experimental evidence of levels to which animals may physiologically or behaviorally respond. Further, not all physiological or behavioral responses are biologically significant and rise to the level of take as defined in the ESA.

Impacts to marine mammals associated with electromagnetic fields are dependent on the animal's proximity to the source and the strength of the magnetic field. Electromagnetic fields associated with naval training activities are relatively weak (only 10 percent of the earth's magnetic field at 79 ft.), temporary, and localized. Once the source is turned off or moves from the location, the electromagnetic field is gone. A marine mammal would have to be present within the electromagnetic field (approximately 656 ft. [200 m] from the source) during the activity in order to detect it. Again, detection does not necessarily signify a significant biological response rising to the level of take as defined under the ESA. Given the small area associated with mine fields, the infrequency and short duration of magnetic energy use, the low intensity of electromagnetic energy sources, and the density of cetaceans in these areas, the likelihood of

ESA-listed cetaceans or pinnipeds being exposed to electromagnetic energy at sufficient intensities to create a biologically relevant response is so low as to be discountable. Therefore, electromagnetic energy is not likely to adversely affect ESA-listed cetaceans or pinnipeds and will not be considered further in this opinion.

In a recent literature review on the effects of electromagnetic fields on marine species (Fisher and Slater 2010), teleost fish were noted to potentially use magnetic fields for orientation. There are several potential mechanisms that Pacific salmon use for navigation, including orienting to the Earth's magnetic field, utilizing a celestial compass, and using the odor of their natal stream to migrate back to their original spawning grounds (Groot and Margolis 1998; Quinn and Groot 1983). Crystals of magnetite have been found in four species of Pacific salmon, though not in sockeye salmon (Mann et al. 1988; Walker et al. 1988). These magnetite crystals are believed to serve as a compass that orients to the Earth's magnetic field. Putnam et al. (2013) provided empirical evidence that salmon use cues from the magnetic field to navigate in the open ocean. Quinn and Brannon (1982) conclude that while salmon can apparently detect B-fields, their behavior is likely governed by multiple stimuli as demonstrated by the ineffectiveness of artificial B-field stimuli. Supporting this, (Yano et al. 1997) found no observable effect on the horizontal and vertical movements of adult chum salmon that had been fitted with a tag that generated an artificial B-field around the head of each fish. Furthermore, research conducted by Ueda et al. (1998) on adult sockeye salmon suggests that, rather than magnetoreception, this species relies on visual cues to locate natal stream and on olfactory cues to reach its natal spawning channel. Blockage of magnetic sense had no effect on the ability of the fish to locate their natal stream.

As stated above, the strength of the electromagnetic devices used by the Navy is relatively minute and quickly dissipates at short distances away from the source. The devices work by emitting an electromagnetic field and mechanically generated underwater sound to simulate the presence of a ship. The magnetic field four meters away from the device is comparable to the Earth's magnetic field. Based on the small area around each electromagnetic device that will have an altered magnetic field, we assume that any potential disruption in an individual fish's orientation ability in the Gulf of Alaska TMAA would only occur very close to the source. Additionally, this disruption would be temporary and only last as long as the fish remains within the area where the magnetic field is altered (likely seconds). Further, most fish would be expected to avoid the device and associated vessel prior to entering the area where the magnetic field would be altered. Therefore, the likelihood of ESA-listed fish being exposed to electromagnetic energy at sufficient intensities to create a biologically relevant response is so low as to be discountable. We conclude that ESA-listed fish are not likely to be adversely affected by electromagnetic devices during Navy training activities. The potential effects of electromagnetic devices will not be considered further in this opinion.

6.2.9 Effects of In-water Devices and expended material from direct physical strikes

Despite thousands of Navy exercises in which torpedoes, in-water devices (unmanned surface vehicles, unmanned undersea vehicles, towed devices), or expended materials have been used, there have been no recorded instances of marine species strikes from these devices. Therefore, the likelihood of striking a marine mammal or ESA-listed fish is so unlikely as to be discountable. Any ESA-listed marine mammal or fish species is likely to exhibit behavioral avoidance of in-water devices in the event they are within range to detect them. However, the potential effect of behavioral reponses to avoid direct physical strike is insignificant. Any avoidance behavior would be of short duration and intensity such that it would not rise to the level of take. For these reasons, we conclude that ESA-listed fish and marine mammals are not likely to be adversely affected by electromagnetic devices during Navy training activities. The potential effects of electromagnetic devices will not be considered further in this opinion.

6.2.10 Seafloor devices

Seafloor devices include moored mine shapes, anchors, bottom placed instruments, and robotic vehicles referred to as "crawlers." Seafloor devices are either stationary or move very slowly along the bottom and do not pose a threat to highly mobile organisms. Objects falling through the water column will slow in velocity as they sink toward the bottom and would be avoided by ESA-listed species. The only seafloor device used during training activities that has the potential to strike an ESA-listed species at or near the surface is an aircraft deployed mine shape, which is used during aerial mine laying activities. These devices are identical to non-explosive practice bombs, therefore the analysis of the potential impacts from those devices are covered in the military expended material strike section. We consider the likelihood of mobile ESA-listed species being struck by a slow moving seafloor device to be so low as to be discountable. Any individuals encountering seafloor devices are likely to behaviorally avoid them. Given the slow movement of seafloor devices, the effort expended by individuals to avoid them will be minimal, temporary, and will not have fitness consequences. Therefore, behavioral avoidance of mobile seafloor devices by ESA-listed species will be insignificant and not rise to the level of take. For these reasons, we conclude that ESA-listed fish and marine mammals are not likely to be adversely affected by seafloor devices associated with the proposed action. The potential effects of seafloor devices will not be considered further in this opinion.

6.2.11 Secondary Stressors

This section analyzes potential impacts to marine mammals, and ESA-listed fish exposed to stressors indirectly through impacts to their habitat or prey. Stressors from Navy training activities could pose indirect impacts to ESA-listed species via habitat or prey. These include (1) explosives and byproducts, (2) sonar, (3) metals, and (4) chemicals. These secondary stressors have the potential to decrease water and sediment quality.

Underwater explosions may reduce available prey items for ESA-listed species by either directly killing prey or by scaring them from the area. Behavioral avoidance of explosive ordnance by prey species may facilitate behavioral avoidance of additional explosives by ESA-listed species as they follow their food source as it flees. This benefit would remove ESA-listed species from blast locations while not interrupting feeding behavior. Due to the infrequent use of underwater explosives and the limited locations at which underwater explosives are used, it is not expected their use will have a persistent effect on prey availability of the health of the aquatic food web.

High-order explosions consume most of the explosive material, creating typical combustion byproducts. Explosion by-products associated with high order detonations present no indirect stressors to marine ESA-listed species because most byproducts are common in seawater and the rest are quickly diluted below appreciable levels. Explosive byproducts are not expected to result in detectable changes in sediment or water quality. Low-order explosives leave more explosive material in the water but this material is not water soluble, degrades quickly, and is quickly diluted. The levels of explosive materials and byproducts are not detectable above background levels 1 to 2 m from a degrading source.

Due to the lack of fast rise times, lack of high peak pressures, and the lack of high acoustic impulse, sonar would not be expected to cause injury or mortality in the prey of ESA-listed species. Because injury and mortality of prey is not expected to result from sonar, prey would still be available in the environment following exposure to Navy activities and remain available to any ESA-listed species foraging in the area. Sonar has the potential to cause behavioral responses in the prey of ESA-listed species. Studies of caged fish have identified three basic behavioral reactions to sound: startle, alarm, and avoidance (McCauley et al. 2000; Pearson et al. 1992). Should the prey of an ESA-listed species elicit a behavioral reaction from exposure to sonar, we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-exposure behavior immediately following each exposure and still be available as forage for ESA-listed species.

Metals are introduced into seawater and sediments as a result of training activities involving the destruction of ship hulls, targets, ordnance, munitions, and other military expended materials. Concentrations of metals in sea water are lower than concentrations in sediments. Metals deposited on the sea floor will be buried in sediment and slowly degrade over time. Most metals used in Navy expendables are benign and all corroding metals would either be diluted into the ocean currents or be sequestered in the sediments immediately surrounding the source (Navy 2013). It is extremely unlikely that marine mammals would be indirectly impacted by metals via the water and few marine ESA-listed species feed primarily on the seafloor where they would come into contact with marine sediments.

Chemicals are introduced into the marine environment primarily through the degradation of expendables. Flares, missiles, rockets, and torpedoes combust most of their propellants and fuels; leaving benign or readily diluted soluble combustion byproducts; however, operational failures result in the release of propellants and degradation products to be released (Navy 2013). The greatest risk to marine mammals from flares, missile, and rocket propellants that operationally fail is perchlorate, which is highly soluble in water, persistent, and impacts metabolic processes. However, due to the infrequent use of flares, missiles, and rocket propellants and the limited locations at which these devices are used, perchlorate would be expected to quickly dilute below appreciable levels. Propylene glycol dinitrate and nitrodiphenylamine (toxic component of torpedo fuel) adsorb to sediments, have relatively low toxicity, and are readily degraded by biological processes (Navy 2013).

Given the information provided above regarding the potential for explosives and byproducts, metals, chemicals, and transmission of disease and parasites to indirectly affect marine ESA-listed cetacean, pinniped, and fish species through habitat and prey availability impacts, the likelihood of ESA-listed species being exposed to toxic levels of explosives, explosive byproducts, metals, and other chemicals resulting from Gulf of Alaska activities are so unlikely as to be considered discountable. Therefore, secondary stressors from Gulf of Alaska activities are not likely to adversely affect ESA-listed species and secondary stressors will not be considered further in this opinion.

6.3 Risk Assessment Framework – Marine Mammals

The following is a summary of available information used to develop the Navy's risk assessment criteria for acoustic stressors. We subsequently reviewed and adopted the criteria for this risk analysis.

6.3.1 Direct Injury of Marine Mammals From Acoustic Stressors

The potential for direct injury of marine mammals has been inferred from terrestrial mammal experiments and from post-mortem examination of marine mammals believed to have been exposed to underwater explosions (Ketten et al. 1993; Richmond et al. 1973; Yelverton et al. 1973). Additionally, non-injurious effects on marine mammals (e.g., TTS) are extrapolated to injurious effects (e.g., PTS) based on data from terrestrial mammals to derive the criteria serving as the potential for injury (Southall et al. 2007b). Actual effects on marine mammals may differ from terrestrial animals due to anatomical and physiological adaptations to the marine environment, such as a reinforced trachea and flexible thoracic cavity (Ridgway and Dailey 1972) that may decrease the risk of lung injury.

Potential non-auditory direct injury from non-impulsive sound sources, such as sonar, is unlikely due to relatively lower peak pressures and slower rise times than potentially injurious impulsive

sources such as explosives. Non-impulsive sources also lack the strong shock waves associated with explosions. Therefore, primary blast injury and barotrauma (i.e., injuries caused by large pressure changes; discussed below) would not occur from exposure to non-impulsive sources such as sonar. Further, though there have been marine mammal strandings associated with use of sonar, as Ketten (2012) has recently summarized, "to date, there has been no demonstrable evidence of acute, traumatic, disruptive, or profound auditory damage in any marine mammal as the result [of] anthropogenic noise exposures, including sonar." The theories of sonar induced acoustic resonance and sonar induced bubble formation are discussed below. These phenomena, if they were to occur, would require the co-occurrence of a precise set of circumstances that are unlikely to occur.

6.3.2 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue effects is primary blast injury and barotrauma after exposure to high amplitude impulsive sources, such as explosions. Primary blast injuries result from the initial compression of a body exposed to a blast wave. Primary blast injury is usually limited to gas-containing structures (e.g., lung and gut) and the auditory system (Craig Jr. 2001a; Craig Jr. and Hearn 1998). Barotrauma refers to injuries caused when large pressure changes occur across tissue interfaces, normally at the boundaries of air-filled tissues such as the lungs. Primary blast injury to the respiratory system may be fatal depending upon the severity of the trauma. Rupture of the lung may introduce air into the vascular system, producing air emboli that can restrict oxygen delivery to the brain or heart. Though often secondary in life-threatening severity to pulmonary blast trauma, the gastrointestinal tract can also suffer contusions and lacerations from blast exposure, particularly in air-containing regions of the tract. Although hemorrhage of solid organs (e.g., liver, spleen, and kidney) from blast exposure is possible, rupture of these organs is rarely encountered.

The only known occurrence of mortality or injury to a marine mammal due to Navy training or training involving impulsive sources occurred in March 2011 at the Silver Strand Training Complex. Prior to this incident, this area had been used for underwater demolitions training for at least three decades without incident. On this occasion, however, a group of long-beaked common dolphins entered the mitigation zone surrounding an area where a time-delayed firing device had been initiated on an explosive with a net explosive weight of 8.76 lb (3.97 kg) placed at a depth of 48 ft. (14.6 m). Approximately 1 minute after detonation, three animals were observed dead at the surface; a fourth animal was discovered 3 days later stranded dead 42 nm to the north of the detonation. Upon necropsy, all four animals were found to have sustained typical mammalian primary blast injuries (Danil and St. Leger 2011). See the Gulf of Alaska FSEIS/OEIS for more information on the topic of stranding. Since this incident, the Navy, in consultation with NMFS, developed additional mitigation measures to minimize the potential for similar incidents in the future. Further, it is worth noting that activities similar to those that caused dolphin mortalities at

Silver Strand will not be conducted in the Gulf of Alaska TMAA. For these reasons, primary blast injury and barotrauma are unlikely to occur during the training activities proposed to be conducted in the Gulf of Alaska TMAA.

6.3.3 Auditory Trauma

Relatively little is known about auditory system trauma in marine mammals resulting from a known sound exposure. A single study spatially and temporally correlated the occurrence of auditory system trauma in humpback whales with the detonation of a 5,000 kg (11,023 lb.) explosive (Ketten et al. 1993). The exact magnitude of the exposure in this study cannot be determined, but it is likely the trauma was caused by the shock wave produced by the explosion. There are no known occurrences of direct auditory trauma in marine mammals exposed to tactical sonar or other non-impulsive sound sources (Ketten 2012). The potential for auditory trauma in marine mammals exposed to impulsive sources (e.g., explosions) is inferred from tests of submerged terrestrial mammals exposed to underwater explosions (Ketten et al. 1993; Richmond et al. 1973; Yelverton et al. 1973). As documented further in section 6.6 and 6.7 of this opinion, Navy training activities including the use of sonar and explosives are only expected to result in behavioral harassment and temporary threshold shift of ESA-listed marine mammals. Auditory trauma is not expected to occur.

6.3.4 Acoustic Resonance

Acoustic resonance has been proposed as a hypothesis suggesting that acoustically induced vibrations (sound) from sonar or sources with similar operating characteristics could be damaging tissues of marine mammals. In 2002, NMFS convened a panel of government and private scientists to investigate the issue (NMFS 2002). They modeled and evaluated the likelihood that Navy mid-frequency sonar caused resonance effects in beaked whales that eventually led to their stranding in the Bahamas (DoN 2015a). The conclusions of that group were that resonance in air-filled structures was not likely to have caused the stranding (NMFS 2002). The frequencies at which resonance was predicted to occur were below the frequencies utilized by the mid-frequency sonar systems associated with the Bahamas event. Furthermore, air cavity vibrations, even at resonant frequencies, were not considered to be of sufficient amplitude to cause tissue damage, even under the worst-case scenario in which air volumes would be undamped by surrounding tissues and the amplitude of the resonant response would be maximal. These same conclusions would apply to other training activities involving acoustic sources. Therefore, we conclude that acoustic resonance is not likely under realistic conditions during training activities and this type of impact is not considered further in this analysis.

6.3.5 Bubble Formation (Acoustically Induced)

A suggested cause of injury to marine mammals is rectified diffusion (Crum and Mao 1996), the process of increasing the size of a bubble by exposing it to a sound field. The process is dependent upon a number of factors including the sound pressure level and duration. Under this hypothesis, one of three things could happen: (1) bubbles grow to the extent that tissue hemorrhage (injury) occurs, (2) bubbles develop to the extent an immune response is triggered or nervous tissue is subjected to enough localized pressure that pain or dysfunction occurs (a stress response without injury), or (3) the bubbles are cleared by the lung without negative consequence to the animal. The probability of rectified diffusion, or any other indirect tissue effect, will necessarily be based upon what is known about the specific process involved. Rectified diffusion is more likely if the environment in which the ensonified bubbles exist is supersaturated with gas. Repetitive diving by marine mammals can cause the blood and some tissues to accumulate gas to a greater degree than is supported by the surrounding environmental pressure (Ridgway and Howard 1979). The dive patterns of some marine mammals (e.g., beaked whales) are theoretically predicted to induce greater supersaturation (Houser 2010; Houser et al. 2001b). If rectified diffusion were possible in marine mammals exposed to high-level sound, conditions of tissue supersaturation could theoretically speed the rate of bubble growth and increase the size of the bubbles. Subsequent effects due to tissue trauma and emboli would presumably mirror those observed in humans suffering from decompression sickness. It is unlikely the short duration of sonar or explosion sounds would be long enough to drive bubble growth to any substantial size, if such a phenomenon occurs.

An alternative but related hypothesis has also been suggested: stable microbubbles could be destabilized by high-level sound exposures such that bubble growth then occurs through static diffusion of gas out of the tissues. In such a scenario, the marine mammal would need to be in a gas-supersaturated state for enough time for bubbles to become a problematic size. Recent research with *ex vivo* supersaturated bovine tissues suggested that for a 37 kHz signal, a sound exposure of approximately 215 dB re 1 μPa would be required before microbubbles became destabilized and grew (Crum et al. 2005). Assuming spherical spreading loss and a nominal sonar source level of 235 dB re 1 μPa at 1 m, a whale would need to be within 10 m (33 ft.) of the sonar dome to be exposed to such sound levels. Furthermore, tissues in the study were supersaturated by exposing them to pressures of 400 to 700 kilopascals for hours and then releasing them to ambient pressures. Assuming the equilibration of gases with the tissues occurred when the tissues were exposed to the high pressures, levels of supersaturation in the tissues could have been as high as 400 to 700 percent. These levels of tissue supersaturation are substantially higher than model predictions for marine mammals (Houser et al. 2001a; Saunders et al. 2008). It is improbable this mechanism is responsible for stranding events or traumas associated with beaked whale strandings. Both the degree of supersaturation and exposure levels observed to cause microbubble destabilization are unlikely to occur.

There is considerable disagreement among scientists as to the likelihood of this phenomenon (Evans and Miller 2004; Piantadosi and Thalmann 2004). Although it has been argued that traumas from beaked whale strandings are consistent with gas emboli and bubble-induced tissue separations (Fernandez et al. 2005a; Jepson et al. 2003), bubble formation as the cause of the traumas has not been verified. The presence of bubbles postmortem, particularly after decompression, is not necessarily indicative of bubble pathology (Bernaldo de Quiros et al. 2012; Dennison et al. 2011; Moore et al. 2009).

As documented further in section 6.6 and 6.7 of this opinion, Navy training activities including the use of sonar and explosives are only expected to result in behavioral harassment and temporary threshold shift in ESA-listed marine mammals. Bubble formation is not expected to occur.

6.3.6 Nitrogen Decompression

Although not a direct injury, variations in marine mammal diving behavior or avoidance responses could possibly result in nitrogen tissue supersaturation and nitrogen off-gassing. Nitrogen supersaturation and off-gassing levels could result in deleterious vascular and tissue bubble formation (Hooker et al. 2012; Jepson et al. 2003; Saunders et al. 2008). Nitrogen off-gassing occurring in human divers is called decompression sickness. The mechanism for bubble formation from saturated tissues would be indirect and also different from rectified diffusion, but the effects would be similar. The potential process for this to occur is hypothetical and under debate in the scientific community (Hooker et al. 2012; Saunders et al. 2008). It is speculated if exposure to a startling sound elicits a rapid ascent to the surface, tissue gas saturation sufficient for the evolution of nitrogen bubbles might result (Fernandez et al. 2005a; Hooker et al. 2012; Jepson et al. 2003). In this scenario, the rate of ascent would need to be sufficiently rapid to compromise behavioral or physiological protections against nitrogen bubble formation.

Previous modeling suggests even unrealistically rapid rates of ascent from normal dive behaviors are unlikely to result in supersaturation to the extent bubble formation would be expected in beaked whales (Zimmer and Tyack 2007). Tyack et al. (2006) suggested emboli observed in animals exposed to mid-frequency active (MFA) sonar (Fernandez et al. 2005a; Jepson et al. 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth at which lung collapse would occur. A bottlenose dolphin was trained to repetitively dive to elevate nitrogen saturation to the point that asymptomatic nitrogen bubble formation was predicted to occur. However, inspection of the vascular system of the dolphin via ultrasound did not demonstrate the formation of any nitrogen gas bubbles (Houser 2010).

More recently, modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of tissues (e.g., fat, bone lipid) to the point that they are supersaturated when the animals are at the surface (Hooker et al. 2009; Saunders et

al. 2008). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al. 2006; Hooker et al. 2009). Since bubble formation is facilitated by compromised blood flow, it has been suggested that rapid stranding may lead to bubble formation in animals with supersaturated tissues because of the stress of stranding and the cardiovascular collapse that can accompany it (Houser 2010).

A fat embolic syndrome was identified by Fernandez et al. (2005b) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals, and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream. Recently, Dennison et al. (2011) reported on investigations of dolphins stranded in 2009 and 2010.Using ultrasound the authors identified gas bubbles in kidneys from 21 of 22 live-stranded dolphins and in the liver from two of 22. The authors postulated stranded animals are unable to recompress by diving, and thus may retain bubbles that are otherwise re-absorbed in animals that can continue to dive. The researchers concluded minor bubble formation can be tolerated since the majority of stranded dolphins released did not re-strand (Dennison et al. 2011). Recent modeling by Kvadsheim (2012) determined behavioral and physiological responses to sonar have the potential to result in bubble formation. However, the observed behavioral responses of cetaceans to sonar do not imply any significantly increased risk over what may otherwise occur normally in individual marine mammals. As a result, no marine mammals addressed in this analysis are given differential treatment due to the possibility for acoustically mediated bubble growth.

6.3.7 Hearing Loss

The most familiar effect of exposure to high intensity sound is hearing loss, meaning an increase in the hearing threshold. Both auditory injury and auditory fatigue may result in hearing loss. The meaning of the term "hearing loss" does not equate to "deafness." Hearing loss is a noise-induced threshold shift, or simply a threshold shift. If high-intensity sound over stimulates tissues in the ear, causing a threshold shift, the impacted area of the ear (associated with and limited by the sound's frequency band) no longer provides the same auditory impulses to the brain as before the exposure (Ketten 2012). The distinction between PTS and TTS is based on whether there is a complete recovery of a threshold shift following a sound exposure. If the threshold shift eventually returns to zero (the threshold returns to the pre-exposure value), the threshold shift is a TTS.

For TTS, full recovery of the hearing loss (to the pre-exposure threshold) has been determined from studies of marine mammals, and this recovery occurs within minutes to hours for the small amounts of TTS that have been experimentally induced (Finneran et al. 2005; Finneran and Schlundt 2010; Nachtigall et al. 2004). The recovery time is related to the exposure duration, sound exposure level, and the magnitude of the threshold shift, with larger threshold shifts and

longer exposure durations requiring longer recovery times (Finneran et al. 2005; Finneran and Schlundt 2010; Mooney et al. 2009a; Mooney et al. 2009b). In some cases, threshold shifts as large as 50 dB (loss in sensitivity) have been temporary, although recovery sometimes required as much as 30 days (Ketten 2012). If the threshold shift does not return to zero but leaves some finite amount of threshold shift, then that remaining threshold shift is a PTS. Again for clarity, PTS, as discussed in this document, is not the complete loss of hearing, but instead is the loss of hearing sensitivity over a particular range of frequency. Figure 11 shows one hypothetical threshold shift that completely recovers, a TTS, and one that does not completely recover, leaving some PTS. The actual amount of threshold shift depends on the amplitude, duration, frequency, temporal pattern of the sound exposure, and on the susceptibility of the individual animal.

Many are familiar with hearing protection devices (i.e., ear plugs) required in many occupational settings where pervasive noise could otherwise cause auditory fatigue and possibly result in hearing loss. The mechanisms responsible for auditory fatigue differ from auditory trauma and would primarily consist of metabolic fatigue and exhaustion of the hair cells and cochlear tissues. Note that the term "auditory fatigue" is often used to mean TTS; however, the Navy uses a more general meaning to differentiate fatigue mechanisms (e.g., metabolic exhaustion and distortion of tissues) from trauma mechanisms (e.g., physical destruction of cochlear tissues occurring at the time of exposure).

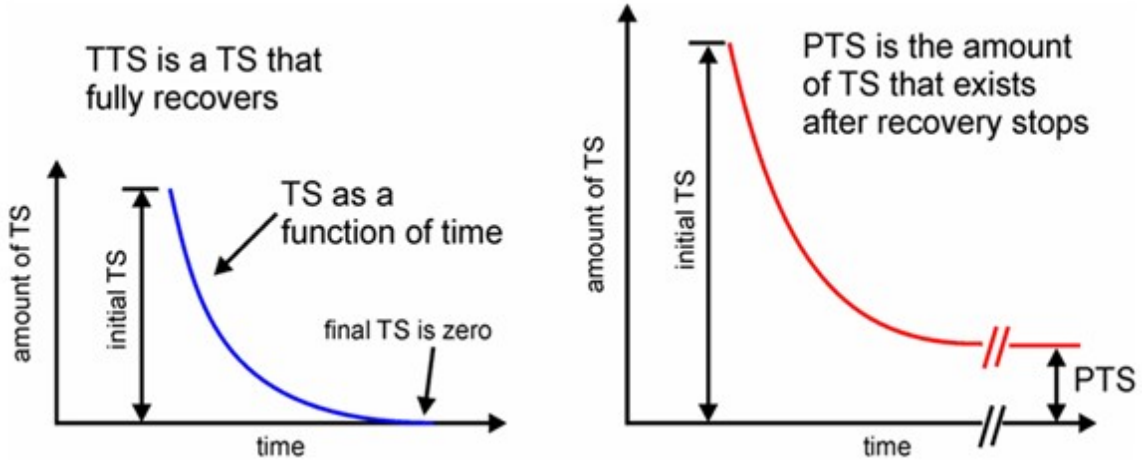

Figure 11. Two Hypothetical Threshold Shifts, Temporary and Permanent

Hearing loss, or auditory fatigue, in marine mammals has been studied by a number of investigators (Finneran et al. 2005; Finneran and Schlundt 2010; Finneran et al. 2007; Finneran et al. 2000b; Finneran et al. 2002; Lucke et al. 2009; Mooney et al. 2009a; Mooney et al. 2009b; Nachtigall et al. 2003; Schlundt et al. 2000a). The studies of marine mammal auditory fatigue

were all designed to determine relationships between TTS and exposure parameters such as level, duration, and frequency.

In these studies, hearing thresholds were measured in trained marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds indicated the amount of TTS. Species studied include the bottlenose dolphin, beluga, harbor porpoise, finless porpoise, California sea lion, harbor seal, and Northern elephant seal. Some of the more important data obtained from these studies are onset-TTS levels—exposure levels sufficient to cause a measurable amount of TTS, often defined as 6 dB of TTS (Schlundt et al. 2000a). These criteria for onset-TTS are very conservative, and it is not clear that this level of threshold shift would have a functional effect on the hearing of a marine mammal in the ocean.

The primary findings of the marine mammal TTS studies are:

- The growth and recovery of TTS shift are analogous to those in terrestrial mammals. This means that, as in terrestrial mammals, threshold shifts primarily depend on the amplitude, duration, frequency content, and temporal pattern of the sound exposure.
- The amount of TTS increases with exposure sound pressure level and the exposure duration.
- For continuous sounds, exposures of equal energy lead to approximately equal effects (Ward 1997). For intermittent sounds, less hearing loss occurs than from a continuous exposure with the same energy (some recovery will occur during the quiet period between exposures (Kastelein et al. 2014; Kryter et al. 1965; Ward 1997).
- Sound exposure level is correlated with the amount of TTS and is a good predictor for onset-TTS from single, continuous exposures with similar durations. This agrees with human TTS data presented by (Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b). However, for longer duration sounds beyond 16 to 32 seconds, the relationship between TTS and sound exposure level breaks down and duration becomes a more important contributor to TTS (Finneran and Schlundt 2010). Still, for a wide range of exposure durations, wound exposure level correlates reasonably well to TTS growth (Popov et al. 2014).
- The maximum TTS after tonal exposures occurs one-half to one octave above the exposure frequency (Finneran et al. 2007; Schlundt et al. 2000a). TTS from tonal exposures can thus extend over a large (greater than one octave) frequency range.
- For bottlenose dolphins, sounds with frequencies above 10 kHz are more hazardous than those at lower frequencies (i.e., lower sound exposure levels required to affect hearing) (Finneran and Schlundt 2010; Finneran and Schlundt 2013).
- The amount of observed TTS tends to decrease with increasing time following the exposure. The time required for complete recovery of hearing depends on the magnitude

of the initial shift; for relatively small shifts recovery may be complete in a few minutes, while large shifts (e.g., 40 dB) may require several days for recovery.

- TTS can accumulate across multiple intermittent exposures, but the resulting TTS will be less than the TTS from a single, continuous exposure with the same sound exposure level. This means that predictions based on total, cumulative sound exposure level will overestimate the amount of TTS from intermittent exposures.

Although there have been no marine mammal studies designed to measure PTS, the potential for PTS in marine mammals can be estimated based on known similarities between the inner ears of marine and terrestrial mammals. Experiments with marine mammals have revealed their similarities with terrestrial mammals with respect to features such as TTS, age-related hearing loss (called Presbycusis), ototoxic drug-induced hearing loss, masking, and frequency selectivity. Therefore, in the absence of marine mammal PTS data, onset-PTS shift exposure levels may be estimated by assuming some upper limit of TTS that equates the onset of PTS, then using TTS relationships from marine and terrestrial mammals to determine the exposure levels capable of producing this amount of TTS.

Hearing loss resulting from auditory fatigue could effectively reduce the distance over which animals can communicate, detect biologically relevant sounds such as predators, and echolocate (for odontocetes). The costs to marine mammals with TTS, or even some degree of PTS have not been studied; however, a relationship between the duration, magnitude, and frequency range of hearing loss could have consequences to biologically important activities (e.g., intraspecific communication, foraging, and predator detection) that affect survivability and reproduction. However, the classification of modeled effects from acoustic stressors, such as TTS and PTS, are performed in a manner as to conservatively overestimate the impacts of those effects. Acoustic stressors are binned and all stressors within each bin are modeled as the loudest source, necessarily overestimating impacts within each bin. Therefore, the temporary duration of TTS may be on the shorter end of the range and last briefly. Even longer duration TTS is only expected to last hours or at most a few days. Additionally, though the frequency range of TTS that marine mammals might sustain would overlap with some of the frequency ranges of their vocalization types, the frequency range of TTS from mid-frequency active sonar (the source from which TTS would most likely be sustained because the higher source level and slower attenuation make it more likely that an animal would be exposed to a higher received level) would not usually span the entire frequency range of one vocalization type, much less span all types of vocalizations or other critical auditory cues. If impaired, marine mammals would typically be aware of their impairment and would likely be able to implement behaviors to compensate. The brief amount of time marine mammals are expected to experience TTS is unlikely to significantly impair their ability to communicate, forage, or breed and will not have fitness level consequences at the individual or population level. Although PTS is a permanent shift in hearing, it is not the same as deafness and to our knowledge there are no published

studies on the long-term effects of PTS on marine mammal fitness. Conceivably, PTS could result in changes to individual's ability to communicate, breed, and forage but it is unclear if these impacts would significantly impact their fitness. For example, results from 2 years (2009 and 2010) of intensive monitoring by independent scientists and Navy observers in the Southern California and Hawaii Range Complexes have recorded an estimated 161,894 marine mammals with no evidence of distress or unusual behavior observed during Navy activities. This supports that TTS and PTS are unlikely to significantly impair their ability to communicate, forage, or breed and will not have fitness level consequences at the individual or population level.

6.3.8 Auditory Masking

Auditory masking occurs when a sound, or noise in general, limits the perception of another sound. As with hearing loss, auditory masking can effectively limit the distance over which a marine mammal can communicate, detect biologically relevant sounds, and echolocate (odontocetes). Unlike hearing loss, which likely results in a behavioral stress response, behavioral changes resulting from auditory masking may not be coupled with a stress response. Another important distinction between masking and hearing loss is that masking only occurs in the presence of the sound stimulus, whereas hearing loss can persist after the stimulus is gone.

Critical ratios, a measure of the relative ability of an animal to extract signals from noise, have been determined for pinnipeds (Southall et al. 2000; Southall et al. 2003) and bottlenose dolphins (Johnson 1967) and detections of signals under varying masking conditions have been determined for active echolocation and passive listening tasks in odontocetes (Au and Pawloski 1989a; Au and Pawloski 1989b; Branstetter 2013; Erbe 2000; Johnson 1971). These studies provide baseline information from which the probability of masking can be estimated.

Clark et al. (2009) developed a methodology for estimating masking effects on communication signals for low frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, their technique calculates that in Stellwagen Bank National Marine Sanctuary, when two commercial vessels pass through a North Atlantic right whale's (a baleen whale like blue, fin, sei, and humpback whales) optimal communication space (estimated as a sphere of water with a diameter of 20 km), that space is decreased by 84 percent. This methodology relies on empirical data on source levels of calls (which is unknown for many species), and requires many assumptions about ancient ambient noise conditions and simplifications of animal behavior. However, it is an important step in determining the impact of anthropogenic noise on animal communication. Subsequent research for the same species and location estimated that an average of 63 to 67 percent of North Atlantic right whale's communication space has been reduced by an increase in ambient noise levels, and that noise associated with transiting vessels is a major contributor to the increase in ambient noise (Hatch et al. 2012).

Vocal changes in response to anthropogenic noise can occur across sounds produced by marine mammals, such as whistling, echolocation click production, calling, and singing. Changes to vocal behavior and call structure may result from a need to compensate for an increase in background noise. In cetaceans, vocalization changes have been reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying. Vocalizations may also change in response to variation in the natural acoustic environment (e.g., from variation in sea surface motion)(Dunlop et al. 2014).

In the presence of low frequency active sonar, humpback whales have been observed to increase the length of their songs (Fristrup et al. 2003; Miller et al. 2000), possibly due to the overlap in frequencies between the whale song and the low frequency active sonar. North Atlantic right whales have been observed to increase the frequency and amplitude (intensity) (Parks 2009) of their calls while reducing the rate of calling in areas of increased anthropogenic noise (Parks et al. 2007a). In contrast, both sperm and pilot whales potentially ceased sound production during experimental sound exposure (Bowles et al. 1994), although it cannot be absolutely determined whether the inability to acoustically detect the animals was due to the cessation of sound production or the displacement of animals from the area.

Different vocal responses in marine mammals have been documented in the presence of seismic survey noise. An overall decrease in vocalization during active surveying has been noted in large marine mammal groups (Potter et al. 2007). In contrast, blue whale feeding and social calls increased when seismic exploration was underway (Di Lorio and Clark 2010), indicative of a potentially compensatory response to the increased noise level. (Melcon et al. 2012) recently documented that blue whales decreased the proportion of time spent producing certain types of calls when simulated mid-frequency sonar was present. At present it is not known if these changes in vocal behavior corresponded to changes in foraging or any other behaviors. Controlled exposure experiments in 2007 and 2008 in the Bahamas recorded responses of false killer whales, short-finned pilot whales, and melon-headed whales to simulated MFA sonar (Deruiter et al. 2013a). The responses to exposures between species were variable. After hearing each MFA signal, false killer whales were found to "increase their whistle production rate and made more-MFA-like whistles" (Deruiter et al. 2013a). In contrast, melon-headed whales had "minor transient silencing" after each MFA signal, while pilot whales had no apparent response. Consistent with the findings of other previous research (see, for example, (Southall et al. 2007b)), Deruiter et al. (2013a) found the responses were variable by species and with the context of the sound exposure.

Evidence suggests that at least some marine mammals have the ability to acoustically identify predators. For example, harbor seals that reside in the coastal waters off British Columbia are frequently targeted by certain groups of killer whales, but not others. The seals discriminate between the calls of threatening and non-threatening killer whales (Deecke et al. 2002), a

capability that should increase survivorship while reducing the energy required for attending to and responding to all killer whale calls. Auditory masking may prevent marine mammals from responding to the acoustic cues produced by their predators. The effects of auditory masking on the predator-prey relationship depends on the duration of the masking and the likelihood of encountering a predator during the time that predator cues are impeded.

Navy training activities in the Gulf of Alaska are not expected to result in extended periods of time where masking could occur. As stated above, masking only lasts the duration of the sound being emitted. The typical duty cycle with most tactical anti-submarine warfare is about once per minute with most active sonar pulses lasting no more than a few seconds (Navy 2013). This indicates biological relevant sounds for individuals in close proximity would only be masked for this short, intermittent, period of time. Moreover, a typical Navy vessel with hull mounted MFA sonar would travel over 300 meters between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Furthermore, events are geographically and temporally dispersed, and most events are limited to a few hours. Tactical sonar has a narrow frequency band (typically less than one-third octave). These factors reduce the likelihood of sources causing significant auditory masking in marine mammals.

6.3.9 Physiological Stress

Marine mammals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to diseases and naturally occurring toxins, lack of prey availability, social interactions with members of the same species, and interactions with predators all contribute to the stress a marine mammal experiences. In some cases, naturally occurring stressors can have profound impacts on marine mammals; for example, chronic stress, as observed in stranded animals with long-term debilitating conditions (e.g., disease), has been demonstrated to result in an increased size of the adrenal glands and an increase in the number of epinephrine-producing cells (Clark et al. 2006).

Anthropogenic activities have the potential to provide additional stressors above and beyond those that occur naturally. For example, various efforts have investigated the impact of vessels on marine mammals (both whale-watching and general vessel traffic noise) and demonstrated that impacts do occur (Bain 2002; Erbe 2002b; Noren et al. 2009; Pirotta et al. 2015; Williams and Ashe 2006; Williams and Noren 2009). In an analysis of energy costs to killer whales, Williams et al. (2009) suggested that whale-watching in the Johnstone Strait resulted in lost feeding opportunities due to vessel disturbance. Ayres et al. (2012) measured fecal hormones of southern resident killer whales in the Salish Sea to assess the lack of prey (salmon) and vessel traffic on species recovery. Ayres et al. (2012) suggested that the lack of prey overshadowed any population-level physiological impacts on southern resident killer whales from vessel traffic.

Marine mammals may exhibit a behavioral response or combinations of behavioral responses upon exposure to anthropogenic sounds. If a sound is detected by a marine mammal, a stress response (e.g., startle or annoyance) or a cueing response (based on a past stressful experience) can occur. Although preliminary because of the small numbers of samples collected, different types of sounds have been shown to produce variable stress responses in marine mammals. Belugas demonstrated no catecholamine (hormones released in situations of stress) response to the playback of oil drilling sounds (Thomas et al. 1990) but showed an increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al. 2004). A bottlenose dolphin exposed to the same seismic water gun signals did not demonstrate a catecholamine response, but did demonstrate an elevation in aldosterone, a hormone that may be a significant indicator of stress in odontocetes (St. Aubin et al. 2001; St. Aubin and Geraci 1989). Increases in heart rate were observed in bottlenose dolphins to which conspecific calls were played, although no increase in heart rate was observed when tank noise was played back (Miksis et al. 2001). Collectively, these results suggest a variable response that depends on the characteristics of the received signal and prior experience with the received signal.

Other types of stressors include the presence of vessels, fishery interactions, acts of pursuit and capture, the act of stranding, and pollution. In contrast to the limited amount of work performed on stress responses resulting from sound exposure, a considerably larger body of work exists on stress responses associated with pursuit, capture, handling and stranding. A study compared pathological changes in organs/tissues of odontocetes stranded on beaches or captured in nets over a 40-year period (Cowan and Curry 2008). The type of changes observed indicate multisystemic harm caused in part by an overload of catecholamines into the system, as well as a restriction in blood supply capable of causing tissue damage or tissue death. This response to a stressor or stressors is thought be mediated by the over-activation of the animal's normal physiological adaptations to diving or escape. Pursuit, capture and short-term holding of belugas have been observed to result in decreased thyroid hormones (St. Aubin and Geraci 1988) and increases in epinephrine (St. Aubin and Dierauf 2001). In dolphins, the trend is more complicated with the duration of the handling time potentially contributing to the magnitude of the stress response (Ortiz and Worthy 2000; St. Aubin 2002; St. Aubin et al. 1996). Male grey seals subjected to capture and short-term restraint showed an increase in cortisol levels accompanied by an increase in testosterone (Lidgard et al. 2008). This result may be indicative of a compensatory response that enables the seal to maintain reproduction capability in spite of stress. Elephant seals demonstrate an acute cortisol response to handling, but do not demonstrate a chronic response; on the contrary, adult females demonstrate a reduction in the adrenocortical response following repetitive chemical immobilization (Engelhard et al. 2002). Similarly, no correlation between cortisol levels and heart/respiration rate changes were seen in harbor porpoises during handling for satellite tagging (Eskesen et al. 2009). Taken together, these

studies illustrate the wide variations in the level of response that can occur when faced with these stressors.

Factors to consider when trying to predict a stress or cueing response include the mammal's life history stage and whether they are experienced with the stressor. Prior experience with a stressor may be of particular importance as repeated experience with a stressor may dull the stress response via acclimation (St. Aubin and Dierauf 2001).

The sound characteristics that correlate with specific stress responses in marine mammals are poorly understood. Therefore, in practice and for the purposes of this opinion, a stress response is assumed if a physical injury such as hearing loss or trauma is predicted; or if a significant behavioral response is predicted.

6.3.10 Behavioral Reactions

The response of a marine mammal to an anthropogenic sound will depend on the frequency, duration, temporal pattern and amplitude of the sound as well as the animal's prior experience with the sound and the context in which the sound is encountered (i.e., what the animal is doing at the time of the exposure). The distance from the sound source and whether it is perceived as approaching or moving away can affect the way an animal responds to a sound (Wartzok et al. 2003). For marine mammals, a review of responses to anthropogenic sound was first conducted by Richardson et al. (1995c). More recent reviews (e.g., Ellison et al. 2012; Nowacek et al. 2007; Southall et al. 2009; Southall et al. 2007a) address studies conducted since 1995 and focus on observations where the received sound level of the exposed marine mammal(s) was known or could be estimated.

Except for some vocalization changes that may be compensating for auditory masking, all behavioral reactions are assumed to occur due to a preceding stress or cueing response, however stress responses cannot be predicted directly due to a lack of scientific data (see preceding section). Responses can overlap; for example, an increased respiration rate is likely to be coupled with a flight response. Differential responses are expected among and within species since hearing ranges vary across species, the behavioral ecology of individual species is unlikely to completely overlap, and individuals of the same species may react differently to the same, or similar, stressor.

Southall et al. (2007a) synthesized data from many past behavioral studies and observations to determine the likelihood of behavioral reactions to specific sound levels exposures. While in general, the louder the sound source the more intense the behavioral response, it was clear that the proximity of a sound source and the animal's experience, motivation, and conditioning were also critical factors influencing the response (Southall et al. 2007b). After examining all of the available data, the authors felt that the derivation of thresholds for behavioral response based solely on exposure level was not supported because context of the animal at the time of sound

exposure was an important factor in estimating response. Nonetheless, in some conditions consistent avoidance reactions were noted at higher sound levels dependent on the marine mammal species or group, allowing conclusions to be drawn. Most low-frequency cetaceans (mysticetes) observed in studies usually avoided sound sources at levels of greater than or equal to 160 dB re 1 μPa. Published studies of mid-frequency cetaceans analyzed include sperm whales, belugas, bottlenose dolphins, and river dolphins. These groups showed no clear tendency, but for non-impulsive sounds, captive animals tolerated levels in excess of 170 dB re 1 μPa before showing behavioral reactions, such as avoidance, erratic swimming, and attacking the test apparatus. High-frequency cetaceans (observed from studies with harbor porpoises) exhibited changes in respiration and avoidance behavior at levels between 90 and 140 dB re 1 μPa, with profound avoidance behavior noted for levels exceeding this. Phocid seals showed avoidance reactions at or below 190 dB re 1 μPa, thus seals may actually receive levels adequate to produce TTS before avoiding the source. Recent studies with beaked whales have shown them to be particularly sensitive to noise, with animals during 3 playbacks of sound breaking off foraging dives at levels below 142 dB re 1 μPa, although acoustic monitoring during actual sonar exercises revealed some beaked whales continuing to forage at levels up to 157 dB re 1 μPa (Tyack et al. 2011b). Passive acoustic monitoring of beaked whales, classified as Blainville's beaked whales and Cross-seamount type beaked whales, at Pacific Missile Range Facility, Kauai, Hawaii showed statistically significant differences in dive rates, diel occurrence patterns, and spatial distribution of dives after the initiation of a training event. However, for the beaked whale dives that continued to occur during MFAS activity, differences from normal dive profiles and click rates were not detected with estimated receive levels up to 137 dB re 1 uPa while the animals were at depth during their dives (Manzano-Roth et al. 2013).

6.3.10.1.1 Behavioral Reactions of Marine Mammals to Impulsive Sound Sources

The following sections describe the behavioral reactions of marine mammals to impulsive sound sources such as underwater explosions.

6.3.10.1.2 Mysticetes

Baleen whales have shown a variety of responses to impulsive sound sources, including avoidance, reduced surface intervals, altered swimming behavior, and changes in vocalization rates (Richardson et al. 1995c; Southall et al. 2007d). While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al. 1995b), some whales avoided vessels by more than 20 km at received levels as low as 120 dB re 1 μPa root mean square. Additionally, Malme et al. (1988) observed clear changes in diving and respiration patterns in bowheads at ranges up to 73 km from seismic vessels, with received levels as low as 125 dB re 1 μPa.

Gray whales migrating along the U.S. west coast showed avoidance responses to seismic vessels at 164 dB re 1 μPa (10 percent of animals showed avoidance response), and at 190 dB re 1 μPa (90 percent of animals showed avoidance response), with similar results for whales in the Bering Sea (Malme et al. 1986; Malme et al. 1988). In contrast, noise from seismic surveys was not found to impact western North Pacific gray whale feeding behavior or exhalation rates off the coast of Russia (Gailey et al. 2007; Yazvenko et al. 2007).

Humpback whales showed avoidance behavior at ranges of 5 to 8 km from a seismic array during observational studies in western Australia (McCauley et al. 1998; Todd et al. 1996) found no clear short-term behavioral responses by foraging humpbacks to explosions associated with construction operations in Newfoundland, but did see a trend of increased rates of net entanglement and a shift to a higher incidence of net entanglement closer to the noise source.

Seismic pulses at average received levels of 131 dB re 1 μPa^2-s caused blue whales to increase call production (Di Lorio and Clark 2010). McDonald et al. (1995) tracked a blue whale with seafloor seismometers and reported that it stopped vocalizing and changed its travel direction at a range of 10 km from the seismic vessel (estimated received level 143 dB re 1 μPa peak-to-peak). Castellote et al. (2012a) found that vocalizing fin whales in the Mediterranean left the area where a seismic survey was being conducted and that their displacement persisted beyond the completion of the survey. These studies demonstrate that even low levels of noise received far from the noise source can induce behavioral responses.

6.3.10.1.3 Odontocetes

Madsen et al. (2006) and Miller et al. (2009) tagged and monitored eight sperm whales in the Gulf of Mexico exposed to seismic airgun surveys. Sound sources were approximately 2 to 7 nm away from the whales and, based on multipath propagation received levels, were as high as 162 dB SPL re 1 μPa with energy content greatest between 0.3 kHz to 3.0 kHz (Madsen et al. 2006). The whales showed no horizontal avoidance, although the whale that was approached most closely had an extended resting period and did not resume foraging until the airguns had ceased firing (Miller et al. 2009). The remaining whales continued to execute foraging dives throughout exposure, however swimming movements during foraging dives were 6 percent lower during exposure than control periods, suggesting subtle effects of noise on foraging behavior (Miller et al. 2009). Captive bottlenose dolphins sometimes vocalized after an exposure to impulsive sound from a seismic watergun (Finneran and Schlundt 2010; Thompson et al. 2013).

6.3.10.2 *Pinnipeds*

A review of behavioral reactions by pinnipeds to impulsive noise can be found in Richardson et al. (1995a) and Southall et al. (2007c). Blackwell et al. (2004) observed that ringed seals exhibited little or no reaction to drilling noise with mean underwater levels of 157 dB re 1 μPa

root mean square and in air levels of 112 dB re 20 µPa, suggesting the seals had habituated to the noise. In contrast, captive California sea lions avoided sounds from an impulsive source at levels of 165 to 170 dB re 1 µPa (Finneran et al. 2003b).

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

6.3.10.3 *Behavioral Reactions of Marine Mammals to Non-Impulsive Sources*

The following sections describe the behavioral reactions of marine mammals to non-impulsive sound sources such as sonar.

6.3.10.3.1 Mysticetes

Specific to Navy systems using low frequency sound, studies were undertaken pursuant to the Navy's Low Frequency Sound Scientific Research Program. These studies found only short-term responses to low frequency sound by mysticetes (fin, blue, and humpback whales) including changes in vocal activity and avoidance of the source vessel (Clark and Fristrup 2001; Croll et al. 2001b; Fristrup et al. 2003; Miller et al. 2000; Nowacek et al. 2007). Work by Risch et al. (2012) found that humpback whale vocalizations were reduced concurrently with pulses from the low frequency Ocean Acoustic Waveguide Remote Sensing (OAWRS) source located approximately 200 km away. However, (Gong et al. 2014), disputes these findings, suggesting that Risch et al. (2012) mistakes natural variations in humpback whale song occurrence for changes caused by OAWRS activity approximately 200 km away. Risch et al. (2014) responded to Gong et al. (2014) and highlighted the context-dependent nature of behavioral responses to acoustic stressors.

Baleen whales exposed to moderate low-frequency signals demonstrated no variation in foraging activity (Croll et al. 2001b). However, five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives, although the alarm signal was long in duration, lasting several minutes, and purposely designed to elicit a reaction from the animals as a prospective means to protect them from ship strikes (Nowacek et al. 2004a). Although the animal's received sound pressure level was similar in the latter two studies (133 to 150 dB re 1 µPa), the frequency, duration, and temporal pattern of signal presentation were different. Additionally, the right whales did not respond to playbacks of either right whale social sounds or

vessel noise, highlighting the importance of the sound characteristics, species differences, and individual sensitivity in producing a behavioral reaction.

Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were not found to affect dive times of humpback whales in Hawaiian waters (Frankel and Clark 2000) or to overtly affect elephant seal dives off California (Costa et al. 2003). However, they did produce subtle effects that varied in direction and degree among the individual seals, again illustrating the uncertain nature of behavioral effects and consequent difficulty in defining and predicting them.

Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007b; Stafford and Moore 2005a), preliminary results from the 2010 and 2011 field seasons of an ongoing behavioral response study in Southern California waters indicated that in some cases and at low received levels, tagged blue whales responded to mid-frequency sonar. However, those responses were mild and there was a quick return to their baseline activity (Southall et al. 2011a). Blue whales appeared to ignore sonar transmissions at received levels lower than approximately 150 dB and generally ignored received levels greater than these when they were engaged in feeding behavior (Southall et al. 2011a).

Goldbogen et al. (2013) used DTAGs to test the response of blue whales in the Southern California Bight to playbacks of simulated MFA sonar. Source levels of simulated sonar and control sounds (pseudo-random noise or PRN) in the 3.5 to 4.0 kHz range were ramped up in 3 dB increments from 160 to 210 dB re 1 μPa. Responses varied depending on the whales' prior behavioral state: surface feeding whales showed no response, while deep feeding whales and whales that were not feeding were affected. Responses among affected animals ranged from termination of deep foraging dives to prolonged mid-water dives. The reactions were context dependent, leading the authors to conclude a combination of received sound level and the behavioral state of the animal are likely to influence behavioral response. The authors note that whales responded even at low SPLs, suggesting that received level alone may not be a reliable predictor of behavior. Goldbogen et al. (2013) noted that behavioral responses observed were temporary and whales typically resumed normal behavior quickly after the cessation of sound exposure. Perhaps the most significant response documented in the study resulted from an experiment involving PRN rather than simulated sonar, which corresponded with a blue whale terminating a foraging bout. The more significant reaction to PRN may be indicative of habituation to mid-frequency sonar signals; the authors noted that the responses they documented were in a geographical region with a high level of naval activity and where mid-frequency sonar use is common.

Melcon et al. (2012) tested whether MFA sonar and other anthropogenic noises in the mid-frequency band affected the "D-calls" produced by blue whales in the Southern California Bight. The authors used passive acoustic monitoring data recorded with stationary High-frequency

Acoustic Recording Packages in the Southern California Bight. The likelihood of an animal calling decreased with the increased received level of mid-frequency sonar, beginning at a sound pressure level of approximately 110 to 120 dB re 1 µPa. Observations indicated that after sonar cessation, blue whales start producing D calls again. The authors concluded that blue whales heard and devoted attention to the sonar, despite its high frequency (relative to their putative hearing sensitivity) and its low received level. However, the authors noted that while D calls are typically associated with blue whale foraging behavior, they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior.

Martin et al. (2015a) used bottom mounted hydrophone arrays to estimate minke whale densities in the Pacific Missile Range Facility (PMRF) located off Kauai, Hawaii before, during, and after Navy training events involving active sonar. The study indicated minke whales decreased calling during the transmission of MFA sonar, but could not determine whether or not the whales left the area. The authors also suggested the presence of Naval surface ships during MFA transmission should be considered as a factor in the cessation of calling, rather than assuming the MFA sonar itself is the sole cause for the cessation of calling.

6.3.10.3.2 Odontocetes

From 2007 to present, behavioral response studies have been conducted through the collaboration of various research organizations in the Bahamas, Southern California, Mediterranean, Cape Hatteras, and Norwegian waters. These studies attempted to define and measure responses of beaked whales and other cetaceans to controlled exposures of sonar and other sounds to better understand their potential impacts. Results from the 2007 to 2008 study conducted near the Bahamas showed a change in diving behavior of an adult Blainville's beaked whale to playback of mid-frequency source and predator sounds (Boyd et al. 2008; Southall et al. 2009; Tyack et al. 2011c). Reaction to mid-frequency sounds included premature cessation of clicking, termination of a foraging dive, and a slower ascent rate to the surface. Preliminary results have been presented for the behavioral response study in Southern California waters (e.g., (Boyd et al. 2008; Southall et al. 2013; Southall et al. 2012a; Southall et al. 2011a).

For example, Stimpert et al. (2014) tagged a Baird's beaked whale and exposed it to simulated mid-frequency sonar. Some changes in the animal's dive behavior and locomotion were observed when received level reached 127 dB re 1µPa. Deruiter et al. (2013b) presented results from two Cuvier's beaked whales that were tagged and exposed to simulated MFA sonar during the 2010 and 2011 field seasons of the southern California behavioral response study. The 2011 whale was also incidentally exposed to MFA sonar from a distant naval exercise. Received levels from the MFA sonar signals from the controlled and incidental exposures were calculated as 84 to 144 and 78 to 106 dB re 1 µPa root mean square, respectively. Both whales showed responses to the controlled exposures, ranging from initial orientation changes to avoidance responses

characterized by energetic fluking and swimming away from the source. However, the authors did not detect similar responses to incidental exposure to distant naval sonar exercises at comparable received levels, indicating that context of the exposures (e.g., source proximity, controlled source ramp-up) may have been a significant factor. Cuvier's beaked whale responses suggested particular sensitivity to sound exposure as consistent with results for Blainville's beaked whale. Similarly, beaked whales exposed to sonar during British training activities stopped foraging (DSTL 2007).

Miller et al. (2011) reported on behavioral responses of pilot whales, killer whales, and sperm whales off Norway to Norwegian Navy mid-frequency sonar sources (a 3-year effort called the 3S experiments) (see also (Antunes et al. 2014; Kuningas et al. 2013; Kvadsheim et al. 2011; Miller et al. 2014; Miller et al. 2012; Sivle et al. 2012)). Reactions at different distances and received levels were variable, and types of responses observed included cessation of feeding, avoidance, changes in vocalizations, and changes in dive behavior. Some exposures elicited no observable reactions, and others resulted in brief or minor reactions, such as minor changes in vocalizations or locomotion. The experimental exposures occurred across different behavioral and environmental contexts, which may have played a role in the type of response observed, at least for killer whales (Miller et al. 2014). Some aspects of the experiment differ from typical Navy actions and may have exacerbated observed reactions; for example, animals were directly approached by the source vessel, researchers conducted multiple approaches toward the same animal groups over the course of each session, some exposures were conducted in bathymetrically restricted areas, and, in some cases, researchers "leapfrogged" the boat to repeatedly move ahead of the animals in order to repeatedly approach animals on their travel path. For example, separation of a killer whale calf from its mother occurred during the fifth vessel approach towards a killer whale group in a fjord. In contrast, Navy vessels avoid approaching marine mammals head-on, and vessels will maneuver to maintain a distance of at least 500 yd. (457 m) from observed animals. Furthermore, Navy mitigation measures would dictate powerdown of hull-mounted ASW sonars within 1,000 yd. (914m) of marine mammals and ultimately shutdown if an animal is within 200 yd. (183 m).

In the 2007 to 2008 Bahamas study, playback sounds of a potential predator—a killer whale—resulted in a similar but more pronounced reaction, which included longer inter-dive intervals and a sustained straight-line departure of more than 20 km from the area. The authors noted, however, that the magnified reaction to the predator sounds could represent a cumulative effect of exposure to the two sound types since killer whale playback began approximately 2 hours after mid-frequency source playback (Boyd et al. 2008; Tyack et al. 2011c). In contrast, preliminary analyses suggest that none of the pilot whales or false killer whales in the Bahamas showed an avoidance response to controlled exposure playbacks (Southall et al. 2009).

Through analysis of the behavioral response studies, a preliminary overarching effect of greater sensitivity to all anthropogenic exposures was seen in beaked whales compared to the other odontocetes studied (Southall et al. 2009). Therefore, recent studies have focused specifically on beaked whale responses to MFA sonar transmissions or controlled exposure playback of simulated sonar on various military ranges (Claridge and Durban 2009; DSTL 2007; McCarthy et al. 2011; Moretti 2009; Tyack et al. 2011c). In the Bahamas, Blainville's beaked whales located on the range will move off-range during MFA sonar use and return only after the sonar transmissions have stopped, sometimes taking several days to do so (Claridge and Durban 2009; McCarthy et al. 2011; Moretti 2009; Tyack et al. 2011c). Moretti et al. (2014) used recordings from seafloor mounted hydrophones at the Atlantic Undersea Test and Evaluation Center (AUTEC) to analyze the probability of Blainsville's beaked whale dives before, during, and after Navy sonar exercises. They developed an empirical risk function and predicted a 0.5 probability of disturbance at received levels of 150 dB.

Claridge (2013) used photo-recapture methods to estimate population abundance and demographics of Blainville's beaked whale (*Mesoplodon densirostris*) in the Bahamas at two sample locations; one within the bounds of the AUTEC where sonar training occurs and the second along the edge of Abaco Island approximately 170 km to the north. To investigate the potential effect of beaked whale exposure to MFA sonar, Claridge assumed that the two sample sites should have equal potential abundances and hypothesized that a lower abundance found at the AUTEC was due to either reduced prey availability at AUTEC or due to population level effects from the exposure to MFA sonar at AUTEC.

There are two major issues with this study. First, all of the re-sighted whales during the 5-year study at both sites were female. Claridge acknowledges that this can lead to a negative bias in the estimation of abundances. It has been shown in other cetacean species that females with calves may prefer "nursery" habitats or form nursery groups with other mother-calf pairs (e.g., (Claridge 2006; Scott et al. 1990; Weir et al. 2008)). It may be that the site at Abaco is a preferred site for females with calves, while the site at AUTEC is not, and therefore over the 5-year study period fewer females with calves were observed at AUTEC as these females went elsewhere in the area during the 3-year weaning period. In addition, Marques et al. (2009) estimated the Blainville's beaked whale population at AUTEC to be between 22.5 and 25.3 animals per 1,000 km^2. This density was estimated over 6 days using passive acoustic methods, which is a method Claridge identified as one that may be better for estimating beaked whale densities than visual methods. The results at AUTEC are also biased by reduced effort and a shorter overall study period that did not capture some of the emigration/immigration trends Claridge identified at Abaco. For these reasons among others, it is unclear whether there are significant differences in the abundances between the two sites.

Second, Claridge assumed that the two sites are identical and therefore should have equal potential abundances; Abaco is a "control" site with the difference being the use of sonar at AUTEC. Although the sample boundaries at each location were drawn to create samples "of comparable size," there are differences between the two sample area locations as follows: the Abaco site is along a leeward shore, AUTEC is windward; the Abaco sample area is a long narrow margin along a canyon wall, the rectangular AUTEC sample site is a portion of a deep and landlocked U-shaped trough. In addition to the physical differences, Claridge notes that it remains unclear whether or not variation in productivity between sites influenced what she refers to as the substantial differences in abundance. Claridge reports that a study investigating prey distributions at her sample locations was unable to sample prey at the beaked whale foraging depth. Claridge dismisses the possibility of differences in prey availability between the sites noting that there is no supporting evidence that prey availability differs between the two sites. As this study illustrates, the multiple and complex factors required by investigations of potential long-term cause and effect from actions at sea require a comprehensive assessment of all factors influencing potential trends in species abundances that are not likely attributable to a single cause and effect.

In the Caribbean, research on sperm whales in 1983 coincided with the U.S. intervention in Grenada. Sperm whales interrupted their activities by stopping echolocation and leaving the area. This response was assumed to be the result of underwater sounds originating from submarine mid to high-frequency sonar signals (Watkins et al. 1985b; Watkins and Schevill 1975b). The authors did not provide any sound levels associated with these observations, although they did note getting a similar reaction from banging on their boat hull. It was unclear if the sperm whales were reacting to the sonar signal itself or to a potentially new unknown sound as had been demonstrated on another occasion during which sperm whales in the Caribbean stopped vocalizing when presented with sounds from nearby acoustic pingers (Watkins and Schevill 1975b).

Researchers at the Navy's Marine Mammal Program facility in San Diego, California have conducted a series of controlled experiments on bottlenose dolphins and beluga whales to study TTS (Finneran 2010; Finneran 2011; Finneran et al. 2001; Finneran et al. 2003a; Finneran et al. 2010; Finneran and Schlundt 2004; Schlundt et al. 2000a). Ancillary to the TTS studies, scientists evaluated whether the marine mammals performed their trained tasks when prompted, during and after exposure to mid-frequency tones. Altered behavior during experimental trials usually involved refusal of animals to return to the site of the sound stimulus. This refusal included what appeared to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al. 2002; Schlundt et al. 2000a). Bottlenose dolphins exposed to 1-second tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 μPa root mean square, and beluga whales did so at received levels of 180 to 196 dB re 1 μPa and above. In some instances, animals exhibited

aggressive behavior toward the test apparatus (Ridgway et al. 1997; Schlundt et al. 2000a). While these studies were not designed to test avoidance behavior and animals were commonly reinforced with food, the controlled environment and ability to measure received levels provide insight on received levels at which animals will behaviorally responds to noise sources. More recently, a controlled-exposure study was conducted with Navy bottlenose dolphins at the Navy Marine Mammal Program facility specifically to study behavioral reactions to simulated mid-frequency sonar (Houser et al. 2013). Animals were trained to swim across a pen, touch a panel, and return to the starting location. During transit, a simulated mid-frequency sonar signal was played. Behavioral reactions were more likely with increasing received level and included increased respiration rates, fluke or pectoral fin slapping, and refusal to participate, among others. From these data, it was determined that bottlenose dolphins were more likely to respond to the initial trials, but habituated to the sound over the course of 10 trials except at the highest received levels. All dolphins responded at the highest received level (185 dB re 1 μPa).

Studies with captive harbor porpoises showed increased respiration rates upon introduction of acoustic alarms, such as those used on fishing nets to help deter marine mammals from becoming caught or entangled (Kastelein et al. 2001; Kastelein et al. 2006a) and emissions for underwater data transmission (Kastelein et al. 2005b). However, exposure of the same acoustic alarm to a striped dolphin under the same conditions did not elicit a response (Kastelein et al. 2006b), again highlighting the importance in understanding species differences in the tolerance of underwater noise (Southall et al. 2007b). Henderson et al. (2014) observed behavioral responses of delphinids to MFA sonar in the Southern California Bight from 2004 to 2008. The authors observed responses ranging from changes in behavioral state or direction of travel, to changes in vocalization activity. Behavioral responses were generally observed at received sound pressure levels ranging from 107 to 117 dB_{rms} re: 1 μPa. We are not reasonably certain that exposure to such sound pressure levels will elicit a substantive behavioral reaction and rise to the level of take per the ESA.

6.3.10.4 *Pinnipeds*

Different responses displayed by captive and wild phocid seals to sound judged to be 'unpleasant' have been reported; where captive seals habituated (did not avoid the sound), and wild seals showed avoidance behavior (Götz and Janik 2011). Captive seals received food (reinforcement) during sound playback, while wild seals were exposed opportunistically. These results indicate that motivational state (e.g., reinforcement via food acquisition) can be a factor in whether or not an animal habituates to novel or unpleasant sounds. Another study found that captive hooded seals reacted to 1 to 7 kHz sonar signals by moving to the areas of least sound pressure level, at levels between 160 and 170 dB re 1 μPa (Kvadsheim et al. 2010).

Captive studies with other pinnipeds have shown a reduction in dive times when presented with qualitatively 'unpleasant' sounds. These studies indicated that the subjective interpretation of the

pleasantness of a sound, minus the more commonly studied factors of received sound level and sounds associated with biological significance, can affect diving behavior (Götz and Janik 2011). More recently, a controlled-exposure studies was conducted with U.S. Navy California sea lions (*Zalophus californianus*) at the Navy Marine Mammal Program facility specifically to study behavioral reactions (Houser et al. 2013). Animals were trained to swim across a pen, touch a panel, and return to the starting location. During transit, a simulated mid-frequency sonar signal was played. Behavioral reactions included increased respiration rates, prolonged submergence, and refusal to participate, among others. Younger animals were more likely to respond than older animals, while some sea lions did not respond consistently at any level.

6.3.11 Repeated Exposures of Marine Mammals

Navy sonar systems are generally deployed from highly mobile vessels or in-water devices which do not directly target marine mammals. Further, the typical duty cycle with most tactical anti-submarine warfare is lower than used in the controlled exposure experiments described above, transmitting for a few seconds about once per minute (Navy 2013). For example, a typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. For sonar devices that are stationary (e.g. dipped sonar), due to the duty cycle, duration of active transmission in a specific location, and mitigation measures (e.g. avoidance of visible marine mammals), we would not expect repeated exposures. Additionally, it is worth noting here that contrary to other range complexes, the Navy does not train year round in the Gulf of Alaska, but instead Navy training activities in the Gulf of Alaska occur over a maximum of 21 days annually. The infrequent nature of Navy training in the Gulf of Alaska further reduces the likelihood of repeat exposures.

Some individuals may be exposed to multiple sound-producing activities over a season, year, or life stage. Repeated exposure to acoustic and other anthropogenic stimuli has been studied in several cases, especially as related to vessel traffic and whale watching. Common dolphins in New Zealand responded to dolphin-watching vessels by interrupting foraging and resting bouts, and took longer to resume behaviors in the presence of the vessel (Stockin 2008). Bejder et al. (2006a) studied responses of bottlenose dolphins to vessel approaches and found shorter lasting reactions in populations exposed to higher levels of vessel traffic. The authors indicated that lesser reactions in populations of dolphins regularly subjected to high levels of vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity.

Marine mammals exposed to high levels of human activities may leave the area, habituate to the activity, or tolerate the disturbance and remain in the area. Individual marine mammals that are

more tolerant may stay in a disturbed area, whereas individuals that are more sensitive may leave for areas with less human disturbance. Animals that remain throughout the disturbance may be unable to leave the area for a variety of physiological or environmental reasons. However, given the highly migratory, wide ranging life histories, and open ocean environments of the species considered in this opinion, we do not believe this will result from Navy training activities in the Gulf of Alaska TMAA. Longer-term displacement can lead to changes in abundance or distribution patterns of the species in the affected region if they do not become acclimated to the presence of the sound (Bejder et al. 2006b; Blackwell et al. 2004; Teilmann et al. 2006). Gray whales in Baja California abandoned a historical breeding lagoon in the mid-1960s due to an increase in dredging and commercial shipping operations. Whales did repopulate the lagoon after shipping activities had ceased for several years (Bryant et al. 1984). Over a shorter time scale, studies on the Atlantic Undersea Test and Evaluation Center instrumented range in the Bahamas have shown that some Blaineville's beaked whales may be resident during all or part of the year in the area, and that individuals may move off of the range for several days during and following a sonar event. However animals are thought to continue feeding at short distances (a few kilometers) from the range out of the louder sound fields (less than 157 dB re 1 μPa) (McCarthy et al. 2011; Tyack et al. 2011c). Mysticetes in the northeast tended to adjust to vessel traffic over a number of years, trending towards more neutral responses to passing vessels (Watkins 1986) indicating that some animals may habituate or otherwise learn to cope with high levels of human activity. Nevertheless, the long-term consequences of these habitat utilization changes are unknown, and likely vary depending on the species, geographic areas, and the degree of acoustic or other human disturbance.

Moore and Barlow (2013) have noted a decline in beaked whales in a broad area of the Pacific Ocean area out to 300 nm from the coast and extending from the Canadian-U.S. border to the tip of Baja Mexico. There are scientific caveats and limitations to the data used for that analysis, as well as oceanographic and species assemblage changes not thoroughly addressed in Moore and Barlow (2013), although the authors suggest Navy sonar as one possible explanation for the apparent decline in beaked whale numbers over that broad area. In the small portion of the Pacific coast overlapping the Navy's SOCAL Range Complex, long-term residency by individual Cuvier's beaked whales and documented higher densities of beaked whales provide indications that the proposed decline in numbers elsewhere along the Pacific coast is not apparent where the Navy has been intensively training with sonar and other systems for decades. While it is possible that a downward trend in beaked whales may have gone unnoticed at the range complex (due to a lack of survey precision) or that beaked whale densities may have been higher before the Navy began using sonar more than 60 years ago, there is no data available to suggest that beaked whale numbers have declined on the range where Navy sonar use has routinely occurred. As Moore and Barlow (2013) point out, it remains clear that the Navy range in SOCAL continues to support high densities of beaked whales. Furthermore, a large part of the U.S. West Coast action area

used by Moore and Barlow (2013) in their assessment of possible reasons for the decline include vast areas where the Navy does not conduct in-water training with sonar or explosives.

Establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013; Read et al. 2014). Assessing the effects of sounds, both individually and cumulatively, on marine species is difficult because responses depend on a variety of factors including age class, prior experience, behavioral state at the time of exposure, and indirect effects. Responses may be also be influenced by other non-sound related factors (Ellison et al. 2011; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014; Williams et al. 2014a). McGregor (2013) summarized sound impacts and described two types of possible effects based on the studies they reviewed: 1) an apparent effect of noise on communication, but with a link between demonstrated proximate cost and ultimate cost in survival or reproductive success being inferred rather than demonstrated, and 2) studies showing a decrease in population density or diversity in relation to noise, but with a relationship that is usually a correlation, so factors other than noise or its effect on communication might account for the relationship. Within the ocean environment, aggregate anthropogenic impacts have to be considered in context of natural variation and climate change (Boyd and Hutchins 2012). These contexts can include additive effects from two or more factors, multiplicity where response from two or more factors is greater than the sum of individual effects, synergism between factors and response, antagonism as a negative feedback between factors, acclimation as a short-term individual response, and adaptation as a long-term population change (Boyd and Hutchins 2012). To address aggregate impacts and responses from any changes due to processes such as habituation, tolerance, and sensitization, future experiments over an extended period of time still need further research (Bejder et al. 2009; Blickley et al. 2012; Read et al. 2014).

Some, including Goldbogen et al. (2013) and Stockin et al. (2008) have speculated that repeated interruptions of a marine mammal's normal activity could lead to fitness consequences and eventually, long-term implications for the population. For example, Goldbogen et al. (2013) suggested that if a blue whale responded to MFA sonar by temporarily interrupting feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since unconsumed prey would still be available in the environment following the cessation of acoustic exposure.

If sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks), it would be possible for individuals confined to a specific area to be exposed to acoustic stressors (e.g., MFA sonar) multiple times during a relatively short

time period. However, we do not expect this to occur as we would expect individuals to move and avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, even temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect equivalent foraging to be available in close proximity. Furthermore, Navy training in the Gulf of Alaska would only persist up to a maximum of one exercise period of 21 days between April and October on an annual basis. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

6.3.12 Stranding

When a marine mammal swims or floats (live or dead) onto shore and becomes "beached" or incapable of returning to sea, the event is termed a "stranding" (Geraci et al. 1999; Geraci and Lounsbury 2005). Animals outside of their "normal" habitat are also sometimes considered "stranded" even though they may not have beached themselves. The legal definition for a stranding within the United States is that: (A) a marine mammal is dead and is (i) on a beach or shore of the United States; or (ii) in waters under the jurisdiction of the United States (including any navigable waters); or (B) a marine mammal is alive and is (i) on a beach or shore of the United States and is unable to return to the water; (ii) on a beach or shore of the United States and, although able to return to the water, is apparently in need of medical attention; or (iii) in the waters under the jurisdiction of the United States (including any navigable waters), but is unable to return to its natural habitat under its own power or without assistance" (16 United States Code Section 1421h).

Marine mammals are subjected to a variety of natural and anthropogenic factors, acting alone or in combination, which may cause a marine mammal to strand on land or die at-sea (Geraci et al. 1999; Geraci and Lounsbury 2005). Even for the fractions of more thoroughly investigated strandings involving post-stranding data collection and necropsies, the cause (or causes) for the majority of strandings remain undetermined. Natural factors related to strandings include the availability of food, predation, disease, parasitism, climatic influences, and aging (Bradshaw et al. 2006; Culik 2004; Geraci et al. 1999; Geraci and Lounsbury 2005; Hoelzel 2003; NRC 2003b; Perrin and Geraci 2002; Walker et al. 2005). Anthropogenic factors include pollution (Anonmyous 2010; Elfes et al. 2010; Hall et al. 2006a; Hall et al. 2006b; Jepson et al. 2005; Tabuchi et al. 2006), vessel strike (Berman-Kowalewski et al. 2010; De Stephanis and Urquiola 2006; Geraci and Lounsbury 2005; Jensen and Silber 2003; Laist et al. 2001), fisheries

interactions (Look 2011; Read et al. 2006), entanglement (Baird and Gorgone 2005; Johnson et al. 2005; Saez et al. 2013), and noise (Cox et al. 2006; NRC 2003b; Richardson et al. 1995d).

Along the coasts of the continental United States and Alaska between 2001 and 2009, there were approximately 1,400 cetacean strandings and 4,300 pinniped strandings (5,700 total) per year (NMFS 2011d; NMFS 2011e; NMFS 2011f). More recently (beginning in May 2015) and in close proximity to the GOA TMAA, elevated large whale mortalities have occurred in the western portion of the Gulf of Alaska, particularly Kodiak Island. As of September 12, 2016, at least 12 fin, 39 humpback, 2 gray, and 8 unidentified cetaceans have stranded in the area. As of the signing of this opinion, no definitive cause has been determined for this event. Lefebvre et al. (2016) indicate that toxic algal blooms are suspected as a causative agent for Alaska wide marine mammal strandings. Several "mass stranding" events—strandings that involve two or more individuals of the same species (excluding a single cow-calf pair)—that have occurred over the past two decades have been associated with naval operations, seismic surveys, and other anthropogenic activities that introduced sound into the marine environment. An in-depth discussion of strandings is presented in DoN (2013b).

Sonar use during exercises involving Navy (most often in association with other nations' defense forces) has been identified as a contributing cause or factor in five specific mass stranding events: Greece in 1996; the Bahamas in March 2000; Madeira Island, Portugal in 2000; the Canary Islands in 2002, and Spain in 2006 (MMC 2006). While not considered an official stranding event pursuant to U.S. law, these five mass stranding events have resulted in approximately 40 known stranding deaths among cetaceans, consisting mostly of beaked whales, with a potential link to sonar (ICES 2005a; ICES 2005b; ICES 2005c). The U.S.-Navy-funded research involving Behavioral Response Studies in SOCAL and the Bahamas discussed previously were motivated by the desire to understand any links between the use of mid-frequency sonar and cetacean behavioral responses, including the potential for strandings. Although these events have served to focus attention on the issue of impacts resulting from the use of sonar, as Ketten (2012) recently pointed out, "ironically, to date, there has been no demonstrable evidence of acute, traumatic, disruptive, or profound auditory damage in any marine mammal as the result [of] anthropogenic noise exposures, including sonar."

In these previous circumstances, exposure to non-impulsive acoustic energy has been considered a potential indirect cause of the death of marine mammals (Cox et al. 2006). One hypothesis regarding a potential cause of the strandings is tissue damage resulting from "gas and fat embolic syndrome" (Fernandez et al. 2005a; Jepson et al. 2003; Jepson et al. 2005). Models of nitrogen saturation in diving marine mammals have been used to suggest that altered dive behavior might result in the accumulation of nitrogen gas such that the potential for nitrogen bubble formation is increased (Houser 2010; Houser et al. 2001b; Zimmer and Tyack 2007). If so, this mechanism might explain the findings of gas and bubble emboli in stranded beaked whales. It is also

possible that stranding is a behavioral response to a sound under certain contextual conditions and that the subsequently observed physiological effects (e.g., overheating, decomposition, or internal hemorrhaging from being on shore) were the result of the stranding rather than direct physical impact from exposure to sonar (Cox et al. 2006).

In May 2003 there was an incident involving the use of mid-frequency sonar by the USS SHOUP, which was portrayed in some media reports at the time as having potentially causing harbor porpoise strandings in the region. On May 5, 2003, in the area of Admiralty Inlet, the USS SHOUP began the use of mid-frequency sonar as part of a training event, which continued until later that afternoon and ended as the USS SHOUP transited Haro Strait heading north. Between May 2 and June 2, 2003, approximately 16 strandings involving 15 harbor porpoises (*Phocoena phocoena*) and 1 Dall's porpoise (*Phocoenoides dalli*) had been reported to the Northwest Marine Mammal Stranding Network, and allegations were made that these strandings had been caused by the USS SHOUP's use of sonar. A comprehensive review of all strandings and the events involving USS SHOUP on May 5, 2003, were subsequently presented in a report by U.S. Department of Navy (DoN 2004).

Additionally NMFS undertook a series of necropsy analyses on the stranded animals to determine the cause of the strandings (NMFS 2005a; Norman et al. 2004a). Necropsies were performed on 10 of the porpoises and two heads were selected for computed tomographic imaging (Norman et al. 2004a).

None of the 11 harbor porpoises demonstrated signs of acoustic trauma. A putative cause of death was determined for five of the porpoises based only on the necropsy results; two animals had blunt trauma injuries and three animals had indication of disease processes. A cause of death could not be determined in the remaining animals, which is consistent with the expected percentage of marine mammal necropsies conducted within the northwest region. It is important to note, that these determinations were based only on the evidence from the necropsy to avoid bias with regard to determinations of the potential presence or absence of acoustic trauma. For example, the necropsy investigators had no knowledge of other potential external causal factors, such as Specimen 33NWR05005 having been found tangled in a fishing net, which may have otherwise assisted in their determination regarding the likely cause of death for that animal. Additionally, seven of the porpoises collected and analyzed died prior to SHOUP departing to sea on May 5, 2003. Of these seven, one, discovered on May 5, 2003, was in a state of moderate decomposition, indicating it died before May 5; the cause of death was determined, most likely, to be Salmonella septicemia. Another porpoise, discovered at Port Angeles on May 6, 2003, was in a state of moderate decomposition, indicating that this porpoise also died prior to May 5. One stranded harbor porpoise discovered fresh on May 6 is the only animal that could potentially be linked to the USS SHOUP's May 5 active sonar use. Necropsy results for this porpoise found no evidence of acoustic trauma. The remaining eight strandings were discovered 1 to 3 weeks after

the USS SHOUP's May 5 use of sonar. Two of the eight porpoises died from blunt trauma injury and a third suffered from parasitic infestation, which possibly contributed to its death (Norman et al. 2004a). For the remaining five porpoises, NMFS was unable to identify the causes of death.

NMFS concluded from a retrospective analysis of stranding events that the number of harbor porpoise stranding events in the approximate month surrounding the USS SHOUP's use of sonar was higher than expected based on annual strandings of harbor porpoises (Norman et al. 2004a). This conclusion in the NMFS report also conflicts with data from The Whale Museum, which has documented and responded to harbor porpoise strandings since 1980 (Osborne 2003). According to The Whale Museum, the number of strandings as of May 15, 2003, was consistent with what was expected based on historical stranding records and was less than that occurring in certain years. For example, since 1992, the San Juan Stranding Network has documented an average of 5.8 porpoise strandings per year. In 1997, there were 12 strandings in the San Juan Islands, with more than 30 strandings throughout the general Puget Sound area. In reporting their findings, NMFS acknowledged that the intense level of media attention to the 2003 strandings likely resulted in increased reporting effort by the public over that which is normally observed (Norman et al. 2004a). NMFS also noted in its report that the "sample size is too small and biased to infer a specific relationship with respect to sonar usage and subsequent strandings." It was also clear that in 2003, the number of strandings from May to June was also higher for the outer coast, indicating a much wider phenomena than use of sonar by USS SHOUP in Puget Sound for one day in May. It was later determined by NMFS that the number of harbor porpoise strandings in the northwest had been increased beginning in 2003 and through 2006. On November 3, 2006, an Unusual Mortality Event in the Pacific Northwest was declared by NMFS (see (DoN 2013b), Cetacean Stranding Report for more detail on this Unusual Mortality Event).

The speculative association of the harbor porpoise strandings to the use of sonar by the USS SHOUP was inconsistent with prior stranding events linked to the use of mid-frequency sonar. Specifically, in prior events strandings occurred shortly after the use of sonar (less than 36 hours) and stranded individuals were spatially co-located. Although MFA sonar was used by the USS SHOUP, the distribution of harbor porpoise strandings by location and with respect to time surrounding the event do not support the suggestion that MFA sonar was a cause of harbor porpoise strandings. Rather, a lack of evidence of any acoustic trauma within the harbor porpoises, and the identification of probable causes of stranding or death in several animals, supports the conclusion that harbor porpoise strandings in 2003 in the Pacific Northwest were unrelated to the sonar activities by the USS SHOUP.

As the ICES (2005c) noted, taken in context of marine mammal populations in general, sonar is not a major threat, or significant portion of the overall ocean noise budget. This has also been demonstrated by monitoring in areas where the Navy operates (Bassett et al. 2010; Baumann-Pickering et al. 2010; Hildebrand et al. 2011; McDonald et al. 2006a; Tyack et al. 2011a).

Regardless of the direct cause, the Navy considers potential sonar related strandings important and continues to fund research and work with scientists to better understand circumstances that may result in strandings. During a Navy training event on March 4, 2011, at the Silver Strand Training Complex in San Diego, California, four long-beaked common dolphins were killed by the detonation of an underwater explosive (Danil and St. Leger 2011). This area has been used for underwater demolitions training for at least 3 decades without incident. During this underwater detonation training event, a pod of 100 to 150 long-beaked common dolphins were moving towards the explosive's 700-yd. (640 m) exclusion zone monitored by a personnel in a safety boat and participants in a dive boat. Within the exclusion zone, approximately 5 minutes remained on a time-delayed firing device connected to a single 8.76 lb (3.8 kg) explosive charge set at a depth of 48 ft. (14.6 m), approximately 0.5 to 0.75 nm from shore. Although the dive boat was placed between the pod and the explosive in an effort to guide the dolphins away from the area, that effort was unsuccessful and three long-beaked common dolphins died as a result of being in proximity to the explosion. In addition, to the three dolphins found dead on March 4th at the event site, the remains of a fourth dolphin were discovered on March 7th (3 days later and approximately 42 mi. (68 km) from the location where the training event occurred), which was assessed as being related to this event (Danil and St. Leger 2011). Details such as the dolphins' depth and distance from the explosive at the time of the detonation could not be estimated from the 250-yd (229 m) standoff point of the observers in the dive boat or the safety boat.

These dolphin mortalities are the only known occurrence of a Navy training event involving impulsive energy (underwater detonation) that has resulted in injury to a marine mammal. Despite this being a rare occurrence, the Navy has reviewed training requirements, safety procedures, and potential mitigation measures and, along with NMFS, is determining appropriate changes to reduce the potential for this to occur in the future.

In comparison to potential strandings or injury resulting from events associated with Navy activities, marine mammal strandings and injury from commercial vessel ship strike (Berman-Kowalewski et al. 2010; Silber et al. 2010), impacts from urban pollution (Hooker et al. 2007; O'Shea and Brownell Jr. 1994), and annual fishery-related entanglement, bycatch, injury, and mortality (Baird and Gorgone 2005; Forney and Kobayashi 2007; Saez et al. 2013), have been estimated worldwide to be orders of magnitude greater (hundreds of thousands of animals versus tens of animals; (Culik 2004; ICES 2005c; Read et al. 2006)) than the few potential injurious impacts that could be possible as a result of Navy activities. This does not negate the potential influence of mortality or additional stress to small, regionalized sub-populations which may be at greater risk from human related mortalities (fishing, vessel strike, sound) than populations with larger oceanic level distributions, but overall the Navy's impact in the oceans and inland water areas where training occurs is small by comparison to other human activities. Nonetheless, the focus of our analysis is to determine, considering the status of the resources, the environmental

baseline and effects from future non-federal activities, whether the Navy's activities are likely to jeopardize listed species or are likely to destroy or adversely modify critical habitat.

6.3.13 Long-term Consequences to the Individual and the Population

Long-term consequences to a population are determined by examining changes in the population growth rate. Individual effects that could lead to a reduction in the population growth rate include mortality or injury (that removes animals from the reproductive pool), hearing loss (which depending on severity could impact navigation, foraging, predator avoidance, or communication), chronic stress (which could make individuals more susceptible to disease), long-term displacement of individuals (especially from preferred foraging or mating grounds), and disruption of social bonds (due to masking of conspecific signals or displacement). However, the long-term consequences of any of these effects are difficult to predict because individual experience and time can create complex contingencies, especially for intelligent, long-lived animals like marine mammals. While a lost reproductive opportunity could be a measureable cost to the individual, the outcome for the animal, and ultimately the population, can range from insignificant to significant. Any number of factors, such as maternal inexperience, years of poor food supply, or predator pressure, could produce a cost of a lost reproductive opportunity, but these events may be "made up" during the life of a normal healthy individual. The same holds true for exposure to human-generated noise sources. These biological realities must be taken into consideration when assessing risk, uncertainties about that risk, and the feasibility of preventing or recouping such risks. The long-term consequence of relatively trivial events like short-term masking of a conspecific's social sounds, or a single lost feeding opportunity, can be exaggerated beyond its actual importance by focusing on the single event rather than the totality of the individual and its lifetime parameters of growth, reproduction and survival.

Population models are well known from many fields in biology including fisheries and wildlife management. These models accept inputs for the population size and changes in vital rates of the population such as the mean values for survival age, lifetime reproductive success, and recruitment of new individuals into the population. The time-scale of the inputs in a population model for long-lived animals such as marine mammals is on the order of seasons, years, or life stages (e.g., neonate, juvenile, reproductive adult), and are often concerned only with the success of individuals from one time period or stage to the next. Unfortunately, information is not available to accurately assess the impact of acoustic and explosive exposure on individual marine mammal vitals rates. Further, for assessing the impact of acoustic and explosive impacts to marine mammal populations, many of the inputs required by population models are not known.

Recently, efforts have been made to understand the linkage between a stressor, such as anthropogenic sound, and its immediate behavioral or physiological consequences for the individual, and then the subsequent effects on that individual's vital rates (growth, survival and reproduction), and the consequences, in turn, for the population. In 2005, a panel convened by

the National Research Council of the United States National Academy of Sciences published a report on 'Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects'. The panel developed what it called "a conceptual model" that outlined how marine mammals might be affected by anthropogenic noise and how population level effects could be inferred on the basis of observed behavioral changes. They called this model 'Population Consequences of Acoustic Disturbance' (PCAD). In 2009 the US Office of Naval Research (ONR) set up a working group to transform this framework into a formal mathematical structure and determine how that structure could be parameterized using data from a number of case studies. The ONR working group extended the PCAD framework so that it could be used to consider other forms of disturbance and to address the impact of disturbance on physiology as well as behavior. Their current version of that framework is now known as PCoD (Population Consequences of Disturbance) (New et al. 2014). It is important to note that PCoD is ongoing and is an exploratory project to determine how an interim PCoD approach might inform analysis. It is not intended to provide an actual assessment of the population-level consequences of disturbance for beaked whale populations on Navy ranges.

New et al. (2013) developed a mathematical model simulating a functional link between feeding energetics and a species' requirements for survival and reproductions for 21 species of beaked whale. The authors report "reasonable confidence" in their model, although approximately 29 percent (6 of 21 beaked whale species modeled) failed to survive or reproduce, which the authors attribute to possible inaccuracies in the underlying parameters. Based on the model simulation, New et al. (2013) determined that if habitat quality and "accessible energy" (derived from the availability of either plentiful prey or prey with high energy content) are both high, then survival rates are high as well. If these variables are low, then adults may survive but calves will not. For the 29 percent of beaked whale species for which the model failed (within the assumed range of current inputs), the assumption was a 2-year calving period (or inter-calf interval), however, for species with longer gestation periods (such as the 17-month gestation period of Baird's beaked whale (*Berardius bairdii*), this inter-calf interval may be too short. For Blainville's beaked whale, (Claridge 2013) has shown that calf age at separation is at least 3 years, and that the inter-calf interval at Abaco in the Bahamas may be 4 years. New et al. (2013) acknowledge that an assumed 2-year calving period in the modeling may not be long enough to build up the energetic resources necessary for mother and calf survival.

As another critical model assumption, prey preferences were modeled based on stomach content analyses of stranded animals, which the authors acknowledge are traditionally poor estimates of the diets of healthy animals, as stranded animals are often sick prior to stranding. Stomach content remnants of prey species do not digest equally, as only the hard parts of some prey types remain (e.g., fish otoliths, beaks of cephalopods) and thus often provide an incomplete picture of diet. Given these unknowns and the failure of the simulation to work for 29 percent of beaked whale species, the modeled survival rates of all beaked whales, particularly those modeled with

prey having low energy content, may be better than simulated if higher-energy prey makes up a larger part of the diet than assumed by the model simulations.

In short, for the model output New et al. (2013) created to correctly represent links between the species and their environment, that model must identify all the critical and relevant ecological parameters as input variables, provide the correct values for those parameters, and then the model must appropriately integrate modeling functions to duplicate the complex relationships the model intends to represent. If an assumption (model input) such as calving period or prey preferences is incorrect (and there is presently no way to know), then the model would not be representing what may actually be occurring. New et al. (2013) report that their simulations suggest that adults will survive but not reproduce if anthropogenic disturbances result in being displaced to areas of "impaired foraging." Underlying this suggestion is the additional unstated assumption that habitat capable of sustaining a beaked whale is limited in proximity to where any disturbance has occurred and there are no data to indicate that is a valid assumption.

While the New et al. (2013) model provides a test case for future research, this pilot study has very little of the critical data necessary to form any conclusions applicable to current management decisions. The authors note the need for more data on prey species and reproductive parameters including gestation and lactation duration, as the model results are particularly affected by these assumptions. Therefore, any suggestion of biological sensitivity to the simulation's input parameters is uncertain.

New et al. (2014) used a simulation model to assess how behavioral disruptions (e.g., significant disruption of foraging behavior) may affect southern elephant seal health, offspring survival, individual fitness, and population growth rate. They suggested their model can determine the population consequences of disturbance from short-term changes in individual animals. Their model assumed that disturbance affected behavior by reducing the number of drift dives in which the animals were feeding and increasing the time they spent in transit. For example, they suggested a disturbance lasting 50 percent of an average annual foraging trip would reduce pup survival by 0.4 percent. If this level of disturbance continued over 30 years and the population did not adapt, the authors found that the population size would decrease by approximately 10 percent.

The findings of New et al. (2014) are not applicable to the temporary behavioral disruptions that may be caused by Navy training activities for a number of reasons. First, the model assumed that individuals would be unable to compensate for lost foraging opportunities. As described previously, available empirical data does not confirm this would be the case. For example, elephant seals are unlikely to be affected by short-term variations in prey availability because they take long foraging trips, allowing for some margin of error in prey availability (Costa 1993), as cited in New et al. (2014). We expect the species considered in this opinion to be similarly unaffected. We have no information to suggest animals eliciting a behavioral response (e.g.,

temporary disruption of feeding) to Navy training activities would be unable to compensate for this temporary disruption in feeding activity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Additionally, the behavioral disruption of ESA-listed species reasonably expected to occur due to Navy training activities will not have as long of a duration as those considered in the New et al. (2014) study. As discussed in Section 6.3.11, no individual animals will be exposed to Navy training activities for a long enough duration to disrupt 50 percent of its annual feeding opportunities over multiple years. New et al. (2014) suggest it would be unlikely even for episodic environmental change, such as El Niño events, to affect the probability of population persistence. As suggested by the authors, the New et al. (2014) model may be more applicable to the consideration of potential long-term behavioral disruptions (e.g., those that may result from climate change).

Until an appropriate quantitative model is developed and until all relevant empirical data is collected to support such a model for the species considered in this opinion, the best assessment of long-term consequences from training activities will be to monitor the populations over time within a given Navy range complex. A Navy workshop on Marine Mammals and Sound (Fitch et al. 2011) indicated a critical need for baseline biological data on marine mammal abundance, distribution, habitat, and behavior over sufficient time and space to evaluate impacts from human-generated activities on long-term population survival. The Navy has developed monitoring plans for protected marine mammals and sea turtles occurring on Navy ranges with the goal of assessing the impacts of training activities on marine species and the effectiveness of the Navy's current mitigation practices. Monitoring has resulted in data on occurrence, exposure, and behavioral response in the Gulf of Alaska TMAA. All monitoring reported are available to the public and posted at www.navymarinespeciesmonitoring.us/.

6.3.14 Criteria for Predicting Acoustic and Explosive Impacts – Marine Mammals

When Navy activities introduce sound or explosive energy into the marine environment, an analysis of potential impacts to marine mammals is conducted. To do this, information about the numerical sound and energy levels that are likely to elicit certain types of physiological and behavioral reactions is needed. The following contains information on the criteria, thresholds, and methodology for quantifying effects from acoustic and explosive sources, which were jointly developed by the Navy and NMFS. While we provide the criteria for all functional hearing groups and taxa, we did not assess effects to high-frequency cetaceans, phocid seals, or sea otters because they were not relevant to this consultation.

As discussed further in section 3.1.4 of this opinion, NMFS recently released its Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing. This new Guidance established new thresholds and associated weighting functions for predicting auditory injury, or permanent threshold shift and temporary threshold shift. The new criteria

were not available for the Navy's acoustic effects modeling used to calculate distances to harassment thresholds and resulting take estimates for this consultation. Therefore, the Navy did not directly use the new auditory weighting functions and PTS/TTS criteria in its acoustic modeling or the GOA FSEIS/OEIS. However, the Navy was able to use the Phase II model to reprocess anticipated explosive ranges to effects for PTS based on the criteria presented in the new Guidance to assess if the new criteria could result in any additional species-specific injury exposures. The conclusion from that analysis was that the new impulsive criteria would not change previous species-specific quantities of impulsive PTS, TTS, or behavioral exposures for any ESA-listed species. The sonar exposures were not remodeled because a qualitative assessment of the new Guidance and the activities showed that it was not necessary in order to support the analysis, in addition to being impractical.

The results of prior Navy modeling described in this opinion represent the best available estimate of the number and type of take of ESA-listed marine mammals that may result from the Navy's use of acoustic sources in the action area. Modeling that incorporated the updated acoustic thresholds could result in minor changes to the enumerations of take estimates. However, as described above in section 3.1.4, use of the new acoustic thresholds would not alter our assessment of the likely responses of affected ESA-listed species to acoustic sources employed by Navy in the action area, or the likely fitness consequences of those responses.

6.3.14.1 *Frequency Weighting*

Frequency-weighting functions are used to adjust the received sound level based on the sensitivity of the animal to the frequency of the sound. The weighting functions de-emphasize sound exposures at frequencies to which marine mammals are not particularly sensitive. This effectively makes the acoustic thresholds frequency-dependent, which means they are applicable over a wide range of frequencies and therefore applicable for a wide range of sound sources. Frequency-weighting functions, called "M-weighting" functions, were proposed by Southall et al. (2007b) to account for the frequency bandwidth of hearing in marine mammals. These M-weighting functions were derived for each marine mammal hearing group based on an algorithm using the range of frequencies that are within 80 kHz of an animal or group's best hearing. The Southall et al. (2007b) M-weighting functions are nearly flat between the lower and upper cutoff frequencies, and thus were believed to represent a conservative approach to assessing the effects of noise (Figure 12). For the purposes of this analysis, we refer to these as Type I auditory weighting functions. Otariid seal thresholds and weighting functions were applied to sea otter as described in Finneran and Jenkins (2012).

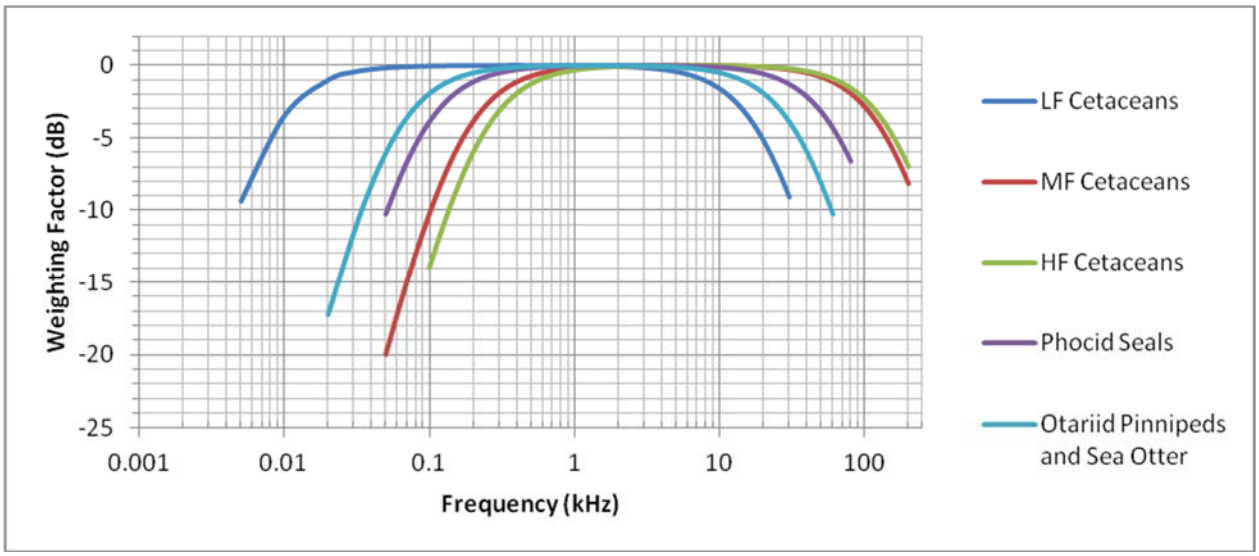

Figure 12. Type I Auditory Weighting Functions Modified from the Southall et al. (2007) M-Weighting Functions

Finneran and Jenkins (2012) considered data since Southall et al. (2007b) and determined two published studies suggested adjustments to the weighting functions were appropriate. The first experiment measured TTS in a bottlenose dolphin after exposure to pure tones with frequencies from 3 to 28 kHz (Finneran and Schlundt 2010). These data were used to derive onset-TTS values as a function of exposure frequency, and demonstrate that the use of a single numeric threshold for onset-TTS, regardless of frequency, is not correct. The second experiment examined how subjects perceived the loudness of sounds at different frequencies to derive equal loudness contours (Finneran and Schlundt 2011a). These data are important because human auditory weighting functions are based on equal loudness contours. The dolphin equal loudness contours provide a means to generate auditory weighting functions in a manner directly analogous to the approach used to develop safe exposure guidelines for people working in noisy environments (NIOSH 1998).

Taken together, the recent higher-frequency TTS data and equal loudness contours provide the underlying data necessary to develop new weighting functions, referred to as Type II auditory weighting functions. Type II auditory weighting functions improve accuracy and avoid underestimating the impacts to animals at higher frequencies as shown in Figure 13. To generate the new Type II weighting functions, Finneran and Schlundt (2011b) substituted lower and upper frequency values which differ from the values used by Southall et al. (2007b).

The new weighting curve predicts appreciably higher (almost 20 dB) susceptibility for frequencies above 3 kHz for bottlenose dolphins, a mid-frequency cetacean. Since data below 3 kHz are not available, the original weighting functions from Southall et al. (2007b) were substituted below this frequency. Low- and high-frequency cetacean weighting functions were

extrapolated from the dolphin data as well, because of the suspected similarities of greatest susceptibility at best frequencies of hearing. Similar Type II weighting curves were not developed for pinnipeds since their hearing is markedly different from cetaceans, and because they do not hear as well at higher frequencies. Their weighting curves do not require the same adjustment (see Finneran and Jenkins (2012) for additional details).

The Type II auditory cetacean weighting functions (Figure 13) are applied to the received sound level before comparing it to the appropriate sound exposure level thresholds for TTS or PTS, or the impulse behavioral response threshold. Note that for pinnipeds and sea otters, the Southall et al. (2007b) weighting functions (Figure 13) are used in lieu of any new weighting functions. For some criteria, received levels are not weighted before being compared to the thresholds to predict effects. These include the peak pressure criteria for predicting TTS and PTS from underwater explosions, the acoustic impulse metrics used to predict onset-mortality and slight lung injury, and the thresholds used to predict behavioral responses from harbor porpoises and beaked whales from sonar and other active acoustic sources.

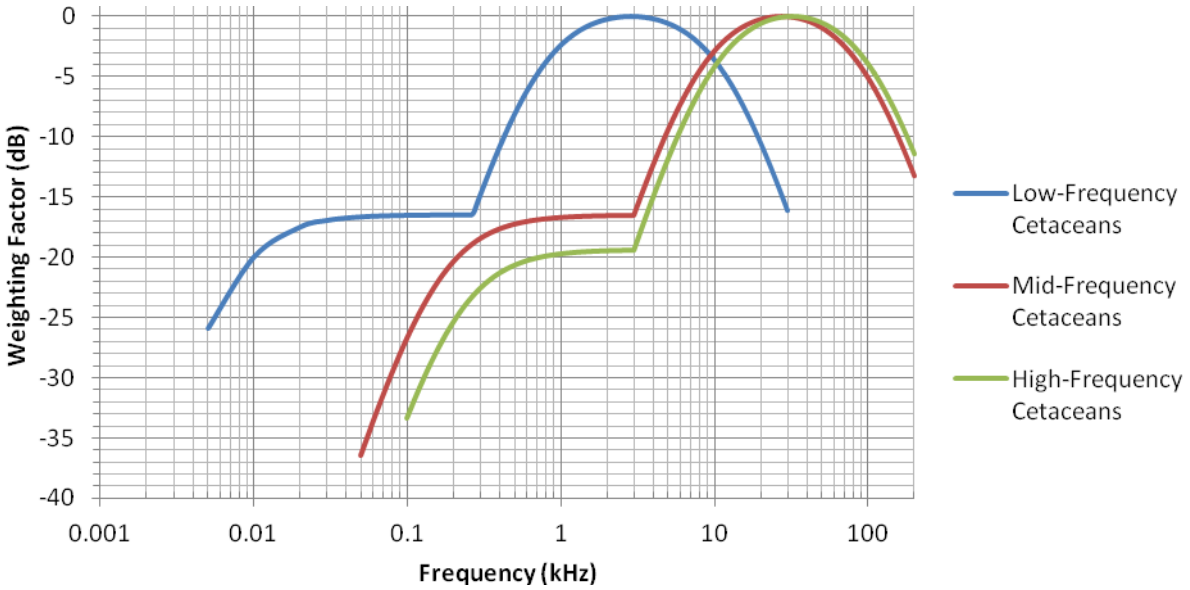

Figure 13. Type II Weighting Functions for Low-, Mid-, and High-Frequency Cetaceans

6.3.14.2 *Summation of Energy from Multiple Sources*

In most cases, an animal's received level will be the result of exposure to a single sound source. In some scenarios, however, multiple sources will be operating simultaneously, or nearly so, creating the potential for accumulation of energy from multiple sources. Energy is summed for multiple exposures of similar source types. For sonar, including use of multiple systems within any scenario, energy will be summed for all exposures within a cumulative exposure band, with the cumulative exposure bands defined in four bands: 0 to 1.0 kHz (low-frequency sources), 1.1

to 10.0 kHz (mid-frequency sources), 10.1 kHz to 100.0 kHz (high-frequency sources), and above 100.0 kHz (very high-frequency sources). Sources operated at frequencies above 200 kHz are considered to be inaudible to all groups of marine mammals and are not analyzed in the quantitative modeling of exposure levels. After the energy has been summed within each frequency band, the band with the greatest amount of energy is used to evaluate the onset of PTS or TTS. For explosives, including use of multiple explosives in a single scenario, energy is summed across the entire frequency band.

6.3.14.3 *Hearing Loss – Temporary and Permanent Threshold Shift*

Criteria for physiological effects (Table 18) from sonar and other active acoustic sources are based on TTS and PTS with thresholds based on cumulative sound exposure levels. The onset of TTS or PTS from exposure to impulsive sources is predicted using a sound exposure level-based threshold in conjunction with a peak pressure threshold. The horizontal ranges are then compared, with the threshold producing the longest range being the one used to predict effects. For multiple exposures within any 24-hour period, the received sound exposure level for individual events is accumulated for each animal. Since no studies have been designed to intentionally induce PTS in marine mammals, onset-PTS levels have been estimated using empirical TTS data obtained from marine mammals and relationships between TTS and PTS established in terrestrial mammals.

Temporary and permanent threshold shift thresholds are based on TTS onset values for impulsive and non-impulsive sounds obtained from representative species of mid- and high-frequency cetaceans and pinnipeds. These data are then extended to the other marine mammals for which data are not available. The *Criteria and Thresholds for Navy Acoustic and Explosive Effects Analysis Technical Report* provides a detailed explanation of the selection of criteria and derivation of thresholds for temporary and permanent hearing loss for marine mammals (Finneran and Jenkins 2012).

Table 18. Acoustic Criteria and Thresholds for Predicting Physiological Effects to Marine Mammals Underwater from Sonar and Other Active Acoustic Sources

Hearing Group	Species	Onset temporary threshold shift	Onset permanent threshold shift
Low-Frequency Cetaceans	All mysticetes	178 dB re 1 μPa^2-s SEL (Type II weighting)	198 dB re 1 μPa^2-s SEL (Type II weighting)
Mid-Frequency Cetaceans	Dolphins, beaked whales, and medium and large toothed whales	178 dB re 1 μPa^2-s SEL (Type II weighting)	198 dB re 1 μPa^2-s SEL (Type II weighting)
Otariidae (underwater)	Sea Lion & Fur Seals	206 dB re 1 μPa^2-s SEL (Type I weighting)	220 dB re 1 μPa^2-s SEL (Type I weighting)

Notes: dB = decibels, SEL = Sound Exposure Level, dB re 1 μPa^2-s = decibels referenced to 1 micropascal squared second

6.3.14.3.1 Temporary Threshold Shift – Non-Impulsive Sources

The onset of TTS in mid-frequency cetaceans exposed to non-impulsive sound are derived from multiple studies (Finneran et al. 2010; Finneran et al. 2005; Finneran and Schlundt 2010; Mooney 2009; Schlundt et al. 2000a) from two species, bottlenose dolphins and beluga whales. Especially notable are data for frequencies above 3 kHz, where bottlenose dolphins have exhibited lower TTS onset thresholds than at 3 kHz (Finneran 2011; Finneran and Schlundt 2010). This difference in TTS onset at higher frequencies is incorporated into the weighting functions.

Lucke et al. (2009) measured TTS in a harbor porpoise exposed to a small seismic air gun and those results are reflected in the current impulsive sound TTS thresholds described below. The beluga whale, which had been the only species for which both impulsive and non-impulsive TTS data exist, has a non-impulsive TTS onset value about 6 dB above the (weighted) impulsive threshold (Finneran et al. 2002; Schlundt et al. 2000a). Therefore, 6 dB was added to the harbor porpoise's impulsive TTS threshold demonstrated by Lucke et al. (2009) to derive the non-impulsive TTS threshold used in the current Navy modeling for high frequency cetaceans. The first direct measurements of TTS from non-impulsive sound was presented by Kastelein et al. (2012b) for harbor porpoise. These data are fully consistent with the current harbor porpoise thresholds used in the modeling of effects from non-impulsive sources.

There are no direct measurements of TTS or hearing abilities for low-frequency cetaceans. The Navy uses mid-frequency cetacean thresholds to assess PTS and TTS for low-frequency cetaceans, since mid-frequency cetaceans are the most similar to the low frequency group (see Finneran and Jenkins (2012) on the development of the thresholds and criteria).

Pinniped TTS criteria are based on data provided by Kastak et al. (2005) for representative species of both of the pinniped hearing groups: harbor seals (Phocidae) and California sea lions (Otariidae). Kastak et al. (2005) used octave band noise centered at 2.5 kHz to extrapolate an onset TTS threshold. More recently Kastelein et al. (2012a) used octave band noise centered at 4 kHz to obtain TTS thresholds in the same two species resulting in similar levels causing onset-TTS as those found in Kastak et al. (2005). For sea otters, the otariid TTS threshold and weighting function are applied due to similarities in taxonomy and auditory performance. The appropriate frequency weighting function for each species group is applied when using the sound exposure level-based thresholds to predict TTS.

Table 19 illustrates the ranges to the onset of TTS (i.e., the maximum distances to which TTS would be expected) for one, five, and ten pings from four representative source bins and sonar systems. Due to the lower acoustic thresholds for TTS versus PTS, ranges to TTS are longer; this can also be thought of as a larger volume acoustic footprint for TTS effects. Because the effects threshold is total summed sound energy and because of the longer distances, successive pings can add together, further increasing the range to onset-TTS.

Table 19. Approximate Maximum Ranges to the Onset of Temporary Threshold Shift for Four Representative Sonar Bins Over a Representative Range of Ocean Environments

Functional Hearing Group	Approximate Ranges to the Onset of TTS (meters)											
	Sonar Bin MF1 (e.g., SQS-53; ASW Hull-Mounted Sonar)			Sonar Bin MF4 (e.g., AQS-22; ASW Dipping Sonar)			Sonar Bin MF5 (e.g., SSQ-62; ASW Sonobuoy)			Sonar Bin HF4 (e.g., SQQ-32; MIW Sonar)		
	One Ping	Five Pings	Ten Pings	One Ping	Five Pings	Ten Pings	One Ping	Five Pings	Ten Pings	One Ping	Five Pings	Ten Pings
Low-frequency cetaceans	560-2,280	1,230-6,250	1,620-8,860	220-240	490-1,910	750-2,700	110-120	240-310	340-1,560	100-160	150-730	150-820
Mid-frequency cetaceans	150-180	340-440	510-1,750	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50
Otariid seals, sea lion, & Mustelid (sea otter)	230-570	1,240-1,300	1,760-1,780	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50

6.3.14.3.2 Temporary Threshold Shift – Impulsive Sources

The TTS sound exposure level thresholds for cetaceans are consistent with the USS MESA VERDE ship shock trial that was approved by NMFS (73 FR 143) and are more representative of TTS induced from impulses (Finneran et al. 2002) rather than pure tones (Schlundt et al. 2000a). In most cases, a total weighted sound exposure level is more conservative than greatest sound

exposure level in one-third octave bands, which was used prior to the USS MESA VERDE ship shock trials. There are no data on TTS obtained directly from low-frequency cetaceans, so mid-frequency cetacean impulse threshold criteria from Finneran et al. (2002) have been used. High frequency cetacean TTS thresholds are based on research by Lucke et al. (2009), who exposed harbor porpoises to pulses from a single air gun.

Pinniped criteria were not included for prior ship shock trials, as pinnipeds were not expected to occur at the shock trial sites, and TTS criteria for previous Navy EIS/OEISs did not differentiate between cetaceans and pinnipeds (NMFS 2008b; NMFS 2008d). TTS values for impulse sound criteria have not been obtained for pinnipeds, but there are TTS data for octave band sound from representative species of both major pinniped hearing groups (Kastak et al. 2005). Impulsive sound TTS criteria for pinnipeds were estimated by applying the difference between mid-frequency cetacean TTS onset for impulsive and non-impulsive sounds to the pinniped non-impulsive TTS data (Kastak et al. 2005), a methodology originally developed by Southall et al. (Southall et al. 2007b). Therefore, the TTS criteria for impulsive sounds from explosions for pinnipeds is 6 dB less than the non-impulsive onset-TTS criteria derived from Kastak et al. (2005).

Table 20 illustrates the average approximate range to TTS from explosions for marine mammals in the TMAA. These are conservative estimates (i.e., longer ranges), based on assuming all impulses are one second in duration. Most impulses are much less than one second and therefore, contain less energy than what is being used to produce the estimated ranges below.

Table 20. Average approximate range to TTS from explosions for marine mammals within the TMAA. Ranges are in meters.

Hearing Group	Bin E5 (> 5–10 lb. NEW)	Bin E9 (> 100–250 lb. NEW)	Bin E10 (> 250–500 lb. NEW)	Bin E12 (> 650–1,000 lb. NEW)
Low-frequency cetaceans	445	515	690	1,760
Mid-frequency cetaceans	215	355	435	720
Otariidae	85	220	260	400

6.3.14.3.3 Permanent Threshold Shift – Non-Impulsive Sources

There are no direct measurements of PTS onset in marine mammals. Well understood relationships between terrestrial mammalian TTS and PTS have been applied to marine mammals. Threshold shifts up to 40 to 50 dB have been induced in terrestrial mammals without resultant PTS (Miller et al. 1963; Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b). These data would suggest that a PTS criteria of 40 dB would be reasonable for conservatively

predicting (overestimating) PTS in marine mammals. Data from terrestrial mammal training (Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b) show growth of TTS by 1.5 to 1.6 dB for every 1 dB increase in exposure level. The difference between measureable TTS onset (6 dB) and the selected 40 dB upper safe limit of TTS yields a difference in TTS of 34 dB which, when divided by a TTS growth function of 1.6 indicates that an increase in exposure of 21 dB would result in 40 dB of TTS. For simplicity and additional conservatism we have rounded that number down to 20 dB (Southall et al. 2007b).

Therefore, exposures to sonar and other active acoustic sources with levels 20 dB above those producing TTS are assumed to produce a PTS. For example, an onset-TTS criteria of 195 dB re 1 μPa^2-s would have a corresponding onset-PTS criteria of 215 dB re 1 μPa^2-s. This extrapolation process is identical to that recently proposed by Southall et al. (Southall et al. 2007b). The method overestimates or predicts greater effects than have actually been observed in tests on a bottlenose dolphin (Finneran and Schlundt 2010; Schlundt et al. 2006) and is therefore protective.

Kastak et al. (2007) obtained different TTS growth rates for pinnipeds than Finneran and colleagues obtained for mid-frequency cetaceans. NMFS recommended reducing the estimated PTS criteria for both groups of pinnipeds, based on the difference in TTS growth rate reported by Kastak et al. (2007) (14 dB instead of 20 dB).

The appropriate frequency weighting function for each species group is applied when using the sound exposure level-based thresholds to predict PTS.

Table 21 lists the ranges to the PTS threshold (i.e., range to the onset of PTS: the maximum distance to which PTS would be expected), relative to the marine mammal's functional hearing group, from three of the most powerful sonar systems. For a SQS-53 sonar transmitting for 1 second at 3 kHz and a representative source level of 235 dB re 1 μPa^2-s at 1 m, the range to PTS for the most sensitive species (the high-frequency cetaceans) extends from the source to a range of 100 m (110 yd.). Since any hull mounted sonar, such as the SQS-53, engaged in anti-submarine warfare training would be moving at 10 to 15 knots (5.1 to 7.7 m/second) and nominally pinging every 50 seconds, the vessel will have traveled a minimum distance of approximately 260 m (280 yd) during the time between those pings (10 knots is the speed used in the Navy Acoustic Effects Model). As a result, there is little overlap of PTS footprints from successive pings, indicating that in most cases, an animal predicted to receive PTS would do so from a single exposure (i.e., one ping). For all other functional hearing groups (low-frequency cetaceans, mid-frequency cetaceans, and phocid seals and manatees) single-ping PTS zones are within 100 m of the sound source. A scenario could occur where an animal does not leave the vicinity of a ship or travels a course parallel to the ship within the PTS zone; however, the distances required make PTS exposure less likely. For a Navy vessel moving at a nominal 10

knots, it is unlikely a marine mammal could maintain the speed to parallel the ship and receive adequate energy over successive pings to suffer PTS. For all sources except hull-mounted sonar (e.g., SQS-53 and BQQ-10) ranges to PTS are well within 50 m (55 yd), even for multiple pings (up to five pings) and the most sensitive functional hearing group (high-frequency cetaceans).

Table 21. Approximate Ranges to Permanent Threshold Shift Criteria for Each Functional Hearing Group for a Single Ping from Three of the Most Powerful Sonar Systems within Representative Ocean Acoustic Environments

Functional Hearing Group	Ranges to the Onset of PTS for One Ping (meters)		
	Sonar Bin MF1 (e.g., SQS-53; Anti-Submarine Warfare Hull Mounted Sonar)	Sonar Bin MF4 (e.g., AQS-22; Anti-Submarine Warfare Dipping Sonar)	Sonar Bin MF5 (e.g., SSQ-62; Anti-Submarine Warfare Sonobuoy)
Low-Frequency Cetaceans	70	10	<2
Mid-Frequency Cetaceans	10	<2	<2
Otariid Seals & Sea Lion, & Mustelid (Sea Otter)	10	<2	<2

6.3.14.3.4 Permanent Threshold Shift – Impulsive Sources

Since marine mammal PTS data from impulsive exposures do not exist, onset PTS levels for these animals are estimated by adding 15 dB to the sound exposure level-based TTS threshold and by adding 6 dB to the peak pressure based thresholds. These relationships were derived by Southall et al. (Southall et al. 2007b) from impulsive noise TTS growth rates in chinchillas. The appropriate frequency weighting function for each species group is applied when using the resulting sound exposure level-based thresholds, as shown in Figure 13, to predict PTS.

Table 22 illustrates the average approximate range to PTS from explosions for marine mammals in the TMAA. These are conservative estimates (i.e., longer ranges), based on assuming all impulses are one second in duration. Most impulses are much less than one second and therefore, contain less energy than what is being used to produce the estimated ranges below.

Table 22. Average approximate range to PTS from explosions for marine mammals within the TMAA. Ranges are in meters.

Hearing Group	Bin E5 (> 5–10 lb. NEW)	Bin E9 (> 100–250 lb. NEW)	Bin E10 (> 250–500 lb. NEW)	Bin E12 (> 650–1,000 lb. NEW)
Low-frequency cetaceans	170	255	305	485
Mid-frequency cetaceans	70	170	205	265
Otariidae	50	50	85	150

6.3.14.3.5 Mortality and Injury from Explosives

There is a considerable body of laboratory data on actual injury for impulse sound, usually from explosive pulses, obtained from tests with a variety of lab animals (mice, rats, dogs, pigs, sheep, and other species). Onset Slight Gastrointestinal (GI) Tract Injury, Onset Slight Lung Injury, and Onset Mortality (a 50 percent lung injury with mortality occurring in 1 percent of those having this injury) represent a series of effects with increasing likelihood of serious injury or lethality. Primary impulse injuries from explosive blasts are the result of differential compression and rapid re-expansion of adjacent tissues of different acoustic properties (e.g., between gas-filled and fluid-filled tissues or between bone and soft tissues). These injuries usually manifest themselves in the gas-containing organs (lung and gut) and auditory structures (e.g., rupture of the eardrum across the gas-filled spaces of the outer and inner ear) (Craig and Hearn 1998; Craig Jr. 2001a).

Criteria and thresholds for predicting injury and mortality to marine mammals from explosive sources are listed in Table 23. Upper and lower frequency limits of hearing are not applied for lethal and injurious exposures. These criteria and their origins are explained in greater detail in Finneran and Jenkins (2012), who covered the development of the thresholds and criteria for assessment of impacts.

Table 23. Criteria and Thresholds for Predicting Physiological Effects to Marine Mammals Underwater for Explosives

Group	Species	Onset TTS	Onset PTS	Onset Slight GI Tract Injury	Onset Slight Lung Injury	Onset Mortality
Low-Frequency Cetaceans	Mysticetes	172 dB re 1 μPa^2-s (low-freq weighting) or 224 dB Peak SPL	187 dB re 1 μPa^2-s (low-freq weighting) or 230 dB Peak SPL	237 dB re 1 μPa	Equation 1	Equation 2
Mid-Frequency Cetaceans	Odontocetes (Toothed Whales)	172 dB re 1 μPa^2-s (mid-freq weighting) or 224 dB Peak SPL	187 dB re 1 μPa^2-s (mid-freq weighting) or 230 dB Peak SPL			

Equations:

(1) $$= 39.1M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-\sec$$

(2) $$= 91.4M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-\sec$$

D_{rm} = depth of the receiver (animal) in meters; M = mass of the animals in kg; SPL = sound pressure level

[1] **Impulse calculated over a delivery time that is the lesser of the initial positive pressure duration or 20 percent of the natural period of the assumed-spherical lung adjusted for animal size and depth.**

Notes: TTS = temporary threshold shift, PTS = permanent threshold shift, GI = gastrointestinal, M = mass of animals in kilograms, D_{Rm} = depth of receiver (animal) in meters, SEL = Sound Exposure Level, SPL = Sound Pressure Level (re 1 μPa), dB = decibels, dB re 1 μPa = decibels referenced to 1 micropascal, dB re 1 μPa^2-s = decibels referenced to 1 micropascal squared second

Table 24 illustrates the average approximate ranges to onset mortality, onset slight lung injury, and onset slight GI tract injury from explosions for marine mammals in the TMAA. These are conservative estimates (i.e., longer ranges), based on assuming all impulses are one second in duration. Most impulses are much less than one second and therefore, contain less energy than what is being used to produce the estimated ranges below.

Table 24. Average approximate range to onset mortality, onset slight lung injury, and onset slight GI tract injury from explosions for marine mammals within the TMAA. Ranges are in meters.

Hearing Group	Predicted Impact	Bin E5 (> 5–10 lb. NEW)	Bin E9 (> 100–250 lb. NEW)	Bin E10 (> 250–500 lb. NEW)	Bin E12 (> 650–1,000 lb. NEW)
Low-frequency cetaceans	Onset mortality	20	65	80	95
	Onset slight lung injury	40	110	135	165
	Onset slight GI tract injury	80	145	180	250
Mid-frequency cetaceans	Onset mortality	45	135	165	200
	Onset slight lung injury	85	235	285	345
	Onset slight GI tract injury	80	145	180	250
Otariidae	Onset mortality	65	175	215	260
	Onset slight lung injury	115	307	370	450
	Onset slight GI tract injury	8	145	180	250

6.3.14.3.6 Onset of Gastrointestinal Tract Injury

Evidence indicates that gas-containing internal organs, such as lungs and intestines, are the principal damage sites from shock waves in submerged terrestrial mammals (Clark and Ward 1943; Greaves et al. 1943; Richmond et al. 1973; Yelverton et al. 1973). Furthermore, slight injury to the gastrointestinal tract may be related to the magnitude of the peak shock wave pressure over the hydrostatic pressure and would be independent of the animal's size and mass (Goertner 1982).

There are instances where injury to the gastrointestinal tract could occur at a greater distance from the source than slight lung injury, especially for animals near the surface. Gastrointestinal tract injury from small test charges (described as "slight contusions") was observed at peak pressure levels as low as 104 pounds per square inch (psi), equivalent to a sound pressure level of 237 dB re 1 µPa (Richmond et al. 1973). This criterion was previously used by the Navy and NMFS for ship shock trials (Finneran and Jenkins (2012); 63 FR 230, 66 FR 87, 73 FR 143).

6.3.14.3.7 Slight Lung Injury and Mortality

The most commonly reported internal bodily injury from impulse energy is hemorrhaging in the fine structure of the lungs. Biological damage is governed by the impulse of the underwater blast (pressure integrated over time), not peak pressure or energy (Richmond et al. 1973; Yelverton and Richmond 1981; Yelverton et al. 1973; Yelverton et al. 1975). Therefore, impulse was used as a metric upon which internal organ injury could be predicted. Species-specific minimal animal masses are used for determining impulse-based thresholds of slight lung injury and mortality. The Criteria and Thresholds for Navy Acoustic and Explosive Effects Analysis technical report (Finneran and Jenkins 2012) provides a nominal conservative body mass for each species based on newborn weights. In some cases body masses were extrapolated from similar species rather than the listed species. The scaling of lung volume to depth is conducted for all species since data is from experiments with terrestrial animals held near the water's surface.

Because the thresholds for onset of mortality and onset of slight lung injury are proportional to the cube root of body mass, the use of all newborn, or calf, weights rather than representative adult weights results in an over-estimate of effects to animals near an explosion. The range to onset mortality for a newborn compared to an adult animal of the same species can range from less than twice to over four times as far from an explosion, depending on the differences in calf versus adult sizes for a given species and the size of the explosion. Considering that injurious high pressures due to explosions propagate away from detonations in a roughly spherical manner, the volumes of water in which the threshold for onset mortality may be exceeded are generally less than a fifth for an adult animal versus a calf.

The use of onset mortality and onset slight lung injury is a conservative method to estimate potential mortality and recoverable (non-mortal, non-PTS) injuries, respectively. When analyzing impulse-based effects, all animals within the range to these thresholds are assumed to experience the effect. The onset mortality and onset slight lung injury criteria is based on the impulse at which these effects are predicted for 1 percent of animals; the portion of animals affected would increase closer to the explosion. All animals receive the effect vice a percentage; therefore, these criteria conservatively over-estimate the number of animals that could be killed or injured.

Impulse thresholds for onset mortality and slight injury are indexed to 75 and 93 lb. (34 and 42 kg) for mammals, respectively (Richmond et al. 1973). The regression curves based on these experiments were plotted such that a prediction of mortality to larger animals could be determined as a function of positive impulse and mass (Craig Jr. 2001a). After correction for atmospheric and hydrostatic pressures and based on the cube root scaling of body mass, as used in the Goertner injury model (Goertner 1982), the minimum impulse for predicting onset of extensive (50 percent) lung injury for "1 percent Mortality" (defined as most survivors had moderate blast injuries and should survive on their own) and slight lung injury for "zero percent

Mortality" (defined as no mortality, slight blast injuries) (Yelverton and Richmond 1981) were derived for each species. As the mortality threshold, the Navy chose to use the minimum impulse level predictive of 50 percent lung injury, even though this injury is likely to result in mortality to only 1 percent of exposed animals. Because the mortality criteria represents a threshold at which 99 percent of exposed animals would be expected to recover, this analysis greatly overestimates the impact on individuals and populations from exposure to impulse sources.

6.3.14.4 *Behavioral Responses*

The behavioral response criteria are used to estimate the number of animals that may exhibit a behavioral response. In this analysis, animals may be behaviorally harassed in each modeled scenario (using the Navy Acoustic Effects Model) or within each 24-hour period, whichever is shorter. Therefore, the same animal could have a behavioral reaction multiple times over the course of a year.

6.3.14.4.1 Sonar and Other Active Acoustic Sources

Potential behavioral effects to marine mammals from non-impulse sound sources underwater were predicted using a behavioral response function. The received sound level is weighted with Type I auditory weighting functions (Southall et al. 2007b) before the behavioral response function is applied. The behavioral response functions estimate the percentage of an exposed population that is likely to exhibit reactions to the sound source. This effects analysis assumes that the potential response from an exposure to non-impulsive sound on individual animals would be a function of the received sound pressure level (SPL; dB re 1 μPa). For example, at 165 dB SPL (dB re 1μPa root mean square), the risk (or probability) of response is defined according to this function as 50 percent. This means that 50 percent of the individuals exposed at that received level would be predicted to exhibit behavioral response.

The behavioral response functions are based on three sources of data: behavioral observations during TTS experiments conducted at the US Navy Marine Mammal Program (Finneran and Schlundt 2004); reconstruction of sound fields produced by the USS Shoup associated with the behavioral responses of killer whales observed in Haro Strait (DoN 2003; Fromm 2009); and observations of the behavioral response of North Atlantic right whales exposed to alert stimuli containing mid-frequency components (Nowacek et al. 2004a). For a detailed discussion of the derivation of the BRFs, see the 2013 Atlantic Fleet Training and Testing FEIS (DoN 2013a). The behavioral response function applied to mysticetes (Figure 14) differs from that used for odontocetes and pinnipeds (Figure 15) in having a shallower slope, which results in the inclusion of more behavioral events at lower amplitudes, consistent with observational data from North Atlantic right whales (Nowacek and Tyack 2007). Although the response functions differ, the intercepts on each figure highlight that each function has a 50 percent probability of harassment at a received level of 165 dB SPL. These analyses assume that sound poses a negligible risk to

marine mammals if they are exposed to sound pressure levels below a certain basement value. For both behavioral response functions, the basement received level is 120 dB re 1 μPa.

In some circumstances, some individuals will continue normal behavioral activities in the presence of high levels of human-made noise. In other circumstances, the same individual or other individuals may avoid an acoustic source at much lower received levels (Richardson et al. 1995c; Southall et al. 2007c; Wartzok et al. 2003). These differences within and between individuals appear to result from a complex interaction of experience, motivation, and learning that are difficult to quantify and predict. Therefore, the behavioral response functions represent a relationship that is deemed generally accurate, but may not be true in specific circumstances. Specifically, the behavioral response function treats the received level as the only variable that is relevant to a marine mammal's behavioral response. However, many other variables such as the marine mammal's gender, age, and prior experience; the activity it is engaged in during a sound exposure; its distance from a sound source; the number of sound sources; and whether the sound sources are approaching or moving away from the animal can be critically important in determining whether and how a marine mammal will respond to a sound source (Southall et al. 2007c). At present, available data do not allow for incorporation of these other variables in the current behavioral response functions; however, the response function represents the best use of the data that are available.

The Navy uses the behavioral response function to quantify the number of behavioral responses that could qualify as Level B behavioral harassment under the MMPA. As the statutory definition is currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated using the behavioral response functions do not differentiate between the different types of potential reactions nor the significance of those potential reactions. These estimates also do not provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. Therefore, our analysis considers the available scientific evidence to determine the likely nature of modeled behavioral responses and potential fitness consequences for affected individuals.

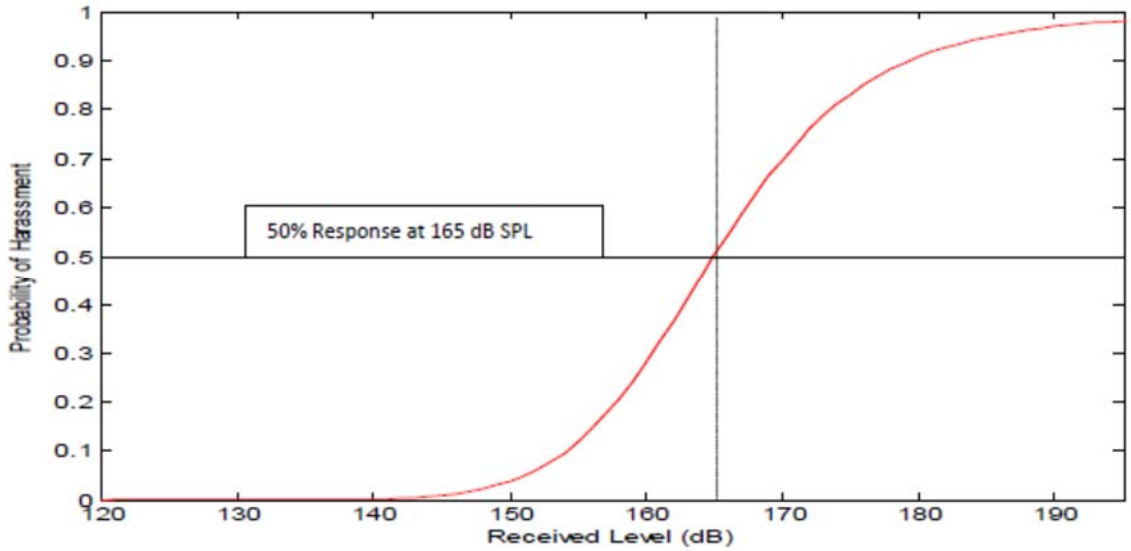

Figure 14. Behavioral response function applied to mysticetes.

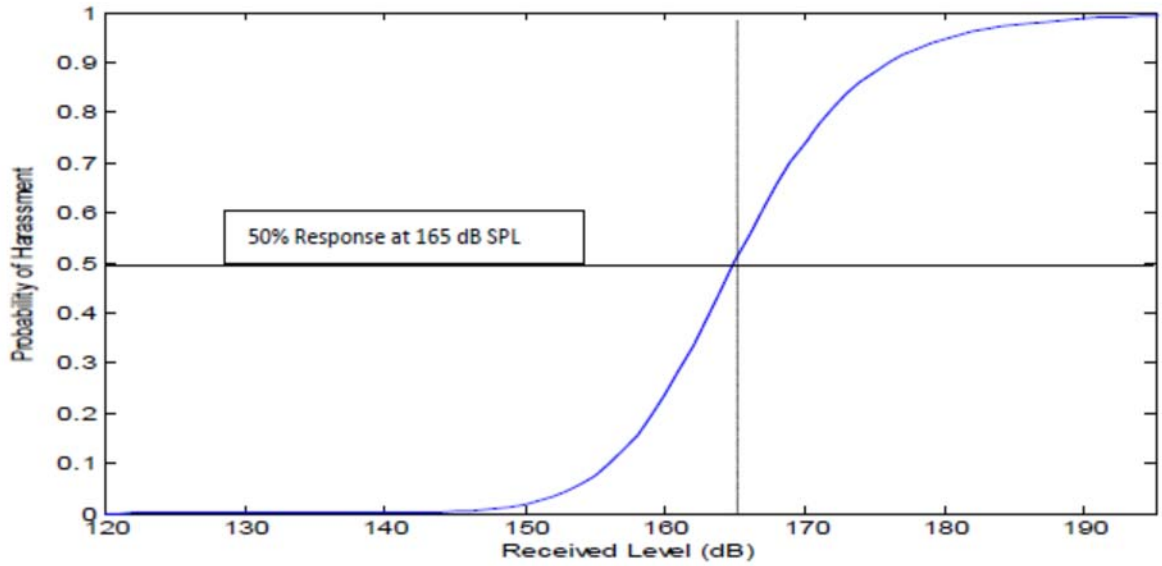

Figure 15. Behavioral response function applied to odontocetes and pinnipeds.

The distances over which the sound pressure level from four representative sonar sources is within the indicated 6-dB bins, and the percentage of animals that may exhibit a significant behavioral response under the mysticete and odontocete/pinniped behavioral response function, are shown in Table 25 and Table 26 respectively.

Table 25. Range to Received Sound Pressure Level (SPL) in 6-dB Increments and Percentage of Behavioral Harassments for Low-Frequency Cetaceans under the Mysticete Behavioral Response Function for Four Representative Source Bins for the Action Area

Received in 6-dB Increments	Source Bin MF1 (e.g., SQS-53; Anti-Submarine Warfare Hull Mounted Sonar)		Source Bin MF4 (e.g., AQS- 22; Anti-Submarine Warfare Dipping Sonar)		Source Bin MF5 (e.g., SSQ-62; Anti-Submarine Warfare Sonobuoy)		Source Bin HF4 (e.g., SQQ- 32; Mine Integrated Warfare Sonar)	
	Approximate Distance (m)	Behavioral Harassment % from SPL Increment	Approximate Distance (m)	Behavioral Harassment % from SPL	Approximate Distance (m)	Behavioral Harassment % from SPL Increment	Approximate Distance (m)	Behavioral Harassment % from SPL
120 <= SPL <126	172,558 – 162,925	0.00%	40,000 – 40,000	0.00%	23,880 – 17,330	0.00%	3,100 – 2,683	0.00%
126 <= SPL <132	162,925 – 117,783	0.00%	40,000 – 40,000	0.00%	17,330 – 12,255	0.10%	2,683 – 2,150	0.01%
132 <= SPL <138	117,783 – 108,733	0.04%	40,000 – 12,975	3.03%	12,255 – 7,072	4.12%	2,150 – 1,600	0.48%
138 <= SPL <144	108,733 – 77,850	1.57%	12,975 – 12,800	0.14%	7,072 – 3,297	23.69%	1,600 – 1,150	4.20%
144 <= SPL <150	77,850 – 58,400	5.32%	12,800 – 6,525	27.86%	3,297 – 1,113	42.90%	1,150 - 575	24.79%
150 <= SPL <156	58,400 – 53,942	4.70%	6,525 – 2,875	36.83%	1,113 - 255	24.45%	575 - 300	28.10%
156 <= SPL <162	53,942 – 8,733	83.14%	2,875 – 1,088	23.78%	255 - 105	3.52%	300 - 150	24.66%
162 <= SPL <168	8,733 – 4,308	3.51%	1,088 - 205	7.94%	105 - <50	1.08%	150 - 100	9.46%
168 <= SPL <174	4,308 – 1,950	1.31%	205 - 105	0.32%	<50	0.00%	100 - <50	8.30%
174 <= SPL <180	1,950 – 850	0.33%	105 - <50	0.10%	<50	0.00%	<50	0.00%
180 <= SPL <186	850 – 400	0.06%	<50	0.01%	<50	0.13%	<50	0.00%
186 <= SPL <192	400 – 200	0.01%	<50	0.00%	<50	0.00%	<50	0.00%
192 <= SPL <198	200 – 100	0.00%	<50	0.00%	<50	0.00%	<50	0.00%

Table 26. Range to Received Sound Pressure Level (SPL) in 6-dB Increments and Percentage of Behavioral Harassments for Mid-Frequency Cetaceans and Pinnipeds under the Odontocete Response Function for Four Representative Source Bins

Received Level in 6-dB Increments	Source Bin MF1 (e.g., SQS-53; Anti-Submarine Warfare Hull Mounted Sonar)		Source Bin MF4 (e.g., AQS- 22; Anti-Submarine Warfare Dipping Sonar)		Source Bin MF5 (e.g., SSQ-62; Anti-Submarine Warfare Sonobuoy)		Source Bin HF4 (e.g., SQQ- 32; Mine Integrated Warfare Sonar)	
	Approximate Distance (m)	Behavioral Harassment % from SPL Increment	Approximate Distance (m)	Behavioral Harassment % from SPL	Approximate Distance (m)	Behavioral Harassment % from SPL Increment	Approximate Distance (m)	Behavioral Harassment % from SPL Increment
120 <= SPL <126	172,592 – 162,933	0.00%	40,000 – 40,000	0.00%	24,205 – 18,872	0.00%	4,133 – 3,600	0.00%
126 <= SPL <132	162,933 – 124,867	0.00%	40,000 – 40,000	0.00%	18,872 – 12,697	0.10%	3,600 – 3,075	0.00%
132 <= SPL <138	124,867 – 108,742	0.07%	40,000 – 12,975	2.88%	12,697 – 7,605	3.03%	3,075 – 2,525	0.01%
138 <= SPL <144	108,742 – 78,433	1.54%	12,975 – 12,950	0.02%	7,605 – 4,080	17.79%	2,525 – 1,988	0.33%
144 <= SPL <150	78,433 – 58,650	5.41%	12,950 – 6,725	26.73%	4,080 – 1,383	46.83%	1,988 – 1,500	2.83%
150 <= SPL <156	58,650 – 53,950	4.94%	6,725 – 3,038	36.71%	1,383 - 300	27.08%	1,500 – 1,000	14.92%
156 <= SPL <162	53,950 – 8,925	82.62%	3,038 – 1,088	25.65%	300 - 155	3.06%	1,000 - 500	40.11%
162 <= SPL <168	8,925 – 4,375	3.66%	1,088 - 255	7.39%	155 - 55	2.02%	500 - 300	22.18%
168 <= SPL <174	4,375 – 1,992	1.34%	255 - 105	0.52%	55 - <50	0.00%	300 - 150	14.55%
174 <= SPL <180	1,992 – 858	0.34%	105 - <50	0.09%	<50	0.00%	150 - <50	5.07%
180 <= SPL <186	858 – 408	0.06%	<50	0.01%	<50	0.09%	<50	0.00%
186 <= SPL <192	408 – 200	0.01%	<50	0.00%	<50	0.00%	<50	0.00%
192 <= SPL <198	200 – 100	0.00%	<50	0.00%	<50	0.00%	<50	0.00%

6.3.14.4.2 Explosives

The thresholds for a behavioral response from explosives are listed in Table 27. Appropriate weighting values will be applied to the received impulse in one-third octave bands and the

energy summed to produce a total weighted SEL value. For impulsive behavioral criteria, the new weighting functions (Table 27) are applied to the received sound level before being compared to the threshold.

Table 27. Summary of behavioral response thresholds for marine mammals.

Group	Species	Behavioral thresholds for sonar and other active acoustic sources	Behavioral thresholds for explosions
Low-frequency cetaceans	All mysticetes	SPL: BRF (Type I Weighting)	167 dB re 1µPa2 -s SEL (Type II Weighting)
Mid-frequency cetaceans	Dolphins, beaked whales, and medium and large toothed whales	SPL: BRF (Type I Weighting)	167 dB re 1µPa2 -s SEL (Type II Weighting)
High-frequency cetaceans	Porpoises and *Kogia* spp.	SPL: BRF (Type I Weighting)	141 dB re 1µPa2 -s SEL (Type II Weighting)
Otariid seals (underwater)	Stellar Sea Lion	SPL: BRF (Type I Weighting)	172 dB re 1µPa2 -s SEL (Type I Weighting)

BRF: Behavioral Response Function, SPL: Sound Pressure Level, SEL: Sound Exposure Level

If more than one explosive event occurs within any given 24-hour period within a training activity, criteria are applied to predict the number of animals that may have a behavioral reaction. For events with multiple explosions, the behavioral threshold used in this analysis is 5 dB less than the TTS onset threshold (in SEL). Some multiple explosion events, such as certain gunnery exercises, may be treated as a single impulsive event because a few explosions occur closely spaced within a very short time (a few seconds). For single explosions at received sound levels below hearing loss thresholds, the most likely behavioral response is a brief alerting or orienting response. Since no further sounds follow the initial brief impulse, significant behavioral reactions would not be expected to occur.

Since impulse events can be quite short, it may be possible to accumulate multiple received impulses at sound pressure levels considerably above the energy-based criterion and still not be considered a behavioral take. All individual received impulses were treated as if they were 1 second long for the purposes of calculating cumulative SEL for multiple impulse events. For example, five air gun impulses, each 0.1 second long, received at 178 dB sound pressure level would equal a 175 dB SEL and would not be predicted as leading to a significant behavioral response. However, if the five 0.1 second pulses are treated as a 5-second exposure, it would yield an adjusted value of approximately 180 dB, exceeding the threshold. For impulses associated with explosions that have durations of a few microseconds, this assumption greatly overestimates effects based on SEL metrics such as TTS and PTS and behavioral responses.

Table 22 illustrates the average approximate range to behavioral response from explosions for marine mammals in the TMAA. These are conservative estimates (i.e., longer ranges), based on assuming all impulses are one second in duration. Most impulses are much less than one second and therefore, contain less energy than what is being used to produce the estimated ranges below.

Table 28. Average approximate range to behavioral response from explosions for marine mammals within the TMAA. Ranges are in meters.

Hearing Group	Bin E5 (> 5–10 lb. NEW)	Bin E9 (> 100–250 lb. NEW)	Bin E10 (> 250–500 lb. NEW)	Bin E12 (> 650–1,000 lb. NEW)
Low-frequency cetaceans	525	710	905	2,655
Mid-frequency cetaceans	285	455	555	970
Otariidae	145	300	350	530

6.4 Risk Assessment Framework – Fish

This section is largely based on a technical report prepared for the Navy: Effects of Mid- and High-Frequency Sonars on Fish (Popper 2008b). Additionally, Popper and Hastings (Popper and Hastings 2009a; Popper and Hastings 2009b) provide a critical overview of some of the most recent research regarding potential effects of anthropogenic sound on fish.

Studies of the effects of human-generated sound on fish have been reviewed in numerous places (e.g., (Hastings and Popper 2005; NRC 1994b; Popper 2003; Popper 2008a; Popper and Hastings 2009a; Popper and Hastings 2009b; Popper et al. 2004)). Most investigations, however, have been in the gray literature (non-peer-reviewed reports—see (Hastings and Popper 2005; Popper 2008a; Popper and Hastings 2009a) for extensive critical reviews of this material). Studies have been published assessing the effect on fish of short-duration, high-intensity signals such as might be found near high-intensity sonar, pile driving, or seismic air guns. The investigators in such studies examined short-term effects that could result in death to the exposed fish, as well as hearing loss and long-term consequences (Doksaeter et al. 2009; Govoni et al. 2003; McCauley et al. 2003; Popper et al. 2007; Popper et al. 2005). Information is also discussed from a technical report that resulted from a working group established by the Acoustical Society of America on sound exposure guidelines for fishes and sea turtles (Popper et al. 2014b).

6.4.1 Direct Injury of Fish from Non-Impulsive Acoustic Stressors

Potential direct injuries from non-impulsive sound sources, such as sonar, are unlikely because of the relatively lower peak pressures and slower rise times than potentially injurious sources such as explosives. Non-impulsive sources also lack the strong shock wave such as that

associated with an explosion. The theories of sonar induced acoustic resonance, bubble formation, neurotrauma, and lateral line system injury are discussed below, although these would likely occur only in fish very close to the sound source and are therefore unlikely to impact entire populations of fish or have an impact in a large area.

No studies have indicated any physiological damage to adult fish from mid-frequency active sonar. Kvadsheim and Sevaldsen (2005) showed that intense sonar activities in herring spawning areas affected less than 0.3 percent of the total juvenile stock. Jørgensen et al. (2005) exposed fish larvae and juveniles representing four species (of three families) to sounds that were designed to simulate mid-frequency sonar transmissions (1 to 6.5 kHz) to study the effects of the exposure on the survival, development, and behavior of the larvae and juveniles (the study used larvae and juveniles of Atlantic herring (*Clupea harengus*) Atlantic cod (*Gadus morhua*), saithe (*Pollachius virens),* and spotted wolffish (*Anarhichas minor*)). The researchers placed the fish in plastic bags three meters from the sound source and exposed them to between four and 100 pulses of one-second duration of pure tones at 1.5, 4, and 6.5 kHz. The fish in only two groups out of the 42 tested exhibited adverse effects beyond a behavioral response. These two groups were both composed of herring, a hearing specialist, and were tested with sound pressure levels of 189 dB re 1 μPa, which resulted in a post-exposure mortality of 20 to 30 percent. In the remaining 40 tests, there were no observed effects on behavior, growth (length and weight), or the survival of fish that were kept as long as 34 days post exposure. While statistically significant losses were documented in the two groups impacted, the researchers only tested that particular sound level once, so it is not known if this increased mortality was due to the level of the test signal or to other unknown factors.

Halvorsen et al. (2012) exposed rainbow trout to simulated MFA (2.8 to 3.8 kHz) sonar at received sound pressure levels of 210 dB re 1 uPa, resulting in cumulative sound exposure levels of 220 dB re 1 uPa. The authors did not observe any mortality or hearing sensitivity changes in rainbow trout and suggested that the frequency range of MFA sonar may be above the most sensitive hearing range of the species. Similarly, Kane et al. (2010) found that low-and mid-frequency exposure caused no acute, gross or histopathology, nor any mortality to rainbow trout, *I. punctatus*, or *Lepomis* sp. Popper et al. (2008c; 2007; Popper and Hastings 2009b) investigated the effects of exposing several fish species to the Navy's SURTASS LFA sonar, focusing on the hearing and on non-auditory tissues. Their study exposed the fish to LFA sonar pulses for time intervals that would be substantially longer than what would occur in nature, but the fish did not experience mortalities or damage to body tissues at the gross or histological level.

Swim bladder resonance is a function of the size and geometry of the air cavity, depth of the fish, and frequency of the transmitted signal. Wavelengths associated with mid-frequency sounds are shorter than wavelengths associated with lower frequency sounds. It is the lower frequencies that are expected to produce swim bladder resonance in adult fishes. Resonance frequencies for

juvenile fish are 1 to 8 kHz and can escalate physiological impact (Kvadsheim and Sevaldsen 2005; Lovik and Hovem 1979).

High sound pressure levels may cause bubbles to form from micronuclei in the blood stream or other tissues of animals, possibly causing embolism damage (Ketten 1998). Fish 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 also take place in the eyes of fish due to potentially high gas saturation within the fish's eye tissues (Popper and Hastings 2009a; Popper and Hastings 2009b). As reviewed in Popper and Hastings Popper and Hastings (2009a), Hastings (1990; 1995b) found 'acoustic stunning' (loss of consciousness) in blue gouramis (*Trichogaster trichopterus*) following an 8-minute exposure to a 150 Hz pure tone with a peak sound pressure level (SPL) of 198 dB re 1 μPa. This species of fish has an air bubble in the mouth cavity directly adjacent to the animal's braincase that may have caused this injury. Hastings (1990; 1995b) also found that goldfish exposed to two hours of continuous wave sound at 250 Hz with peak pressures of 204 dB re 1 μPa, and fathead minnows exposed to 0.5 hours of 150 Hz continuous wave sound at a peak level of 198 dB re 1 μPa did not survive. The only study on the effect of exposure of the lateral line system to continuous wave sound (conducted on one freshwater species) suggests no effect on these sensory cells by intense pure tone signals (Hastings et al. 1996).

Popper et al. (2014b) developed sound exposure guidelines for fishes exposed to low and mid-frequency naval sonar. The authors did not did not provide evidence that injury or mortality could occur from naval sonar, and indicated that if injury or mortality occurs, it is thought to begin at higher sound levels than have been tested to date. The authors concluded that the relative risk of injury or mortality to fish with no swim bladders exposed to low and mid-frequency sonar was low, no matter the distance from the source.

6.4.2 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue effects in fish is primary blast injury and barotrauma following exposure to explosions. Primary blast injury refers to those injuries that result from the initial compression of a body exposed to a blast wave. Primary blast injury is usually limited to gas-containing structures (e.g., swim bladder) and the auditory system. Barotrauma refers to injuries caused when the swim bladder or other gas-filled structures vibrate in response to the signal, particularly if there is a relatively sharp rise-time and the walls of the structure strike near-by tissues and damage them. The relative risk of injury or mortality to fish with no swim bladders exposed to explosions is anticipated to be much lower than fish with swim bladders.

An underwater explosion generates a shock wave that produces a sudden, intense change in local pressure as it passes through the water (DoN 1998; DoN 2001). Pressure waves extend to a

greater distance than other forms of energy produced by the explosion (i.e., heat and light) and are therefore the most likely source of negative effects to marine life from underwater explosions (Craig Jr. 2001b; DoN 2006; SIO 2005). The shock wave from an underwater explosion is lethal to fish at close range causing massive organ and tissue damage and internal bleeding (Keevin and Hempen 1997). At greater distance from the detonation point, the extent of mortality or injury depends on a number of factors including fish size, body shape, orientation, and species (Keevin and Hempen 1997; Wright 1982). Additional factors include the current physical condition of the fish and the presence of a swim bladder. At the same distance from the source, larger fish 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 fish oriented sideways to the blast suffer the greatest impact (Edds-Walton and Finneran 2006; O'Keefe and Young 1984; Wiley et al. 1981; Yelverton et al. 1975). Species with gas-filled organs have higher mortality than those without them (Continental Shelf Associates Inc. 2004; Goertner et al. 1994).

Two aspects of the shock wave appear most responsible for injury and death to fish: the received peak pressure and the time required for the pressure to rise and decay (Dzwilewski and Fenton 2002). Higher peak pressure and abrupt rise and decay times are more likely to cause acute pathological effects (Wright and Hopky 1998). Rapidly oscillating pressure waves might rupture the kidney, liver, spleen, and sinus and cause venous hemorrhaging (Keevin and Hempen 1997). They can also generate bubbles in blood and other tissues, possibly causing embolism damage (Ketten 1998). Oscillating pressure waves might also burst gas-containing organs. The swim bladder, the gas-filled organ used by most bony fish to control buoyancy, is the primary site of damage from explosives (Wright 1982; Yelverton et al. 1975). Gas-filled swim bladders resonate at different frequencies than surrounding tissue and can be torn by rapid oscillation between high- and low-pressure waves. The range over which damage may occur in a fish without a swim bladder is on the order of 100 times less than that for swim bladder fish (Popper et al. 2014b).

Studies that have documented fish 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 (Hubbs and Rechnitzer 1952; Yelverton et al. 1975). Fitch and Young (1948) found that the type of fish killed changed when blasting was repeated at the same marine 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. However, fishes collected during these types of studies have mostly been recovered floating on the water's surface. Gitschlag et al. (2001) collected both floating fish and those that were sinking or lying on the bottom after explosive removal of nine oil platforms in the northern Gulf of Mexico. They found that 3 to 87 percent (46 percent average) of the specimens killed during a blast might float to the surface. Other impediments to accurately characterizing the magnitude of fish mortality included currents and winds that transported floating fishes out of the sampling area and predation by seabirds or other fishes. Popper et al. (2014b) developed sound exposure guidelines for fishes

and generally estimated that mortality and mortal injury would occur when the peak sound pressure level from a single explosion exceeds 229 to 234 dB re 1 μPa.

There have been few studies of the impact of underwater explosions on early life stages of fishes (eggs, larvae, juveniles). Fitch and Young (1948) reported the demise of larval anchovies exposed to underwater blasts off California, and Nix and Chapman (1985) found that anchovy and eulachon larvae died following the detonation of buried charges. It has been suggested that impulsive sounds, such as that produced by seismic airguns, may cause damage to the cells of the lateral line in fish larvae and fry when in close proximity (15 ft. [5 m]) to the sound source (Booman et al. 1996). Similar to adult fishes, the presence of a swim bladder contributes to shock wave-induced internal damage in larval and juvenile fishes (Settle et al. 2002). Shock wave trauma to internal organs of larval pinfish and spot from shock waves was documented by Govoni et al. (2003). These were laboratory studies, however, and have not been verified in the field.

Interim criteria for injury of fish were discussed in Stadler and Woodbury (2009). The onset of physical injury would be expected if either the peak sound pressure level exceeds 206 dB re 1 μPa, or the cumulative sound exposure level, accumulated over all pile strikes generally occurring within a single day, exceeds 187 dB re 1 micropascal squared second (μPa^2-s) for fish two grams or larger, or 183 dB re 1 μPa^2-s for smaller fish (Stadler and Woodbury 2009). A more recent study by Halvorsen et al. (2011) used carefully controlled laboratory conditions to determine the level of pile driving sound that may cause a direct injury to the fish tissues (barotrauma). The investigators found that juvenile Chinook salmon (*Oncorhynchus tshawytscha*) received less than a single strike sound exposure level of 179 to 181 dB re 1μ Pa^2-s and cumulative sound exposure level of less than 211 dB re 1 μPa^2-s over the duration of the pile driving activity would sustain no more than mild, non-life-threatening injuries.

6.4.3 Hearing Loss

Available information on the hearing sensitivities of the species considered in this opinion is presented in Section 4. 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 TTS is a temporary, recoverable loss of hearing sensitivity. A TTS may last several minutes to several weeks and the duration may be related to the intensity of the sound source and the duration of the sound (including multiple exposures). A permanent threshold shift (PTS) is non-recoverable, results from the destruction of tissues within the auditory system, and can occur over a small range of frequencies related to the sound exposure. As with temporary threshold shift, 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; however, in this case, the effect is permanent.

Permanent hearing loss, or permanent threshold shift has not been documented in fish. The sensory hair cells of the inner ear in fish can regenerate after they are damaged, unlike in mammals where sensory hair cells loss is permanent (Lombarte et al. 1993; Smith et al. 2006a). As a consequence, any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells.

Sound is composed of two major components, the propagating sound pressure wave and particle motion. All fishes detect particle motion with their inner ear otoliths (otoconia in elasmobranchs) which act as accelerometers (Casper and Mann 2006). Sound pressure, however, can only be detected by fishes which have a pressure-to-displacement transducer, usually the swim bladder in some teleost fishes (Casper and Mann 2006). Some fishes have evolved a specialized connection between the swim bladder and the inner ear which can transmit the sound pressure signal being detected by the bladder. In the case of the otophysans such as goldfish, modified vertebrae known as the Weberian ossicles have evolved for this function.

6.4.3.1 *Non-impulsive Sound Sources*

Studies of the effects of long-duration sounds with sound pressure levels below 170 to 180 dB re 1 μPa indicate that there is little to no effect of long-term exposure on species (such as those considered in this opinion) that lack notable anatomical hearing specialization (Amoser and Ladich 2003; Scholik and Yan 2001; Smith et al. 2004a; Smith et al. 2004b; Wysocki et al. 2007a). The longest of these studies exposed young rainbow trout (*Onorhynchus mykiss*), to a level of noise equivalent to one that fish would experience in an aquaculture facility (e.g., on the order of 150 dB re 1 μPa) for about 9 months. The investigators found no effect on hearing (i.e., TTS) as compared to fish raised at 110 dB re 1 μPa.

In contrast, studies on fish with hearing specializations (i.e., greater sensitivity to lower sound pressures and higher frequencies) have shown that there is some hearing loss after several days or weeks of exposure to increased background sounds, although the hearing loss seems to recover (e.g., (Scholik and Yan 2002; Smith et al. 2006a; Smith et al. 2004b)). Smith et al. (Smith et al. 2006a; Smith et al. 2004b) exposed goldfish to noise at 170 dB re 1 μPa and found a clear relationship between the amount of hearing loss (TTS) and the duration of exposure until maximum hearing loss occurred after 24 hours of exposure. A 10-minute exposure resulted in a 5 dB TTS, whereas a 3-week exposure resulted in a 28 dB TTS that took over 2 weeks to return to pre-exposure baseline levels (Smith et al. 2004b) (Note: recovery time not measured by investigators for shorter exposure durations). However, non of the listed fish species evaluated in this consultation are fish with hearing specializations.

Similarly, Wysocki and Ladich (2005) investigated the influence of noise exposure on the auditory sensitivity of two freshwater fish with notable hearing specializations, the goldfish and the lined Raphael catfish (*Platydoras costatus*), and on a freshwater fish without notable

specializations, the pumpkinseed sunfish (*Lepomis gibbosus*). Baseline thresholds showed greatest hearing sensitivity around 0.5 kHz in the goldfish and catfish and at 0.1 kHz in the sunfish. For the goldfish and catfish, continuous white noise of approximately 130 dB re 1 μPa at 1 m resulted in a significant TTS of 23 to 44 dB. In contrast, the auditory thresholds in the sunfish declined by 7 to 11 dB. The duration of exposure and time to recovery was not addressed in this study. Scholik and Yan (2001) demonstrated TTS in fathead minnows *(Pimephales promelas*) after a 24-hour exposure to white noise (0.3 to 2.0 kHz) at 142 dB re 1 μPa that did not recover as long as 14 days post-exposure.

Studies have also examined the effects of the sound exposures from Surveillance Towed Array Sensor System Low-Frequency Active sonar on fish hearing (Kane et al. 2010; Popper et al. 2007). Hearing was measured both immediately post exposure and for several days thereafter. Maximum received sound pressure levels were 193 dB re 1 μPa for 324 or 628 seconds. Catfish and some specimens of rainbow trout showed 10 to 20 dB of hearing loss immediately after exposure to the low-frequency active sonar when compared to baseline and control animals; however, another group of rainbow trout showed no hearing loss. Recovery in trout took at least 48 hours, but studies were not completed. The different results between rainbow trout groups is difficult to understand, but may be due to developmental or genetic differences in the various groups of fish. Catfish hearing returned to, or close to, normal within about 24 hours after exposure to low-frequency active sonar. Furthermore, examination of the inner ears of the fish during necropsy (note: maximum time fish were held post exposure before sacrifice was 96 hours) revealed no differences from the control groups in ciliary bundles or other features indicative of hearing loss (Kane et al. 2010).

The study of mid-frequency active sonar by the same investigators also examined potential effects on fish hearing and the inner ear (Halvorsen et al. 2012; Kane et al. 2010). Out of the four species tested (rainbow trout, channel catfish, largemouth bass, and yellow perch) only one group of channel catfish, tested in December, showed any hearing loss after exposure to mid-frequency active sonar. The signal consisted of a 2 second (s) long, 2.8 kHz to 3.8 kHz frequency sweep followed by a 3.3 kHz tone of 1 s duration.

The stimulus was repeated five times with a 25 second interval. The maximum received sound pressure level was 210 dB re 1 μPa. These animals, which have the widest hearing range of any of the species tested, experienced approximately 10 dB of threshold shift that recovered within 24 hours. Channel catfish tested in October did not show any hearing loss. The investigators speculated that the difference in hearing loss between catfish groups might have been due to the difference in water temperature of the lake where all of the testing took place (Seneca Lake, New York) between October and December.

Alternatively, the observed hearing loss differences between the two catfish groups might have been due to differences between the two stocks of fish (Halvorsen et al. 2012). Any effects on

hearing in channel catfish due to sound exposure appear to be (Halvorsen et al. 2012; Kane et al. 2010). Investigators observed no damage to ciliary bundles or other features indicative of hearing loss in any of the other fish tested including the catfish tested in October (Kane et al. 2010). Some studies have suggested that there may be some loss of sensory hair cells due to high intensity sources; however, none of these studies concurrently investigated effects on hearing. Enger (1981) found loss of ciliary bundles of the sensory cells in the inner ears of Atlantic cod following 1 to 5 hours of exposure to pure tone sounds between 50 and 400 Hz with a sound pressure level of 180 dB re 1 μPa.

Hastings (1995b) found auditory hair-cell damage in a species with notable anatomical hearing specializations, the goldfish (*Carassius auratus*) exposed to 250 Hz and 500 Hz continuous tones with maximum peak levels of 204 dB re 1 μPa and 197 dB re 1 μPa, respectively, for about 2 hours. Similarly, Hastings et al. (1996) demonstrated damage to some sensory hair cells in oscars (*Astronotus ocellatus*) following a 1-hour exposure to a pure tone at 300 Hz with a peak pressure level of 180 dB re 1 μPa. In none of the studies was the hair cell loss more than a relatively small percent (less than a maximum of 15 percent) of the total sensory hair cells in the hearing organs.

In his sounds exposure guidelines for fishes, Popper et al. (2014b) estimate exposure to low frequency sonar > 193 dB_{rms} re 1 μPa may lead to TTS in bony fish. The authors were not able to estimate a sound exposure level for mid-frequency sonar at which they would expect TTS. As discussed above, studies of the effects of long-duration sounds with sound pressure levels below 170 to 180 dB re 1 μPa indicate that there is little to no effect of long-term exposure on species that lack notable anatomical hearing specialization.

6.4.3.2 *Explosions and Other Impulsive Sound Sources*

Popper et al. (2005) examined the effects of a seismic airgun array on a fish with hearing specializations, the lake chub (*Couesius plumbeus*), and two species that lack notable specializations, the northern pike (*Esox lucius*) and the broad whitefish (*Coregonus nasus*) (a salmonid). In this study the average received exposure levels were a mean peak pressure level of 207 dB re 1 μPa; sound pressure level of 197 dB re 1 μPa; and single-shot sound exposure level of 177 dB re 1 μPa^2-s. The results showed temporary hearing loss for both lake chub and northern pike to both 5 and 20 airgun shots, but not for the broad whitefish. Hearing loss was approximately 20 to 25 dB at some frequencies for both the northern pike and lake chub, and full recovery of hearing took place within 18 hours after sound exposure. Examination of the sensory surfaces of the ears by an expert on fish inner ear structure showed no damage to sensory hair cells in any of the fish from these exposures (Song et al. 2008).

McCauley et al. (2003) showed loss of a small percent of sensory hair cells in the inner ear of the pink snapper (*Pagrus auratus*) exposed to a moving airgun array for 1.5 hours. Maximum received levels exceeded 180 dB re 1 μPa2-s for a few shots. 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. 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 (Lombarte and Popper 1994; Popper and Hoxter 1984) and only a small portion were affected by the sound. The question remains as to why McCauley et al. (2003) found damage to sensory hair cells while Popper et al. (2005) did not. There are many differences between the studies, including species, precise sound source, and spectrum of the sound that it is hard to speculate.

Hastings et al. (2008) exposed the pinecone soldierfish (*Myripristis murdjan*), a fish with anatomical specializations to enhance their hearing; and three species without notable specializations: the blue green damselfish (*Chromis viridis*), the saber squirrelfish (*Sargocentron spiniferum*), and the bluestripe seaperch (*Lutjanus kasmira*) to an airgun array. Fish in cages in 16 ft. (4.9 m) of water were exposed to multiple airgun shots with a cumulative sound exposure level of 190 dB re 1 μPa2-s. The authors found no hearing loss in any fish following exposures.

In his sound exposure guidelines for fishes, Popper et al. (2014b) was not able to develop specific criteria (i.e., sound pressure levels that would result in an impact) for sound exposure from explosions that would lead to TTS or behavioral responses. However, the authors did indicate that that individuals relatively close to the source would have a higher relative risk of being impacted than individuals further away.

6.4.4 Auditory Masking

Auditory masking refers to the presence of a noise that interferes with a fish's ability to hear biologically relevant sounds. Fish use sounds to detect predators and prey, and for schooling, mating, and navigating, among other uses (Myrberg Jr. 1980; Popper et al. 2003). Masking of sounds associated with these behaviors could have impacts to fish by reducing their ability to perform these biological functions.

Any noise (i.e., unwanted or irrelevant sound, often of an anthropogenic nature) detectable by a fish can prevent the fish from hearing biologically important sounds including those produced by prey or predators (Myrberg Jr. 1980; Popper et al. 2003). Auditory masking may take place whenever the noise level heard by a fish exceeds ambient noise levels, the animal's hearing threshold, and the level of a biologically relevant sound. Masking is found among all vertebrate groups, and the auditory system in all vertebrates, including fish, is capable of limiting the effects of masking noise, especially when the frequency range of the noise and biologically relevant signal differ (Fay 1988; Fay and Simmons 1999).

The frequency of the sound is an important consideration for fish because many marine fish are limited to detection of the particle motion component of low frequency sounds at relatively high sound intensities (Amoser and Ladich 2005). The frequency of the acoustic stimuli must first be

compared to the animal's known or suspected hearing sensitivity to establish if the animal can potentially detect the sound.

One of the problems with existing fish auditory masking data is that the bulk of the studies have been done with goldfish, a freshwater fish with well-developed anatomical specializations that enhance hearing abilities. The data on other species are much less extensive. As a result, less is known about masking in marine species, many of which lack the notable anatomical hearing specializations. However, Wysocki and Ladich (2005) suggest that ambient sound regimes may limit acoustic communication and orientation, especially in animals with notable hearing specializations.

Tavolga studied the effects of noise on pure-tone detection in two species without notable anatomical hearing specializations, the pin fish (*Lagodon rhomboids*) and the African mouth-breeder (*Tilapia macrocephala*), and found that the masking effect was generally a linear function of masking level, independent of frequency (Tavolga 1974a; Tavolga 1974b). In addition, Buerkle studied five frequency bandwidths for Atlantic cod in the 20 to 340 Hz region and showed masking across all hearing ranges (Buerkle 1968; Buerkle 1969). Chapman and Hawkins (1973) found that ambient noise at higher sea states in the ocean has masking effects in cod, *Gadus morhua* (L.), haddock, *Melanogrammus aeglefinus* (L.), and pollock, *Pollochinus pollachinus* (L.), and similar results were suggested for several sciaenid species by Ramcharitar and Popper (2004). Thus, based on limited data, it appears that for fish, as for mammals, masking may be most problematic in the frequency region near the signal. There have been a few field studies that may suggest masking could have an impact on wild fish.

Gannon et al. (2005) showed that bottlenose dolphins (*Tursiops truncatus*) move toward acoustic playbacks of the vocalization of Gulf toadfish (*Opsanus beta*). Bottlenose dolphins employ a variety of vocalizations during social communication including low-frequency pops. Toadfish may be able to best detect the low-frequency pops since their hearing is best below 1 kHz, and there is some indication that toadfish have reduced levels of calling when bottlenose dolphins approach (Remage-Healey et al. 2006). Silver perch have also been shown to decrease calls when exposed to playbacks of dolphin whistles mixed with other biological sounds (Luczkovich et al. 2000). Results of the Luczkovich et al. (2000) study, however, must be viewed with caution because it is not clear what sound may have elicited the silver perch response (Ramcharitar et al. 2006). Astrup (1999) and Mann et al. (1998) hypothesized that high frequency detecting species (e.g., clupeids) may have developed sensitivity to high frequency sounds to avoid predation by odontocetes. Therefore, the presence of masking noise may hinder a fish's ability to detect predators and therefore increase predation.

Of considerable concern is that human-generated sounds could mask the ability of fish to use communication sounds, especially when the fish are communicating over some distance. In effect, the masking sound may limit the distance over which fish can communicate, thereby

having an impact on important components of their behavior. For example, the sciaenids, which are primarily inshore species, are one of the most active sound producers among fish, and the sounds produced by males are used to "call" females to breeding sights (Ramcharitar et al. 2001) reviewed in Ramcharitar et al. (2006). If the females are not able to hear the reproductive sounds of the males, there could be a significant impact on the reproductive success of a population of sciaenids. Since most sound production in fish used for communication is generally below 500 Hz (Slabbekoorn et al. 2010), sources with significant low-frequency acoustic energy could affect communication in fish.

Also potentially vulnerable to masking is navigation by larval fish, although the data to support such an idea are still exceedingly limited. There is indication that larvae of some reef fish (species not identified in study) may have the potential to navigate to juvenile and adult habitat by listening for sounds emitted from a reef (either due to animal sounds or non-biological sources such as surf action)(e.g., (Higgs 2005)).

In a study of an Australian reef system, the sound signature emitted from fish choruses was between 0.8 and 1.6 kHz (Cato 1978) and could be detected by hydrophones 3 to 4 nm from the reef (McCauley and Cato 2000). This bandwidth is within the detectable bandwidth of adults and larvae of the few species of reef fish, such as the damselfish, *Pomacentrus partitus*, and bicolor damselfish, *Eupomacentrus partitus*, that have been studied (Kenyon 1996; Myrberg Jr. 1980). At the same time, it has not been demonstrated conclusively that sound, or sound alone, is an attractant of larval fish to a reef, and the number of species tested has been very limited. Moreover, there is also evidence that larval fish may be using other kinds of sensory cues, such as chemical signals, instead of, or alongside of, sound (Atema et al. 2002).

Popper et al. (2014b) evaluated the potential for masking in fishes and concluded there is no reason to expect masking from explosions because while the detection of biologically important sounds would be reduced, this effect would only occur during the brief duration of the sound. The authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, it would only occur during the sonar transmissions and would result in a narrow range of frequencies being masked (Popper et al. 2014b).

6.4.5 Behavioral Reactions

There are little data available on the behavioral reactions of fish, and almost no research conducted on any long-term behavioral effects or the potential cumulative effects from repeated exposures to loud sounds (Popper and Hastings 2009a; Popper and Hastings 2009b). Behavioral effects to fish could include disruption or alteration of natural activities such as swimming, schooling, feeding, breeding, and migrating. Sudden changes in sound level can cause fish to dive, rise, or change swimming direction. There is a lack of studies that have investigated the behavioral reactions of unrestrained fish to anthropogenic sound, especially in the natural

environment. Studies of caged fish have identified three basic behavioral reactions to sound: startle, alarm, and avoidance (McCauley et al. 2000; Pearson et al. 1992). Changes in sound intensity may be more important to a fish's behavior than the maximum sound level. Sounds that fluctuate in level tend to elicit stronger responses from fish than even stronger sounds with a continuous level (Schwartz 1985).

6.4.5.1 *Non-impulsive Sound Sources*

Gearin et al. (2000) studied responses of adult sockeye salmon (*Oncorhynchus nerka*) and sturgeon (*Acipenser* sp.) to pinger sounds produced by acoustic devices designed to deter marine mammals from gillnet fisheries. The pingers produced sounds with broadband energy with peaks at 2 kHz or 20 kHz. They found that fish did not exhibit any reaction or behavior change to the pingers, which demonstrated that the alarm was either inaudible to the salmon and sturgeon, or that neither species was disturbed by the mid-frequency sound (Gearin et al. 2000). Based on hearing threshold data, it is highly likely that the salmonids did not hear the sounds.

Culik et al. (2001) did a very limited number of experiments to determine the catch rate of herring in the presence of pingers producing sounds that overlapped with the frequency range of hearing for herring (2.7 kHz to over 160 kHz). They found no change in catch rates in gill nets with or without the higher frequency (greater than 20 kHz) sounds present, although there was an increase in the catch rate with the signals from 2.7 kHz to 19 kHz (a different source than the higher frequency source). The results could mean that the fish did not "pay attention" to the higher frequency sound or that they did not hear it, but that lower frequency sounds may be attractive to fish. At the same time, it should be noted that there were no behavioral observations on the fish, and so how the fish actually responded when they detected the sound is not known.

Doksaeter et al. (2009) studied the reactions of wild, overwintering herring to Royal Netherlands Navy experimental mid-frequency active sonar and killer whale feeding sounds. The behavior of the fish was monitored using upward looking echosounders. The received levels from the 1 to2 kHz and 6 to 7 kHz sonar signals ranged from 127 to 197 dB re 1 μPa and 139 to 209 dB re 1 μPa, respectively. Escape reactions were not observed upon the presentation of the mid-frequency active sonar signals; however, the playback of the killer whale sounds elicited an avoidance reaction. The authors concluded that mid-frequency sonar could be used in areas of overwintering herring without substantially affecting the fish.

Studies documenting behavioral responses of fish to vessels show that Barents Sea capelin (*Mallotus villosus*) may exhibit avoidance responses to engine noise, sonar, depth finders, and fish finders (Jørgensen et al. 2004). Avoidance reactions are quite variable depending on the type of fish, its life history stage, behavior, time of day, and the sound propagation characteristics of the water (Schwartz 1985). Misund (1997) found that fish ahead of a ship that showed avoidance reactions did so at ranges of 160 to 490 ft. (48.8 to 149.4 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.

In a study by Chapman and Hawkins (1973) the low-frequency sounds of large vessels or accelerating small vessels caused avoidance responses by herring. Avoidance ended within 10 seconds after the vessel departed. Twenty-five percent of the fish groups habituated to the sound of the large vessel and 75 percent of the responsive fish groups habituated to the sound of small boats.

Recently, Neo et al. (2015) studied behavioral changes of captive zebrafish in response to sound exposure (both continuous and intermittent). The authors allowed fish to influence exposure to higher sound levels by swimming freely between an articifially noisy tank and another with ambient noise levels. They observed that despite initial startle responses to elevated sound levels, there was no long-term behavioral reactions such as a preference for one tank or the other.

Popper et al. (2014b) concluded that the relative risk of a fish eliciting a behavioral reaction in response to low-frequency sonar was low, regardless of the distance from the sound source. The authors expected a limited number of species may respond to mid-frequency sonar since most fish do not have specializations that enable them to hear above 2,500 Hz (Halvorsen et al. 2012; Popper et al. 2014b). No ESA-listed fish species in the Gulf of Alaska have these specializations.

6.4.5.2 *Explosions and Other Impulsive Sound Sources*

Pearson et al. (1992) exposed several species of rockfish (*Sebastes spp.*) to a seismic airgun. The investigators placed the rockfish in field enclosures and observed the fish's behavior while firing the airgun at various distances for 10 minute trials. Dependent upon the species, rockfish exhibited startle or alarm reactions between peak to peak sound pressure level of 180 dB re 1 μPa and 205 dB re 1 μPa. The authors reported the general sound level where behavioral alterations became evident was at about 161 dB re 1 μPa for all species. During all of the observations, the initial behavioral responses only lasted for a few minutes, ceasing before the end of the 10-minute trial.

Similarly, Skalski et al. (1992) showed a 52 percent decrease in rockfish (*Sebastes* sp.) caught with hook-and-line (as part of the study—fisheries independent) when the area of catch was exposed to a single airgun emission at 186 to 191 dB re 1 μPa (mean peak level) (See also (Pearson et al. 1987; Pearson et al. 1992)). They also demonstrated that fish would show a startle response to sounds as low as 160 dB re 1 μPa, but this level of sound did not appear to elicit decline in catch. Wright (1982) also observed changes in fish behavior as a result of the sound produced by an explosion, with effects intensified in areas of hard substrate.

Wardle et al. (2001) used a video system to examine the behaviors of fish and invertebrates on reefs in response to emissions from seismic airguns. The researchers carefully calibrated the airguns to have a peak level of 210 dB re 1 μPa at 16 m and 195 dB re 1 μPa at 109 m from the source. There was no indication of any observed damage to the marine organisms. They found no substantial or permanent changes in the behavior of the fish or invertebrates on the reef throughout the course of the study, and no marine organisms appeared to leave the reef.

Engås et al. (1996) and Engås and Løkkeborg (2002) examined movement of fish during and after a seismic airgun study by measuring catch rates of haddock (*Melanogrammus aeglefinus*) and Atlantic cod as an indicator of fish behavior using both trawls and long-lines as part of the experiment. These investigators found a significant decline in catch of both species that lasted for several days after termination of airgun use. Catch rate subsequently returned to normal. The conclusion reached by the investigators was that the decline in catch rate resulted from the fish moving away from the airgun sounds at the fishing site. However, the investigators did not actually observe behavior, and it is possible that the fish just changed depth.

The same research group showed, more recently, parallel results for several additional pelagic species including blue whiting and Norwegian spring spawning herring (Slotte et al. 2004). However, unlike earlier studies from this group, the researchers used fishing sonar to observe behavior of the local fish schools. They reported that fish in the area of the airguns appeared to go to greater depths after the airgun exposure compared to their vertical position prior to the airgun usage. Moreover, the abundance of animals 18 to 31 miles (29 to 50 km) away from the ensonification increased, suggesting that migrating fish would not enter the zone of seismic activity.

Alteration in natural behavior patterns due to exposure to impulsive noise (such as pile driving and explosions) has not been well studied. However, one study (Mueller-Blenkle et al. 2010), which took place with fish enclosed in a mesocosm (an enclosure providing a limited body of water with close to natural conditions), demonstrated behavioral reactions of cod and Dover sole (*Solea solea*) to impulsive sounds from pile driving. Sole showed a significant increase in swimming speed. Cod reacted, but not significantly, and both species showed directed movement away from the sources with signs of habituation after multiple exposures. For sole, reactions were seen with peak sound pressure levels of 144 to 156 dB re 1 μPa; and cod showed altered behavior at peak sound pressure levels of 140 to 161 dB re 1 μPa. For both species, this corresponds to a peak particle motion between 6.51×10^{-3} and 8.62×10^{-4} meters per second squared (m/s^2).

Popper et al. (2014b) indicated very little is known about the effects of explosions of wild fish behavior, but suggested that startle responses, likely lasting less than a second, could occur from exposure to explosives and that such responses would not necessarily result in significant changes to subsequent behavior.

6.4.6 Repeated Exposures of Fish

As noted previously, there is almost no research on any long-term behavioral effects or the potential cumulative effects from repeated exposures of fish to loud sounds (Popper and Hastings 2009a; Popper and Hastings 2009b). Popper et al. (2014b) identify this knowledge gap as a high priority for future research.

For the proposed action, repeated exposure to acoustic stressors that could adversely affect ESA-listed fish species are possible if successive detonations occur in close proximity to one another. However, with the exception of those individuals that are close enough to be killed or seriously injured, we would expect repeated exposures under this scenario to not impact the fitness of individual fish. As suggested by Popper et al. (2014b), most responses to detonations are expected to be temporary startle responses with the animal resuming normal activity shortly after exposure. The same would be expected if multiple startle responses occur resulting from successive detonations. As noted previously, fish may also experience hearing loss if an individual is in close proximity to an underwater detonation. However, hearing loss would be temporary because unlike marine mammals, fish are capable of regenerating sensory hairs and no permanent hearing loss has ever been reported in fish (Lombarte et al. 1993; Smith et al. 2006b). Recovery from hearing loss would be expected whether the individual was exposed to one, or multiple detonations. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these responses.

Multiple stressors may also have synergistic effects. For example, fish that experience injury from acoustic stressors could be more susceptible to physical strike and disturbance stressors via a decreased ability to detect and avoid threats. These interactions are speculative, and without data on the combination of multiple Navy stressors, the synergistic impacts from the combination of Navy stressors are difficult to predict in any meaningful way.

6.4.7 Criteria for Predicting Acoustic and Explosive Impacts - Fish

Navy activities introduce sound or explosive energy into the marine environment, therefore, NMFS must conduct an analysis of potential effects to ESA-listed fish species. To do this, information about the numerical sound and energy levels that are likely to elicit certain types of physiological and behavioral reactions is needed. The following contains information on the criteria, thresholds, and methodology for quantifying impacts from acoustic and explosive sources, which were jointly developed between the Navy, NMFS, and the U.S. Fish and Wildlife Service in 2015. For additional information on the methodology used to develop these criteria, see Renken (2015).

6.4.7.1 *Non-impulsive Acoustic Sources (Sonar)*

As we discussed in Section 3.1.5 of this opinion, potential direct injuries to fish from non-impulsive sound sources, such as sonar, are unlikely because of the relatively lower peak pressures and slower rise times than potentially injurious sources such as explosives. Non-impulsive sources also lack the strong shock wave such as that associated with an explosion. However, if a sonar source level is high enough and an individual fish is close enough to the source, physiological injury may be possible. As such, we established thresholds (Table 29) to assess potential auditory effects such as TTS, recoverable injury, and mortal injury and mortality (i.e., onset mortality) to assess these potential effects. Table 30 provides the estimated range to effects based on these thresholds. As anticipated, these ranges are very small for the most powerful sonar sources in each category.

Table 29. Thresholds for Assessing Effects to Fish from Sonar. All values are SEL (dB re 1 µPa^2-s)

Fish Hearing Group	Effect				
	Sonar Frequency Band	Mortality and Mortal Injury[1]	Recoverable Injury[1]	TTS	Behavioral Harassment
No Swim Bladder	Low Frequency	>>218	>218	218	Qualitative Assessment[2]
	Mid-Frequency	>>221	>221	CH[3]	CH
Swim Bladder (Not Used Involved in Hearing)	Low Frequency	>>218	>218	210	Qualitative Assessment[2]
	Mid-Frequency	>>221	>221	CH	CH
Swim Bladder (Involved in Hearing)	Low Frequency	>>218	>218	210	197
	Mid-Frequency	>>221	>221	220	200

[1] The values presented for mortality, mortal injury, and onset injury represent the highest sound exposure levels which have been tested to date, none of which have resulted in mortality or injury for fish with swim bladders not involved in hearing or fish without swim bladders.

[2]According to Popper et al. (2014b) there is a low probability of a behavioral reaction to low- and mid-frequency sonar at any range (from the source outward) by fishes with a swim bladder that is not involved in hearing (e.g. salmon, steelhead).

[3]Cannot Hear

Table 30. Predicted Range to Effect for Sonar Source bins used in Gulf of Alaska TMAA. (Range is maximum expected range.) All distances are in meters. A value of "0" indicates that the source level is below the criteria threshold even after accumulation of multiple pings.

Sonar Bin	No Swim Bladder			Swim Bladder (Not involved in hearing)			Swim Bladder (Involved in hearing)			
	Mortality and Injury	Recoverable Injury	TTS	Mortality and Injury	Recoverable Injury	TTS	Mortality and Injury	Recoverable Injury	TTS	Behavioral Harassment
MF1	<<12	<12	CH	<<12	<12	CH	<<12	<12	14	138
MF3	<<2	<2	CH	<<2	<2	CH	<<2	<2	2	24
MF4	0	0	CH	0	0	CH	0	0	0	8
MF5	0	0	CH	0	0	CH	0	0	0	0
MF6	0	0	CH	0	0	CH	0	0	0	0
MF11	<<6	<6	CH	<<6	<6	CH	<<6	<6	7	69
ASW2	0	0	0	0	0	1	0	0	1	6
ASW4	<<1	<1	CH	<<1	<1	CH	<<1	<1	1	15

-CH = cannot hear

-For mortality and recoverable injury the effect occurs at a distance either much less than the number provided or less than the number provided, respectively.

6.4.7.2 *Explosives*

The use of explosives in the Gulf of Alaska TMAA have the potential to directly kill or injure ESA-listed fish species. Therefore, the Navy used the threshold criteria to determine the distance from an explosion of a specified net explosive weight that could injure or kill a fish. Ranges to effects were species-specific and varied with fish size (Table 31). Ranges to effects are not presented for some explosive bins where the species or life-stage considered is not expected to co-occur in space or time. Ranges to effects are presented in meters.

Table 31. Ranges to Effects to Fish from Explosives. Ranges to effects are presented in meters.

Explosive BIN	Representative depth of charge[1]		Chinook		Coho		Chum		Sockeye		Steelhead	
		Life Stage	Juveniles	Adult	Juveniles	Adult	Juveniles	Adult	Juveniles	Adult	Juveniles	Adults
		Weight (g)	NA	3,020	14.91	1,468	3.87	399	15.09	440	31	355
E5	1	1% Mort	140	71	136	75	160	89	136	88	124	90
		Onset injury	231	133	228	137	263	155	288	153	208	157
E9	1	1% Mort	NA	NA	NA	182	NA	215	NA	212	298	218
		Onset injury	NA	NA	NA	304	NA	354	NA	250	487	360
E10	1	1% Mort	NA	NA	NA	217	NA	259	NA	256	361	263
		Onset injury	NA	NA	NA	356	NA	419	NA	414	582	426
E12	1	1% Mort	NA	NA	NA	263	NA	314	NA	310	435	319
		Onset injury	NA	NA	NA	427	NA	506	NA	500	703	514

[1]Energy loss into air for surface detonations is considered. See Swisdak, M. (1978). Explosion effects and properties, Part II: Explosion effects in water. Naval Surface Warfare Center/ White Oak Laboratory (NSWC/WOL) Technical Report TR 76-116.
[2]Range to effects were not presented where there is not expected to be co-occurrence between the explosive bin and the species/life stage.

6.5 Density Estimate Use in Exposure Estimates

The following sections discuss the density estimates used in the effects analyses.

6.5.1 Marine Mammals

There is no single source of density data for every area, species, and season because of the fiscal costs, resources, and effort involved to provide enough survey coverage to sufficiently estimate density. Therefore, to characterize marine mammal density for areas of concern such as the action area, the Navy compiled data from multiple sources. Each data source may use different methods to estimate density, of which, uncertainty in the estimate can be directly related to the method applied.

The Navy thus developed a protocol to select the best available data sources based on species, area, and time (season). The Navy then used this protocol to identify the best available density data from available sources, including habitat-based density models, line-transect analyses, and peer-reviewed published studies. During the development of these density data sets, the Navy worked with scientists affilitated with the NMFS to validate the final selections. These data were incorporated into a Geographic Information System database that includes density values for every marine mammal species present within the action area. Detailed information on the Navy's

selection protocol, datasets, and specific density values are provided in a Pacific Navy Marine Species Density Database Technical Report (DoN 2014).

A quantitative impact analysis requires an estimate of the number of animals that might be affected. A key element of this estimation is knowledge of the abundance and concentration of the species in specific areas where those activities will occur. The most appropriate metric for this type of analysis is density or the number of animals present per unit area. Marine species density estimation requires a significant amount of effort to both collect and analyze data to produce a reasonable estimate. Unlike surveys for terrestrial wildlife, many marine species spend much of their time submerged, and are not easily observed. In order to collect enough sighting data to make reasonable density estimates, multiple observations are required, often in areas that are not easily accessible (e.g., far offshore). Ideally, marine species sighting data would be collected for the specific area and time period of interest and density estimates derived accordingly. However, in many places poor weather conditions and high sea states prohibit the completion of comprehensive surveys.

For most cetacean species, abundance is estimated using line-transect surveys or mark-recapture studies.(Barlow 2010b; Barlow and Forney 2007; Calambokidis et al. 2008). The result provides one single density estimate value, for each species, across broad geographic areas, such as waters within the U.S. Exclusive Economic Zone in the Gulf of Alaska. This is the general approach applied in estimating cetacean abundance in the NMFS stock assessment reports. Though the single value provides a good average estimate of abundance (total number of individuals) for a specified area, it does not provide information on the species distribution or concentrations within that area, and does not estimate density for other timeframes/seasons that were not surveyed. More recently, habitat modeling has been used to estimate cetacean densities (Barlow et al. 2009; Becker et al. 2012a; Becker et al. 2012b; Becker et al. 2010; Becker et al. 2012c; Ferguson et al. 2006; Forney et al. 2012; Redfern et al. 2006). These models estimate cetacean density as a continuous function of habitat variables (e.g., sea surface temperature, seafloor depth, etc.) and thus allow predictions of cetacean densities on finer spatial scales than traditional line-transect or mark-recapture analyses. Within the action area that was modeled, densities can be predicted wherever these habitat variables can be measured or estimated.

Currently-published density estimates rely on low numbers of sightings available for their derivation. This can lead to uncertainty which is typically expressed by the coefficient of variation of the estimate, which is derived using standard statistical methods and describes the amount of variation with respect to the population mean. It is expressed as a fraction or sometimes a percentage and can range upward from zero, indicating no uncertainty, to high values. For example, a coefficient of variation of 0.85 would indicate high uncertainty in the population estimate. When the coefficient of variation exceeds 1.0, the estimate is very uncertain. The uncertainty associated with movements of animals into or out of an area (due to factors such

as availability of prey or changing oceanographic conditions) is much larger than is indicated by the coefficient of variation.

The methods used to estimate pinniped at-sea densities are typically different than those used for cetaceans. Pinniped abundance is generally estimated via shore counts of animals at known rookeries and haul-out sites. Translating these numbers to in-water densities is difficult given the variability in foraging ranges, migration, and haul-out behavior between species and within each species, and is driven by factors such as age class, sex class, seasonal variation, etc.

6.5.2 Fish

The methodologies used to estimate densities of ESA-listed fish in the action area are described in section 6.7.2.

6.6 Exposure and Response to Non-Impulsive Acoustic Stressors During Training Activities

The following sections discuss our analysis of stressors that are likely to adversely affect ESA-listed species. If a stressor is likely to adversely affect any of the ESA-listed species in the action area, it is discussed further in this Section. Exposure of fish to acoustic stressors was not modeled by the Navy due to limited information on species distribution and density in the action area. We conducted a separate analysis on the effects to fish.

6.6.1 Exposure and Response of Marine Mammals to Non-Impulsive Acoustic Stressors

For this consultation, we considered exposure estimates from the Phase II NAEMO model at several output points for marine mammals. First, we estimated the total number of ESA-listed species (animats) that would be exposed to acoustic sources prior to the application of a dose-response curve or criteria. We term these the "unprocessed" estimates. This estimate is the number of times individual animats or animals are likely to be exposed to the acoustic environment that is a result of training activities, regardless of whether they are "taken" as a result of that exposure. In most cases, the number of animals "taken" by an action would be a subset of the number of animals that are exposed to the action because (1) in some circumstances, animals might not respond to an exposure or (2) some responses may be negative for an individual animal without constituting a form of "take" (for example, some physiological stress responses only have fitness consequences when they are sustained and would only constitute a "take" as a result of cumulative exposure). A second set of exposure estimates ("model-estimated") of listed species were generated and "processed" using dose-response curves and criteria for temporary and permanent threshold shift developed by the Navy and NMFS' Permits Division for the purpose of identifying harassment pursuant to the MMPA.

Any modeled instances of injury and mortality are further analyzed to account for standard mitigation actions that NMFS's Permits Division requires under the MMPA rule and LOA to avoid marine mammals and avoidance responses that would be expected from individual animals once they sense the presence of Navy vessels or aircraft (post-processing; see Section 3.1.2 for further detail). Mitigation measures are effective at reducing instances of injury or mortality, but would not further reduce potential behavioral impacts to lesser impacts due to the potential distance from the source stressor. Consideration of avoidance and mitigation only reduces some "Level A" (potential to injure or kill) impacts to "Level B" impacts; "Level B" impacts are not reduced in the post-processing stage. The final take estimates are the result of the acoustic analysis, including acoustic effects analysis, followed by consideration of animal avoidance of multiple exposures and Navy mitigation measures. Since these final estimates represent incidental take for purposes of the ESA, we base our jeopardy analyses and determinations on these estimates.

6.6.1.1 ***Blue Whale***

The NAEMO output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The model estimates that some mysticete behavioral reactions could occur at great distances (e.g., exceeding 100 km), though masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b; Baumann-Pickering et al. 2012b; Debich et al. 2013; Debich et al. 2014; Rice et al. 2015) or other potential biological sources (D'Spain and Batchelor 2006), make reaction to the sound from sonar and other active sound sources by mysticetes at these distances less likely. Therefore, the number of exposures that may result in a take is a small subset of the total estimated exposure instances.

Based on the Navy's exposure models, we would expect approximately 1,101 possible instances annually in which a blue whale belonging to the Eastern North Pacific stock might be exposed to active sonar associated with training activities above 120 dB SPL. Of those unprocessed exposures, there were 38 model-estimated instances where a blue whale might exhibit a behavioral response as a result of that exposure, and 9 model estimated instances of TTS. There were no model-estimated exposures leading to injury, including PTS, or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates include 38 instances where a blue whale would be expected to exhibit a behavioral response and 9 instances of TTS). This information is summarized in Table 32 below.

<u>Temporary Threshold Shift -</u> TTS can last from a few minutes to days, be of varying degree, and occur across various frequency bandwidths, all of which determine the severity of the impacts on the affected individual, which can range from minor to more severe. The TTS sustained by an animal is primarily classified by three characteristics:

1. Frequency—Available data (of mid-frequency hearing specialists exposed to mid- or high-frequency sounds; (Southall et al. 2007b)) suggest that most TTS occurs in the frequency range of the source up to one octave higher than the source (with the maximum TTS at ½ octave above). The more powerful mid-frequency sources used have center frequencies between 3.5 kHz and 8 kHz and the other unidentified mid-frequency sources are, by definition, less than 10 kHz, which suggests that TTS induced by any of these mid frequency sources would be in a frequency band somewhere between approximately 2 and 20 kHz. As discussed above, blue whales are low-frequency hearing specialists, producing low-frequency vocalizations that include moans in the range from 12.5 Hz to 400 Hz, with dominant frequencies from 16 Hz to 25 Hz. Therefore, even though recent studies (discussed below) indicate that blue whales hear and respond to mid-frequency sounds, it is unlikely that TTS caused by mid-frequency active sonar transmissions would interfere with an animal's ability to hear vocalizations or most other biologically important sounds. There are fewer hours of high frequency source use and the sounds would attenuate more quickly, plus they have lower source levels, but if an animal were to incur TTS from these sources, it would cover a higher frequency range (sources are between 20 kHz and 100 kHz, which means that TTS could range up to 200 kHz; however, high frequency systems are typically used less frequently and for shorter time periods than surface ship and aircraft mid-frequency systems, so TTS from these sources is even less likely).

2. Degree of the shift (i.e., how many dB is the sensitivity of the hearing reduced by)—Generally, both the degree of TTS and the duration of TTS will be greater if the marine mammal is exposed to a higher level of energy (which would occur when the peak dB level is higher or the duration is longer). The threshold for the onset of TTS was discussed previously in this document. An animal would have to approach closer to the source or remain in the vicinity of the sound source appreciably longer to increase the received SEL, which would be difficult considering the lookouts, the sightability of blue whales and other listed mysticetes, and the nominal speed of an active sonar vessel (10 to 15 knots). In the TTS studies, some using exposures of almost an hour in duration or up to 217 SEL, most of the TTS induced was 15 dB or less, though Finneran et al. (2007) induced 43 dB of TTS with a 64-second exposure to a 20 kHz source. However, mid-frequency active sonar emits a nominal ping every 50 seconds, and incurring those levels of TTS is highly unlikely.

3. Duration of TTS (recovery time)—In the TTS laboratory studies, some using exposures of almost an hour in duration or up to 217 SEL, almost all individuals recovered within 1 day (or less, often in minutes), though in one study (Finneran et al. 2007), recovery took 4 days. Based on the range of degree and duration of TTS reportedly induced by exposures to non-pulse sounds of energy higher than that to which freeswimming marine mammals in the field are likely to be exposed during mid-frequency/high-frequency active sonar training activities in the action area, it is unlikely that blue whales or other listed cetaceans would ever sustain a TTS from MFAS that alters their sensitivity by more than 20 dB for more than a few days (and any incident of TTS would likely be far less severe due to the short duration of the majority of the exercises and the speed of a typical vessel). Also, because of the short distance within which animals would need to

approach the sound source, it is unlikely that animals would be exposed to the levels necessary to induce TTS in subsequent time periods such that their recovery is impeded.

In addition, the mitigation zones prescribed in the final MMPA rule encompass the ranges to PTS for the most sensitive marine mammal functional hearing group, which is usually the high-frequency cetacean hearing group. Therefore, the mitigation zones are even more protective for blue whales and other low-frequency cetaceans, and likely cover a larger portion of the potential range to onset of TTS. Accordingly, the prescribed mitigation and the sightability of blue whales and other mysticetes (due to their large size) reduce the potential for a significant TTS or behavioral reaction threshold shift to occur.

Therefore, for blue whales and other listed mysticetes analyzed in this opinion, even though the modeled TTS estimates may change from year to year over baseline conditions, NMFS does not anticipate TTS of a long duration or severe degree to occur as a result of exposure to mid- or high-frequency active sonar utilized in the Gulf of Alaska TMAA.

The classification of modeled effects from acoustic stressors, such as TTS, are performed in a manner as to conservatively overestimate the impacts of those effects. Acoustic stressors are binned and all stressors within each bin are modeled as the loudest source, overestimating impacts within each bin. Therefore, the temporary duration of many TTS exposures may be on the shorter end of the range and last only a few minutes. Even longer duration TTS are expected to typically last hours. Additionally, though the frequency range of TTS that marine mammals might sustain would overlap with some of the frequency ranges of their vocalization types, the frequency range of TTS from mid-frequency active sonar (the source from which TTS would most likely be sustained because the higher source level and slower attenuation make it more likely that an animal would be exposed to a higher received level) would not usually span the entire frequency range of one vocalization type, much less span all types of vocalizations or other critical auditory cues. If impaired, marine mammals would typically be aware of their impairment and would likely be able to implement behaviors to compensate. Given the brief amount of time blue whales are expected to experience TTS, it is unlikely to significantly impair their ability to communicate, forage, or breed and will not have fitness level consequences at the individual or population level. Navy monitoring of Navy-wide activities since 2006 has documented hundreds of thousands of marine mammals on the range complexes and there are only two instances of overt behavioral change that have been observed and there have been no demonstrable instances of injury to marine mammals as a result of non-impulsive acoustic sources.

Behavioral Responses – The Navy uses the behavioral response function to quantify the number of behavioral responses that would qualify as Level B behavioral harassment under the MMPA. As the statutory definition is currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or

temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated using the behavioral response function do not differentiate between the different types of potential reactions. Nor do the estimates provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. We therefore consider the available scientific evidence to determine the likely nature of the modeled blue whale behavioral responses and the potential fitness consequences for affected individuals.

While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on blue whale hearing so we assume that blue whale vocalizations are partially representative of their hearing sensitivities. Blue whales are not likely to respond to high-frequency sound sources associated with the proposed training activities and training activities because of their hearing sensitivities. Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007b; Stafford and Moore 2005a), recent research has indicated blue whales not only hear mid-frequency active sonar transmissions, in some cases they respond to those transmissions (Goldbogen et al. 2013; Melcon et al. 2012; Southall et al. 2011a).

As discussed previously in this opinion, blue whales hear some sounds in the mid-frequency range and may exhibit behavioral responses to sounds in this range depending on received level and context (Goldbogen et al. 2013; Melcon et al. 2012). However, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational MFA sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). As stated in Goldbogen et al. (2013) when summarizing the response of blue whales to simulated MFA sonar, "We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure."

Melcon et al. (2012) reported that blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low frequency calls (D calls) usually associated with feeding behavior. However, they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior and indicated that implications of the documented responses are unknown. Goldbogen et al. (2013) speculated that if the documented temporary behavioral responses interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case,

particularly since unconsumed prey would likely still be available in the environment in most cases following the cessation of acoustic exposure. Additionally, it is worth noting here that most instances of exposure would be brief.

Moreover, there are important differences between the acoustic sources used in the authors' Controlled Exposure Experiments. As discussed previously, perhaps the most significant response documented in the study occurred during a controlled exposure experiment involving pseudo-random noise rather than simulated sonar, which corresponded with a blue whale terminating a foraging bout. The more significant reaction to pseudo-random noise may be indicative of habituation to mid-frequency sonar signals; the authors noted that the responses they documented were in a geographical region with a high level of naval activity and where mid-frequency sonar use is common. In addition, during the controlled exposure experiments, sound sources were deployed from a stationary source vessel positioned approximately 1,000 m from the focal animals, with one transmission onset every 25 seconds (Southall et al. 2012a). In contrast, most Navy sonar systems are deployed from highly mobile vessels or in-water devices which do not directly target marine mammals. Further, the typical duty cycle with most tactical anti-submarine warfare is lower than used in the controlled exposure experiments described above, transmitting about once per minute with most active sonar pulses lasting no more than a few seconds (Navy 2013). Moreover, a typical Navy vessel with hull mounted MFA sonar would travel over 300 meters between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. In the event an individual is exposed to multiple sound sources that elicit a behavioral response (e.g., disruption of feeding) in a short amount of time, we do not expect these exposures to have fitness consequences as individuals will resume feeding upon cessation of the sound exposure and unconsumed prey will still be available in the environment.

As noted in (Southall et al. 2007b), substantive behavioral reactions to noise exposure (such as disruption of critical life functions, displacement, or avoidance of important habitat) are considered more likely to be significant if they last more than 24 hours, or recur on subsequent days. Major training activities conducted in the Gulf of Alaska TMAA can last several weeks, and during those exercises there may be periods of continuous sonar use. Not every major training exercise has anti-submarine warfare events where sonar is used. However, even the longest periods of "continuous" active sonar use rarely last longer than 12 hours, and active sonar use is not truly continuous because a sonar system is actively transmitting a small portion of the time (once per minute for approximately 10 seconds). For Navy active sonar use, a period of concentrated, near continuous anti-submarine warfare sonar use means that sound energy is being put in the water nearly 2 percent of the time. Sonar sound is not transmitting when trying to listen for returns of a detection of a submarine or contact of something else in the water column. Vessels equipped with the most powerful sonar systems would also generally be moving

at speeds of 10 to 15 knots separated in distances when using active sonar. Thus, even during major training activities, it is unlikely that a specific geographic area or water column would be ensonified at high levels for prolonged periods of time. Even if sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks during major training activities), we would expect that some individual whales would most likely respond by temporarily avoiding areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. However, given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, even temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect similar foraging to be available in close proximity. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

On a related point, for purposes of this opinion, we assume that the Navy's activities and associated impacts will continue into the reasonably foreseeable future at the annual levels discussed herein. This raises the question of whether the annual instances of modeled TTS or behavioral disturbances could indirectly lead to more serious aggregate or long-term impacts over time. Under certain conditions, chronic exposure to acoustic sources or other stimuli that can cause individual stress or behavioral responses can also lead to additional long-term adverse impacts. For example, investigators concluded that gray whales and humpback whales abandoned some of their coastal habitat in California and Hawaii, respectively, because of persistent underwater noise associated with extensive vessel traffic (Gard 1974; Reeves 1977; Salden 1988). Another study of terrestrial mammals suggests that while short-term stress responses are often beneficial, conditions of chronic or long-term stress can lead to adverse physiological effects (Romero and Butler 2007). However, because acoustic disturbances caused by Navy sonar and explosives are short-term, intermittent, and (in the case of sonar) transitory, even during major training activities, we would not expect the Navy's activities to create conditions of long-term, continuous underwater noise leading to habitat abandonment or long-term hormonal or physiological stress responses in marine mammals. Moreover, as discussed previously, Navy training and training activities in the Gulf of Alaska have included the use of MFA sonar since 2011, and explosives for decades. There is no evidence that the activities have caused blue whales or other listed species evaluated in this opinion to abandon important habitat or any other severe adverse impacts.

Further, recent evidence indicates that the Eastern North Pacific blue whale population, which includes blue whales in Southern California and the Pacific Northwest, has likely reached carrying capacity (Monnahan et al. 2014a). As stated previously, Navy training activities in these areas (i.e., SOCAL and NWTT), include the use of MFA sonar and have been ongoing for

decades. Therefore, any potential temporary behavioral impacts on blue whales from the use of MFA during Navy training activities in these areas likely have not prevented the recovery of the blue whales throughout its range.

Establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013; Read et al. 2014). It is difficult to assess the effects of sounds individually and cumulatively on marine species because a number of factors can influence these effects including: indirect effects, age class, prior experience, behavioral state at the time of exposure, and that responses may be influenced by other non-sound related factors (Ellison et al. 2011; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014; Williams et al. 2014a). However, although there is significant uncertainty, based upon the available evidence and the foregoing analysis, we conclude that continuation of annual modeled instances of TTS and behavioral disturbances into the reasonably foreseeable future is unlikely to cause aggregate or long-term adverse effects on blue whales, such as abandonment of important habitat or adverse physiological effects resulting from chronic or long-term stress.

Non-impulsive acoustic stressors from Navy training activities conducted during five-year period of proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of blue whale populations. We anticipate temporary behavioral responses and TTS, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. This conclusion is further supported by Monnahan et al. (2014a) which concluded that the Eastern North Pacific blue whale population has likely reached carrying capacity despite years of Navy training activities, including MFA sonar, in the eastern Pacific (including off the coasts of Washington, Oregon, and California, and in the Gulf of Alaska).

6.6.1.2 *North Pacific Right Whale*

North Pacific right whales could occur in all portions of the action area. The NAEMO output estimates that North Pacific right whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The model estimates that some mysticete behavioral reactions could occur at great distances (e.g., exceeding 100 km), though masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b; Baumann-Pickering et al. 2012b; Debich et al. 2013; Debich et al. 2014; Rice et al. 2015) or other potential biological sources (D'Spain and Batchelor 2006), make reaction to the sound from sonar and other active sound sources by mysticetes at these distances less likely. Therefore, the number of exposures that may result in a take is a small subset of the total estimated exposure instances.

Based on the Navy's exposure models, we would expect approximately 24 instances a year in which a North Pacific right whale belonging to the Eastern North Pacific stock might be exposed to active sonar associated with training activities above 120 dB SPL. Of those unprocessed exposures, there were two model-estimated instances where a North Pacific right whale might exhibit a behavioral response as a result of that exposure, and one model estimated instance of TTS. There were no model-estimated exposures leading to injury, including PTS, or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates include two instances where a North Pacific right whale would be expected to exhibit a behavioral response and one instance of TTS). This information is summarized in Table 32 below.

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to North Pacific right whales. North Pacific right whales are not likely to respond to high-frequency sound sources associated with the proposed training activities because of their hearing sensitivities. While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data specific to North Pacific right whale hearing so we assume that their vocalizations are partially representative of their hearing sensitivities (for additional detail on right whale vocalizations, see Section 4.2.2.9). Sensitivity to varying frequencies of sound by baleen whales has been inferred from observed vocalization frequencies, observed reactions to playback of sounds, and anatomical analyses of the auditory system (Finneran and Jenkins 2012).

Results of studies on blue whales (Goldbogen et al. 2013) (Southall et al. 2011a), which have similar auditory physiology compared to North Pacific right whales and are also generally considered low-frequency cetaceans, indicate that individuals hear some sounds in the mid-frequency range and may exhibit behavioral responses to sounds in this range depending on received level and context. However, as described previously in this opinion regarding the response of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual whales. Based on the best available information, we also assume the response of North Pacific right whales to TTS to be similar to that of blue whales and to not result in fitness consequences.

We also considered the likelihood of harassment of North Pacific right whales occurring in the NMFS-identified Biologically Important Feeding area for the species (Ferguson et al. 2015) which overlaps slightly with the Gulf of Alaska TMAA, or designated critical habitat for the species. The Navy will not use surface ship hull mounted mid-frequency sonar or explosives during training within the portion of the NMFS-identified North Pacific right whale feeding area overlapping the TMAA in the June to September timeframe (see section 2.3.3.2 of this opinion). This mitigation measure is expected to minimize the likelihood that take will occur in a known feeding area for the species. Further, as described in the 2016 Gulf of Alaska SEIS/OEIS, the

typical location of Navy sonar and explosive use in the TMAA is many nautical miles away from the NMFS-identified North Pacific right whale feeding area (i.e., in deep waters, away from the borders of the TMAA) further reducing the likelihood of a take occurring in this area (Navy 2016a). It is also important to note that the areas identified by Ferguson et al. (2015) are not intended to reflect a complete list of areas of biological importance and only represent only a fraction of a species' overall range. Additionally, the delineation of a mapped boundary does not reflect the day-today dynamic nature of marine mammal distributions or of the ocean environment, both of which are subject to perturbation (changes from what is normal due to any outside influence such as climate change, storm events, etc.) along with other key variables such as prey availability and other environmental factors (e.g., sea surface temperature).

Similarly, the TMAA is close to, but does not overlap North Pacific right whale critical habitat in the Gulf of Alaska. The nearest boundary of the Pacific right whale critical habitat is approximately 16 nm (30 km) west of the southwest corner of the TMAA. Sounds from training activities have the potential to reach North Pacific right whale critical habitat in the Gulf of Alaska, but as described further below, are expected to be reduced to negligible levels due to transmission loss by the time it reaches the critical habitat. For example, as described in Table 25 in section 6.3.14.4, results from the Navy's acoustic effects model lists a maximum range to effects for surface ship sonar (i.e., the source with the longest propogation range) of 58.7 to 63.6 nautical miles for low-frequency cetaceans (e.g., North Pacific right whale). At this range, sound levels were estimated to be 132 to 138 dB SPL and only accounted for less than one percent of all behavioral response exposures of low-frequency cetaceans. A higher percentage of behavioral response exposures from surface ship sonar were estimated to occur at much closer range (e.g., 83 percent of low-frequency harassment exposures were estimated to occur between 4.7 and 29 nautical miles from the source at 156-162 dB SPL).

As described in the 2016 FSEIS/OEIS, the Navy rarely, if ever, operates near the corners or edge of the TMAA (i.e., areas of the TMAA closest to North Pacific right whale critical habitat). To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as

Visit, Board, Search, and Seizure training events that are without sonar or explosives (Navy 2016a). North Pacific right whale critical habitat is at least 80 to 120 nautical miles away from areas in the TMAA where the majority of Navy sonar would be used (Navy 2016b) indicating that during the majority of training activities using sonar, sound levels would not reach North Pacific right critical habitat at detectable levels. For this reason, we do not expect any of the two annual instances of behavioral harassment or one instance of TTS to occur in designated critical habitat.

Non-impulsive acoustic stressors from Navy training activities conducted during the five-year year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of North Pacific right whales. We anticipate temporary behavioral responses and instances of TTS, but do not anticipate any injury or mortality from non-impulsive acoustic stressors. We do not anticipate those instances of TTS or behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect non-impulsive acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of this species.

6.6.1.3 *Fin Whale*

The NAEMO output estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The model estimates that some mysticete behavioral reactions could occur at great distances (e.g., exceeding 100 km), though masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b; Baumann-Pickering et al. 2012b; Debich et al. 2013; Debich et al. 2014; Rice et al. 2015) or other potential biological sources (D'Spain and Batchelor 2006), make reaction to the sound from sonar and other active sound sources by mysticetes at these distances less likely. Therefore, the number of exposures that may result in a take is a small subset of the total estimated exposure instances.

Based on the Navy's exposure models, each year we would expect approximately 21,839 instances a year in which a fin whale belonging to the Northeast Pacific stock might be exposed to active sonar associated with training activities above 120 dB SPL. Of those unprocessed exposures, there were 7 model-estimated instances of PTS, 941 model-estimated instances where a fin whale might exhibit a behavioral response as a result of that exposure, and 343 model-estimated instances of TTS. With the expection of the instances of PTS, there were not any model-estimated exposures leading to injury or mortality. The "post-processing" final take estimates include 941 instances where a fin whale would be expected to exhibit a behavioral response, and 350 instances of TTS. This information is summarized in Table 32 below.

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to fin whales. Fin whales are not likely to respond to high-frequency sound sources associated with the proposed training activities because of their hearing

sensitivities (i.e., they likely cannot hear high frequency sources). While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on fin whale hearing so we assume that fin whale vocalizations are partially representative of their hearing sensitivities. Those vocalizations include a variety of sounds described as low frequency moans or long pulses in the 10 Hz to 100 Hz band (Edds 1988; Thompson and Friedl 1982; Watkins 1981a). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15 Hz to 40 Hz range. Ketten (1997a) reports the frequencies of maximum energy between 12 Hz and 18 Hz. Short sequences of rapid calls in the 30 to 90 Hz band are associated with animals in social groups (Clark personal observation and McDonald personal communication cited in Ketten (1997a)). The context for the 30 Hz to 90 Hz calls suggests that they are used to communicate but do not appear to be related to reproduction. Fin whale moans within the frequency range of 12.5 Hz to 200 Hz, with pulse duration up to 36 seconds, have been recorded off Chile (Cummings and Thompson 1994). The whale produced a short, 390 Hz pulse during the moan.

Results of studies on blue whales (Goldbogen et al. 2013) (Southall et al. 2011a), which have similar auditory physiology compared to fin whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. However, as described previously in this opinion, regarding the response of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual whales. Based on the best available information, we assume the response of fin whales to TTS to be similar to that of blue whales. Additionally, several fin whales were tagged during the Southern California-10 BRS and no obvious responses to the controlled exposure to a mid-frequency sound source were detected by the visual observers or in the initial tag analysis (Southall et al. 2011a).

Further, work by Moore and Barlow (2011) indicate that since 1991, there is strong evidence of increasing fin whale abundance in the California Current area, which includes the Southern California Range Complex, an area in which Navy training activities, including the use of MFA sonar, have been ongoing for decades. The authors predict continued increases in fin whale numbers over the next decade in that area, and that perhaps fin whale densities are reaching "current ecosystem limits." This indicates that any potential temporary behavioral impacts on fin whales from the use of MFA during Navy training activities have not prevented fin whale numbers from increasing and potentially nearing ecosystem limits in Southern California. Because the types of training activities in the Gulf of Alaska area are similar to those in Southern California but are significantly smaller in terms of activity levels and much more limited in time, we would expect even lower levels of effects in the Gulf of Alaska action area, and thus we do not expect these activities to be a limiting factor in the recovery of fin whales.

We also considered the likelihood of harassment of fin whales occurring in the NMFS-identified Biologically Important Feeding area for the species (Ferguson et al. 2015) which occurs outside of the TMAA on the northeast and northwest side of Kodiak Island, and around the Semidi Islands (Figure 8). There is the potential for sound from Navy activities to propogate into this feeding area, though in the vast majority of cases, sound from Navy activities would be reduced to negligible levels by the time it reaches this area. As described in the 2016 Gulf of Alaska FSEIS/OEIS, the typical location of Navy sonar and explosive use in the TMAA is many nautical miles away from the NMFS-identified fin whale feeding area (i.e., in deep waters, away from the borders of the TMAA). The Navy rarely, if ever, operates near the corners or edge of the TMAA. To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as Visit, Board, Search, and Seizure training events that are without sonar or explosives usage (Navy 2016a). Because Navy sonar and explosive use is expected to occur such a great distance away from the NMFS-identified Biologically Important Feeding area for fin whales, we would expect sound from Navy activities to be reduced to negligible levels due to transmission loss by the time it reaches this area. In conclusion, for the reasons stated above, we do not expect fin whales to experience TTS or behavioral disturbance while in the BIA around Kodiak Island.

It is also important to note that the areas identified by Ferguson et al. (2015) are not intended to reflect a complete list of areas of biological importance and only represent only a fraction of a species' overall range. Additionally, the delineation of a mapped boundary does not reflect the day-today dynamic nature of marine mammal distributions or of the ocean environment, both of which are subject to perturbation (changes from what is normal due to any outside influence such as climate change, storm events, etc.) along with other key variables such as prey availability and other environmental factors (e.g., sea surface temperature).

Non-impulsive acoustic stressors from Navy training activities and training activities conducted during the five-year year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of fin whales. We anticipate temporary

behavioral responses and instances of TTS, but do not anticipate any injury or mortality from non-impulsive acoustic stressors. We do not anticipate those instances of TTS or behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect non-impulsive acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of this species.

6.6.1.4 *Humpback Whale – Mexico and Western North Pacific DPSs*

The NAEMO output estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The model estimates that some mysticete behavioral reactions could occur at great distances (e.g., exceeding 100 km), though masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b; Baumann-Pickering et al. 2012b; Debich et al. 2013; Debich et al. 2014; Rice et al. 2015) or other potential biological sources (D'Spain and Batchelor 2006), make reaction to the sound from sonar and other active sound sources by mysticetes at these distances less likely. Therefore, the number of exposures that may result in a take is a small subset of the total estimated exposure instances.

Based on the Navy's exposure models, each year we would expect approximately 5,016 potential exposure instances each year in which a humpback whale might be exposed to active sonar associated with training activities above 120 dB SPL. Of those unprocessed exposures, there were 53 model-estimated instances where a humpback whale might exhibit a behavioral response as a result of that exposure, and 16 model estimated instances of TTS. There were no model-estimated exposures leading to injury, including PTS, or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates include 53 instances where a humpback whale would be expected to exhibit a behavioral response and 16 instances of TTS).

The marine mammal density data used in the GOA TMAA modeling is not broken down by DPS. Therefore, this density, and the resulting take estimates, represent a mix of all humpback whale DPSs in the GOA TMAA. In order to estimate how many individuals from each DPS will be affected by Navy non-impulsive acoustic stressors, we pro-rated humpback whale take estimates into three parts based on the probability of encountering humpback whales from each DPS in the Gulf of Alaska summer feeding area. Wade et al. (2016b) analyzed humpback whale movements through the North Pacific Ocean between winter breeding areas and summer feeding areas, using a comprehensive photo-identification stufy of humpback whales in 2004-2005 during the SPLASH project (Structure of Populations, Levels of Abundance and Status of Humpbacks). The authors estimated breeding population composition in summer feeding areas of the North Pacific including Gulf of Alaska. They found that the 89 percent of the humpback whales in the summer Gulf of Alaska feeding grounds are from the Hawaii DPS, 10.5 percent are from the Mexico DPS, and 0.5 percent are from the Western North Pacific DPS. The ESA take

prohibitions for humpback whales affected by Navy activities in the Gulf of Alaska only apply to individuals belonging to whales from the endangered Western North Pacific DPS and the threatened Mexico DPS. Based on the relative abundance information presented above, we anticipate one instance of behavioral take and zero instances of TTS for the endangered Western North Pacific DPS and 5 instances of behavioral take and 2 instances of TTS for the threatened Mexico DPS. All other instances of behavioral harassment and TTS would be assigned to the Hawaii DPS, which is not listed under the ESA. This information is summarized in Table 32 below.

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to humpback whales. Humpback whales are not likely to respond to high-frequency sound sources associated with the proposed training activities because of their hearing sensitivities. We recognize animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations. However, we have no data on humpback whale hearing so we assume that humpback whale vocalizations are partially representative of their hearing sensitivities. As discussed in the *Status of Listed Resources* narrative for humpback whales, these whales produce a wide variety of sounds.

Humpback whales vocalize less frequently in their summer feeding areas than in their winter ranges at lower latitudes. Feeding groups produce distinctive sounds ranging from 20 Hz to 2 kHz, with median durations of 0.2 to 0.8 seconds and source levels of 175 to 192 dB (Thompson et al. 1986b). These sounds are attractive and appear to rally animals to the feeding activity (D'Vincent et al. 1985b; Sharpe and Dill 1997). To summarize, humpback whales produce at least three kinds of sounds:

1. Complex songs with components ranging from 20Hz to 4 kHz with estimated source levels from 144 to 174 dB; these are mostly sung by males on the breeding grounds (Payne and McVay 1971; Winn et al. 1970)

2. Social sounds in the breeding areas that extend from 50Hz to more than 10 kHz with most energy below 3 kHz (Richardson et al. 1995d; Tyack and Whitehead 1983); and

3. Feeding area vocalizations that are less frequent, but tend to be 20Hz to 2 kHz with estimated source levels in excess of 175 dB re 1 μPa-m (Richardson et al. 1995d; Thompson et al. 1986b). Sounds often associated with possible aggressive behavior by males (Silber 1986; Tyack 1983) are quite different from songs, extending from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz. These sounds appear to have an effective range of up to 9 km (Tyack and Whitehead 1983).

Au et al. (2006a) conducted field investigations of humpback whale songs that led these investigators to conclude that humpback whales have an upper frequency limit reaching as high as 24 kHz. Based on this information, it is reasonable to assume that the active mid-frequency

sonar the U.S. Navy uses during active sonar training activities in the Gulf of Alaska TMAA are within the hearing and vocalization ranges of humpback whales. There is limited information on how humpback whales are likely to respond upon being exposed to mid-frequency active sonar (most of the information available addresses their probable responses to low-frequency active sonar or impulsive sound sources). Maybaum (Maybaum 1993) conducted sound playback experiments to assess the effects of mid-frequency active sonar on humpback whales in Hawaiian waters. Specifically, she exposed focal pods to sounds of a 3.3-kHz sonar pulse, a sonar frequency sweep from 3.1 to 3.6 kHz, and a control (blank) tape while monitoring their behavior, movement, and underwater vocalizations. The two types of sonar signals differed in their effects on the humpback whales, although the whales exhibited avoidance behavior when exposed to both sounds. The whales responded to the sounds by increasing their distance from the sound; however, the frequency or duration of their dives or the rate of underwater vocalizations did not change. Similar to the conclusions drawn regarding responses of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual humpback whales. Based on the best available information, we assume the response of humpback whales to TTS to be similar to that of blue whales.

Humpback whales have been known to react to low frequency industrial noises at estimated received levels of 115 to 124 dB (Malme et al. 1985), and to calls of other humpback whales at received levels as low as 102 dB (Frankel et al. 1995). Malme et al. (1985) found no clear response to playbacks of drill ship and oil production platform noises at received levels up to 116 dB re 1 Pa. Frankel and Clark (1998) showed that breeding humpbacks showed only a slight statistical reaction to playbacks of 60 to 90 Hz sounds with a received level of up to 190 dB. Although these studies have demonstrated that humpback whales will exhibit short-term behavioral reactions to boat traffic and playbacks of industrial noise, the long-term effects of these disturbances on the individuals exposed to them are not known. Humpback whales in Stellwagen Bank National Marine Sanctuary reduced their calling rates coincident with an Ocean Acoustic Waveguide Remote Sensing experiment 200 km away with FM pulses centered at 415, 734, and 949 Hz (Risch et al. 2012). However, Gong et al. (2014), disputes these findings, suggesting that Risch et al. (2012) mistakes natural variations in humpback whale song occurrence for changes caused by OAWRS activity approximately 200 km away. Risch et al. (2014) responded to Gong et al. (2014) and highlighted the context-dependent nature of behavioral responses to acoustic stressors.

We also considered the likelihood of harassment of ESA-listed humpback whales occurring in the NMFS-identified Biologically Important Feeding area for the species (Ferguson et al. 2015) which occurs outside of the TMAA and around Kodiak Island (Figure 7). There is the potential for sound from Navy activities to propogate into this feeding area, though in the vast majority of cases, sound from Navy activities would be reduced to negligible levels by the time it reaches this area. As described in the 2016 Gulf of Alaska FSEIS/OEIS, the typical location of Navy

sonar and explosive use in the TMAA is many nautical miles away from the NMFS-identified fin whale feeding area (i.e., in deep waters, away from the borders of the TMAA). The Navy rarely, if ever, operates near the corners or edge of the TMAA (i.e., areas of the TMAA closest to North Pacific right whale critical habitat). To ensure that the Navy is able to conduct realistic training, Navy units must maintain sufficient room to maneuver. Therefore, training activities typically take place some distance away from the TMAA boundary to ensure sufficient sea or air space is available for tactical maneuvers. The Navy also does not typically train next to any limiting boundary because it precludes tactical consideration of the adjacent sea space and airspace beyond the boundary from being a potential threat axis during activities such as anti-submarine warfare training. It is also the case that Navy training activities will generally not be located where it is likely there would be interference from civilian vessels and aircraft that are not participating in the training activity. The nearshore boundary of the TMAA is the location for multiple commercial vessel transit lanes, ship traffic, and low-altitude air routes. This level of civilian activity may otherwise conflict with Navy training activities if those Navy activities were located at that margin of the TMAA and as a result such an area is generally avoided. Given the proximity to Kodiak Island and Kenai Peninsula, the nearshore margin of the TMAA is only likely to involve training activities such as Visit, Board, Search, and Seizure training events that are without sonar or explosives usage (Navy 2016a). Because Navy sonar and explosive use is expected to occur such a great distance away from the NMFS-identified Biologically Important Feeding area for humpback whales, we would expect sound from Navy activities to be reduced to negligible levels due to transmission loss by the time it reaches this area. In conclusion, for the reasons stated above, we do not expect humpback whales to experience TTS or behavioral disturbance while in the BIA around Kodiak Island.

It is also important to note that the areas identified by Ferguson et al. (2015) are not intended to reflect a complete list of areas of biological importance and only represent only a fraction of a species' overall range. Additionally, the delineation of a mapped boundary does not reflect the day-today dynamic nature of marine mammal distributions or of the ocean environment, both of which are subject to perturbation (changes from what is normal due to any outside influence such as climate change, storm events, etc.) along with other key variables such as prey availability and other environmental factors (e.g., sea surface temperature).

Non-impulsive acoustic stressors from Navy training activities conducted during the five-year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of humpback whales from the Mexico or Western North Pacific DPSs. We anticipate temporary behavioral responses and instances of TTS, but do not anticipate any injury or mortality from non-impulsive acoustic stressors. We do not anticipate those instances of TTS or behavioral responses to result in fitness consequences to any individuals. Therefore, we do not anticipate those instances of TTS or behavioral response to result in substantial changes in reproduction, numbers, or distribution of humpback whales from either the Mexico or Western North Pacific DPS.

6.6.1.4.1 Sei Whale

The NAEMO output estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are discussed below. The model estimates that some mysticete behavioral reactions could occur at great distances (e.g., exceeding 100 km), though masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b; Baumann-Pickering et al. 2012b; Debich et al. 2013; Debich et al. 2014; Rice et al. 2015) or other potential biological sources (D'Spain and Batchelor 2006), make reaction to the sound from sonar and other active sound sources by mysticetes at these distances less likely. The number of exposures that may result in a take is a small subset of the total estimated exposure instances.

Based on the Navy's exposure models, each year we would expect approximately 141 instances in which a sei whale might be exposed to active sonar associated with training activities at a levels above 120 dB SPL. Of those unprocessed exposures, there were five model-estimated instances where a sei whale might exhibit a behavioral response as a result of that exposure, and one model estimated instance of TTS. There were no model-estimated exposures leading to injury, including PTS, or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates include five instances where a sei whale would be expected to exhibit a behavioral response and one instance of TTS). This information is summarized in Table 32 below.

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to sei whales. Like blue and fin whales, sei whales are not likely to respond to high-frequency sound sources associated with the proposed training activities and training activities because of their hearing sensitivities. As discussed in the *Status of Listed Resources* section of this opinion, we have no specific information on the hearing sensitivity of sei whales. Based on their anatomical and physiological similarities to both blue and fin whales, we assume that the hearing thresholds of sei whales will be similar as well and will be centered on low-frequencies in the 10 to 200 Hz.

Results of studies on blue whales (Goldbogen et al. 2013; Southall et al. 2011a), which have similar auditory physiology compared to sei whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. However, as we discussed in the response of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual whales. Based on the best available information, we also assume the response of sei whales to TTS to be similar to that of blue whales (i.e., not fitness consequences expected).

Non-impulsive acoustic stressors from Navy training activities and training activities conducted during the five-year year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of sei whales. We anticipate temporary behavioral responses and instances of TTS, but do not anticipate any injury or mortality from non-impulsive acoustic stressors. We do not anticipate those instances of TTS or behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect non-impulsive acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of this species.

6.6.1.4.2 Sperm Whale

The NAEMO output estimates that sperm whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The number of exposures that may result in a take is a small subset of the total estimated exposure instances. Based on the Navy's exposure models, each year we would expect approximately 2,074 potential instances in which sperm whales might be exposed to active sonar associated with training activities above 120 dB SPL. Of those unprocessed exposures, there were 98 model-estimated instances where a sperm whale might exhibit a behavioral response as a result of that exposure, and zero model estimated instances of TTS. There were no model-estimated exposures leading to injury, including PTS, or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates include 98 instances where a sperm whale would be expected to exhibit a behavioral response and zero instances of TTS). This information is summarized in Table 32 below.

Although there is no published audiogram for sperm whales, sperm whales would be expected to have good, high frequency hearing because their inner ear resembles that of most dolphins, and appears tailored for ultrasonic (>20 kHz) reception (Ketten 1994). The only data on the hearing range of sperm whales are evoked potentials from a stranded neonate, which suggest that neonatal sperm whales respond to sounds from 2.5 to 60 kHz. Sperm whales vocalize in high- and mid-frequency ranges; most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz. Other studies indicate sperm whales' wide-band clicks contain energy between 0.1 and 20 kHz (Goold and Jones 1995; Weilgart and Whitehead 1993b). Ridgway and Carder (Ridgway and Carder 2001) measured low-frequency, high amplitude clicks with peak frequencies at 500 Hz to 3 kHz from a neonate sperm whale.

Based on their hearing sensitivities and vocalizations, the active sonar and sound pressure waves from the underwater detonations the Navy conducts might mask sperm whale hearing and vocalizations. There is some evidence of disruptions of clicking and behavior from sonars (Goold 1999a; Watkins 1985), pingers (Watkins and Schevill 1975b), the Heard Island Feasibility Test (Bowles et al. 1994), and the Acoustic Thermometry of Ocean Climate (Costa et al. 1998). Sperm whales have been observed to frequently stop echolocating in the presence of

underwater pulses made by echosounders (Watkins and Schevill 1975b). Goold (1999a) reported six sperm whales that were driven through a narrow channel using ship noise, echosounder, and fish finder emissions from a flotilla of 10 vessels. Watkins and Scheville (Watkins and Schevill 1975b) showed that sperm whales interrupted click production in response to pinger (6 to 13 kHz) sounds. They also stopped vocalizing for brief periods when codas were being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

Sperm whales have been reported to have reacted to military sonar, apparently produced by a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins 1985). Captive bottlenose dolphins and a white whale exhibited changes in behavior when exposed to 1 sec pulsed sounds at frequencies similar to those emitted by multi-beam sonar that is used in geophysical surveys (Ridgway and Carder 1997; Schlundt et al. 2000b), and to shorter broadband pulsed signals (Finneran et al. 2000a; Finneran et al. 2002). Behavioral changes typically involved what appeared to be deliberate attempts to avoid the sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al. 2002; Schlundt et al. 2000b). Dolphins exposed to 1-sec intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 Pa_{rms} and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such reactions to shorter pulses were higher (Finneran et al. 2000a; Finneran et al. 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran et al. 2002). In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway and Carder 1997; Schlundt et al. 2000b). The relevance of these data to free-ranging odontocetes is uncertain. In the wild, cetaceans sometimes avoid sound sources well before they are exposed to the levels listed above, and reactions in the wild may be more subtle than those described by Ridgway et al. (1997) and Schlundt et al. (2000b).

Other studies identify instances in which sperm whales did not respond to anthropogenic sounds. Sperm whales did not alter their vocal activity when exposed to levels of 173 dB re 1 Pa from impulsive sounds produced by 1 g TNT detonators (Madsen and Mohl 2000). Richardson et al. (1995d) citing a personal communication with J. Gordon suggested that sperm whales in the Mediterranean Sea continued calling when exposed to frequent and strong military sonar signals. When Andre et al. (1997) exposed sperm whales to a variety of sounds to determine what sounds may be used to scare whales out of the path of vessels, sperm whales were observed to have startle reactions to 10 kHz pulses (180 dB re 1 Pa at the source), but not to the other sources played to them.

Published reports identify instances in which sperm whales have responded to an acoustic source and other instances in which they did not appear to respond behaviorally when exposed to seismic surveys. Mate et al. (1994) reported an opportunistic observation of the number of sperm

whales to have decreased in an area after the start of airgun seismic training. However, Davis et al. (2000) noted that sighting frequency did not differ significantly among the different acoustic levels examined in the northern Gulf of Mexico, contrary to what Mate et al. (1994) reported. Sperm whales may also have responded to seismic airgun sounds by ceasing to call during some (but not all) times when seismic pulses were received from an airgun array >300 km away (Bowles et al. 1994).

A study offshore of northern Norway indicated that sperm whales continued to call when exposed to pulses from a distant seismic vessel. Received levels of the seismic pulses were up to 146 dB re 1 μPa peak-to-peak (Madsen et al. 2002). Similarly, a study conducted off Nova Scotia that analyzed recordings of sperm whale sounds at various distances from an active seismic program did not detect any obvious changes in the distribution or behavior of sperm whales (McCall-Howard 1999). Recent data from vessel-based monitoring programs in United Kingdom waters suggest that sperm whales in that area may have exhibited some changes in behavior in the presence of operating seismic vessels (Stone 1997; Stone 1998; Stone 2000; Stone 2001; Stone 2003). However, the compilation and analysis of the data led the author to conclude that seismic surveys did not result in observable effects to sperm whales (Stone 2003). The results from these waters seem to show that some sperm whales tolerate seismic surveys.

These studies suggest that the behavioral responses of sperm whales to anthropogenic sounds are highly variable, but do not appear to result in the death or injury of individual whales or result in reductions in the fitness of individuals involved. Responses of sperm whales to anthropogenic sounds probably depend on the age and sex of animals being exposed, as well as other factors. There is evidence that many individuals respond to certain sound sources, provided the received level is high enough to evoke a response, while other individuals do not.

Non-impulsive acoustic stressors from Navy training activities and training activities conducted during the five-year year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of sperm whales. We anticipate temporary behavioral responses and instances of TTS, but do not anticipate any injury or mortality from non-impulsive acoustic stressors. We do not anticipate those instances of TTS or behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect non-impulsive acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of this species.

6.6.1.5 ***Steller Sea Lion – Western DPS***

The NAEMO output estimates that Steller sea lions from the western DPS will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities between April and October. The number of exposures that may result in a take is a small subset of the total estimated exposure instances. Based on the Navy's exposure models, each year we expect approximate 13,689 instances in which a steller sea lion might be exposed to active sonar

associated with training activities at a levels above 120dB SPL. Of those unprocessed exposures, there were 286 model-estimated instances where a Western DPS steller sea lion might exhibit a behavioral response as a result of that exposure. There were no model-estimated exposures leading to TTS, injury (including PTS), or mortality. Therefore, the "post-processing" stage did not change the final take estimates from those estimated by the model (i.e., final take estimates includes 286 instances where a Western DPS steller sea lion would be expected to exhibit a behavioral response). This information is summarized in Table 32 below.

Pinnipeds in the water are tolerant of anthropogenic noise and activity. If sea lions are exposed to sonar or other active acoustic sources they may react in a number of ways depending on their experience with the sound source and what activity they are engaged in at the time of the acoustic exposure. Sea lions may not react at all until the sound source is approaching within a few hundred meters and then may alert, approach, ignore the stimulus, change their behaviors, or avoid the immediate area by swimming away or diving. Due to the short duration of any behavioral responses exhibited, we do not expect exposure to acoustic stressors to result in any fitness consequences to individual sea lions.

Non-impulsive acoustic stressors from Navy training activities and training activities conducted during the five-year year period of the proposed MMPA rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of the Western DPS of steller sea lions. We anticipate temporary behavioral responses, but do not anticipate any injury, mortality, or instances of TTS from non-impulsive acoustic stressors. We do not anticipate behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect non-impulsive acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of this species.

6.6.1.6 *Summary of Effects to Marine Mammals from Non-impulsive Acoustic Stressors*

Predicted effects from training activities using sonar and other active acoustic sources on ESA-listed marine mammals are shown in Table 32. The predicted effects are the result of the acoustic analysis, including acoustic effects modeling, followed by consideration of animal avoidance of multiple exposures, avoidance by sensitive species of areas with a high level of activity, and Navy mitigation measures. Only behavioral responses (non-TTS effects) and TTS effects are predicted.

Table 32. Summary of Effects to Marine Mammals from non-impulsive Acoustic Stressors

Species	Behavioral	TTS	PTS
North Pacific right whale	2	1	0
Humpback whale – Mexico DPS	5	2	0
Humpback whale – Western North Pacific DPS	1	0	0

Blue whale	38	9	0
Fin whale	941	350	0
Sei whale	5	1	0
Sperm whale	98	0	0
Steller sea lion – Western DPS	286	0	0

6.6.2 Exposure and Response of Fish to Non-Impulsive Acoustic Stressors

For the purposes of this and the U.S. Navy Northwest Training and Testing consultation (NMFS 2015d), the Navy and NMFS developed sound exposure criteria for low and mid frequency sonar for fish (see section 6.4.7). Sound exposure criteria varied depending on whether or not the species of fish has a swim bladder and whether or not the swim bladder is involved in hearing. As discussed in Popper et al. (2014b), fishes with swim bladders near the ear that are involved in hearing generally have lower sound pressure thresholds than do purely particle motion species. All fish considered in this opinion have swim bladders that are not involved in hearing so our discussion and analysis focuses on these criteria. The criteria developed are discussed in Section 3.1.2 and shown in Section 6.4.7.

Threshold criteria were not developed for high frequency sonar sources (i.e., source bins HF1, HF6, and ASW3). Only a few species of shad within the Clupeidae family (herrings) are known to be able to detect high-frequency sonar and other active acoustic sources greater than 10,000 Hz. The species considered in this opinion would not detect these sounds and would therefore experience no stress, behavioral disturbance, or auditory masking. High-frequency sonar is not anticipated to cause mortality or injury due to the lack of fast rise times, lack of high peak pressures, and the lack of high acoustic impulse. Also, similar to low and mid-frequency sonar, mortality or injury have not been shown to occur from exposure to high frequency sonar sources. For these reasons, the potential effects of high frequency active sonar will not be discussed further in this opinion.

While we present threshold criteria for mortality, mortal injury, and recoverable injury from exposure to low and mid-frequency sonar, these effects are extremely unlikely to occur. Sonar is not anticipated to cause mortality or injury due to the lack of fast rise times, lack of high peak pressures, and the lack of high acoustic impulse. Additionally, exposure to low and mid-frequency sonar has been tested and has not been shown to cause mortality or injury to any fish with swim bladders (Kane et al. 2010; Popper et al. 2007). The values presented represent the highest sound exposure levels which have been tested to date, none of which have resulted in mortality or injury for fish with swim bladders not involved in hearing. All ESA-listed fish considered in this opinion have swim bladders not involved in hearing. The criteria developed were highly conservative. Therefore, we conclude that ESA-listed fish species considered in this opinion will not experience injury or mortality from exposure to low and mid-frequency sonar.

Sonar has been shown to induce TTS in some fish species with swim bladders (Halvorsen et al. 2013; Halvorsen et al. 2012; Popper et al. 2007). As shown in Section 6.4.7, all fish within a within one meter of sources in the ASW2 source bin (an upper low-frequency source between 895 and 1,005 Hz) could experience TTS. Sources in the ASW2 source bin are proposed to be used for 40 hours annually. It is unlikely that ESA-listed fish in relatively close proximity to sources in the ASW2 bin would not move to actively avoid being within one meter of these sources because they would likely perceive the sonar source as a potential predator. However, if any ESA-listed fish experiences TTS from exposure to these sources, any hearing loss would be temporary, and individuals would be expected to fully recover shortly after exposure (Lombarte et al. 1993; Smith et al. 2006a). Further, the ESA-listed species considered in this opinion lack notable hearing specialization, minimizing the likelihood of any instance of TTS affecting an individual's fitness. To our knowledge, no studies have examined the fitness implications when a fish, without notable hearing specialization, experiences TTS. Popper et al. (2014b) suggested that fishes experiencing TTS may have a decreased ability to communicate, detect predators or prey, or assess their environment. However, the species considered in this opinion are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014b). Additionally, hearing is not thought to play a role in salmon migration (e.g., Putnam et al. 2013). Because any TTS experienced would be temporary and the ESA-listed fish species considered in the opinion are able to rely on alternative mechanisms for these essential life functions, instances of TTS would not kill or injure any fish, nor would any such instances create the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. For these reasons, any TTS experienced by fish would not rise to the level of take and would not have fitness level consequences at the individual or population level. Therefore, the effect of exposure to acoustic stressors that may result in TTS is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

Available evidence indicates that sonar use in the action area would not have the potential to substantially mask key environmental sounds for the ESA-listed fish species considered in this opinion. Mid-frequency sound sources would not mask environmental sounds for the fish considered in this opinion because these species are limited to detection of the particle motion component of low-frequency sounds (DoN 2015c). Low frequency sonar usage could cause brief periods of masking for fish within a few kilometers of the source while the source is active, but the effects would be infrequent and transient as both the vessel and the individual fish are moving while in the ocean environment. Popper et al. (2014b) also noted that the narrow bandwidth of most sonar signals would only result in a limited range of frequencies being masked. Additionally, as noted previously, the species considered in this opinion can utilize other sensory cues (e.g., sight, particle motion detection) during the brief period in which low-frequency sonar may be in close proximity to an individual.

We also assessed the potential for exposure to non-impulsive acoustic stressors to result in behavioral responses. The ESA-listed fish species considered in this opinion cannot hear mid- or high-frequency sonar sources so would not be expected to respond behaviorally to these sound sources. ESA-listed salmonids would be expected to be able to detect low-frequency sources. As documented in section 6.4.5, there is a lack of studies that have investigated the behavioral reactions of unrestrained fish to anthropogenic sound, especially in the natural environment. Studies of caged fish have identified three basic behavioral reactions to sound: startle, alarm, and avoidance (McCauley et al. 2000; Pearson et al. 1992). McCauley et al. (2000) found that fish resumed normal behavioral patterns within minutes after cessation of the sound exposure. Popper et al. (2014b) concluded that the relative risk of a fish eliciting a behavioral reaction in response to low-frequency sonar was low, regardless of the distance from the sound source. Should an ESA-listed fish elicit a behavioral reaction from exposure to low-frequency sonar, we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-exposure behavior immediately or within minutes following each exposure. Exposure time is expected to be brief as both the vessel and the individual fish are moving while in the ocean environment. Similar to instances of TTS, we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering, and such reactions would not rise to the level of take. Therefore, the effect of exposure to non-impulsive acoustic stressors from Navy activities is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

6.7 Exposure and Response to Impulsive Stressors During Training Activities

The following sections discuss our analysis of impulsive acoustic stressors that are likely to adversely affect ESA-listed species. If a stressor is likely to adversely affect any of the ESA-listed species in the Gulf of Alaska TMAA, it is discussed further in this Section.

Marine mammals could be exposed to energy and sound from underwater explosions associated with proposed activities. The NAEMO, in conjunction with the explosive thresholds and criteria (as detailed further in Section 3.8.3.1.4 of the GOA Supplemental EIS/OEIS), are used to predict impacts on marine mammals from underwater explosions.

Energy from explosions is capable of causing mortality, direct injury, hearing loss, or a behavioral response depending on the level of exposure. The death of an animal will, of course, eliminate future reproductive potential and cause a long-term consequence for the individual that must then be considered for potential long-term consequences for the population. Exposures that result in long-term injuries such as PTS may limit an animal's ability to find food, communicate with other animals, or interpret the surrounding environment. Impairment of these abilities can decrease an individual's chance of survival or impact its ability to successfully reproduce. TTS

can also impair an animal's abilities, but the individual may recover quickly with little significant effect. Behavioral responses can include shorter surfacings, shorter dives, fewer blows (breaths) per surfacing, longer intervals between blows, ceasing or increasing vocalizations, shortening or lengthening vocalizations, and changing frequency or intensity of vocalizations (National Research Council 2005). However, it is not clear how these responses relate to long-term consequences for the individual or population (National Research Council 2005).

Explosions in the ocean or near the water surface can introduce loud, impulsive, broadband sounds into the marine environment. These sounds are likely within the audible range of most cetaceans, but the duration of individual sounds is very short. The direct sound from explosions used during Navy training activities last less than a second, and most events involve the use of only one or a few explosions. Furthermore, events are dispersed in time and throughout the action area. These factors reduce the likelihood of these sources causing either substantial or long-term auditory masking in marine mammals.

6.7.1 Exposure and Response of Marine Mammals to Impulsive Acoustic Stressors

As with non-impulsive sources, we considered exposure estimates from the Navy Acoustic Effects Model for marine mammals. First, we estimated the total number of ESA-listed species (animats) that would be exposed to acoustic sources prior to the application of a dose-response curve or criteria. We term these the "unprocessed" estimates. This estimate is the number of times individual animats or animals are likely to be exposed to the acoustic environment that is a result of training activities, regardless of whether they are "taken" as a result of that exposure. In most cases, the number of animals "taken" by an action would be a subset of the number of animals that are exposed to the action because (1) in some circumstances, animals might not respond to an exposure and (2) some responses may be negative for an individual animal without constituting a form of "take" (for example, some physiological stress responses only have fitness consequences when they are sustained and would only constitute a "take" as a result of cumulative exposure).

A second set of exposure estimates ("model-estimated") of listed species were generated and "processed" using criteria for temporary and permanent threshold shift developed by the Navy and NMFS' Permits Division for the purpose of identifying harassment pursuant to the MMPA. If injury and mortality were estimated to occur following processing, the modeled estimates are further analyzed to account for standard mitigation actions that NMFS's Permits Division requires under the MMPA rule and LOA to avoid marine mammals, and avoidance responses that would be expected from individual animals once they sense the presence of Navy vessels or aircraft (post-processing; see Section 3.1.2 for further detail). Mitigation measures are effective at reducing instances of injury or mortality, but would not further reduce potential behavioral impacts to lesser impacts due to the potential distance from the source stressor. The Navy states

that avoidance and mitigation only reduces those "Level A" (potential to injure or kill) impacts to "Level B" impacts. The final take estimates are the result of the acoustic analysis, including acoustic effects analysis, followed by consideration of animal avoidance of multiple exposures and Navy mitigation measures (only if injury and mortality were estimated to occur following processing). Since these final estimates represent incidental take for purposes of the ESA, we base our jeopardy analyses and determinations on these estimates.

6.7.1.1 *Blue Whale*

Acoustic modeling predicts 86 total exposures per year to blue whales from explosive sources above 120dB SPL during training activities in the action area. Of those 86 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any blue whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.2 *North Pacific right whale*

Acoustic modeling predicts 17 total exposures per year to North Pacific right whales from explosive sources above 120dB SPL during training activities in the action area. Of those 17 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any North Pacific right whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.3 *Humpback Whales*

Acoustic modeling predicts 133 total exposures per year to humpback whales (from the Mexico, Western North Pacific and Hawaii DPSs) from explosive sources above 120dB SPL during training activities in the action area. Of those 133 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any humpback whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.4 *Fin Whale*

Acoustic modeling predicts 2,164 total exposures per year to fin whales from explosive sources above 120dB SPL during training activities in the action area. Of those 2,164 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any fin whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.5 *Sei Whale*

Acoustic modeling predicts 20 total exposures per year to sei whales from explosive sources above 120dB SPL during training activities in the action area. Of those 20 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any sei whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.6 *Sperm Whale*

Acoustic modeling predicts 218 total exposures per year to sperm whales from explosive sources above 120dB SPL during training activities in the action area. Of those 218 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any sperm whale acoustic exposure instances will result in responses that rise to the level of take.

6.7.1.7 *Steller Sea Lion – Western DPS*

Acoustic modeling predicts 1,208 total exposures per year to Western DPS steller sea lions from explosive sources above 120dB SPL during training activities in the action area. Of those 1,208 potential exposures, there were zero model-estimated instances of exposure that would exceed the thresholds for mortality, injury (including PTS), TTS, or behavioral response. Therefore, we do not anticipate any Western DPS steller sea lion acoustic exposure instances will result in responses that rise to the level of take.

6.7.2 Exposure and Response of Fish to Impulsive Acoustic Stressors

With the exception of a low number (n=6) of relatively small ordnance (5 to 10 lbs), training activities using explosive ordnance that could affect ESA-listed fish will occur off the Alaskan continental shelf (>40 to 100 nm from shore, depending on which land feature the measurement is made from). Table 33 lists the number of explosives used during training that are likely to adversely affect ESA-listed fish. A more thorough description of each of these activities is included in the *Description of the Action* and the Navy's FSEIS/OEIS (DoN 2015d).

Table 33. Explosive ordnances used during Navy training activities in the Gulf of Alaska TMAA action area that are likely to adversely affect ESA-listed fish.

Source Class: Net Explosive Weight (pounds)	Representative munition	Ordnance per year (annual)
E5: 5-10*	5" projectile	6
E5: 5-10+	5" projectile	50
E9: 100-250	500 lb. bomb	64
E10: 250-500	1,000 lb. bomb	6
E12: 650-1,000	2,000 lb. bomb	2

*Activities with these E5 ordnances could occur on the continental shelf
+Activities with these E5 ordnances would occur off the continental shelf.

Below we analyze the effects of these explosions on ESA-listed fish species. No species of Pacific salmonid originating from freshwater habitat in Alaska are listed under the ESA. West Coast salmonid species currently listed under the ESA originate in freshwater habitat in Washington, Oregon, Idaho, and California. At least some of the listed salmon and steelhead from Oregon, Washington, and Idaho are presumed to range into marine waters off Alaska during ocean migration and growth to maturity phases of their anadromous life history. It is difficult to accurately estimate the number of individuals from each ESU/DPS that will experience adverse effects from elevated underwater noise and sound pressures in the marine environment because fish distribution is influenced by a number of environmental factors. During ocean migration in Pacific marine waters, a portion of the ESA-listed salmonid ESU/DPSs go into the Gulf of Alaska. In that habitat they are mixed with hundreds to thousands of other stocks (Bellinger et al. 2015) originating from the Columbia River, British Columbia, Alaska, and Asia, making it even more difficult to identify how many individuals of certain ESU/DPSs may be affected by an activity at a specified location and time. The listed fish are not visually distinguishable from the other, unlisted, stocks.

We estimated the number of salmon and steelhead that may be killed or injured from detonations in the offshore environment using the acoustic threshold criteria established for this and the U.S. Navy Northwest Training and Testing biological opinion (NMFS 2015d) and available information on the offshore densities and distributions of the ESA-listed fish species considered in this opinion. In this calculation, we also estimate the amount of potential oceanic habitat of each of these species that could be affected (instantaneous effect on water column habitat from an explosive; not a long term effect). Below we summarize the information available regarding the habitat use of ESA-listed Chinook, coho, chum, sockeye, and steelhead from Oregon, Washington, and Idaho in the Gulf of Alaska. This information will allow us to determine which ESUs and DPSs may be affected by a specific activity, which life stage may be affected, and to assess the effect of underwater explosions on the ESA-listed fish species considered in this opinion.

6.7.2.1.1 Chinook

Chinook salmon distribution in marine waters can be identified in general terms only because it varies seasonally and inter-annually due to a variety of environmental factors (PFMC 2014). Two general life history strategies have been described for Chinook outmigrating from their natal rivers: subyearling life history types which enter marine waters during their first year of life and tend to remain in shallow coastal waters, and yearling types, which spend more time in freshwater before migrating to the ocean, and migrate further offshore and north faster than subyearlings (Burke et al. 2013). In general, once Chinook leave their natal rivers, they use the cool, upwelled waters of the continental shelf for migration and feeding (Bellinger et al. 2015).

It appears that 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. 2012a). Trudel et al. (2009) used coded-wire tag recoveries to derive distribution and migration patterns of juvenile Chinook salmon along the continental shelf of North America. They found that the vast majority of juvenile Chinook from the Columbia River did not migrate further north than southeast Alaska. Further, they found that no juvenile Chinook from the coast of Washington, coasts of Oregon and California, or Puget Sound migrated further north than the west coast of Vancouver Island. The authors noted that, except for some fish from the Columbia River, most juvenile Chinook remained with 100-200 km of their natal rivers until their second year at sea, when many then initiated their northward migration. Immature Chinook salmon are also predominantly found on the continental shelf in the Gulf of Alaska (with some exceptions near the Aleutian Islands, far west of the action area), though are distributed more widely throughout the Gulf of Alaska than juveniles (Echave et al. 2012a). Echave et al. (2012a) observed a relatively high abundance of mature Chinook salmon within Southeast Alaska waters (outside of the action area), but the authors noted that this was likely because the surveys were carried out in the summer months of June, July, and August when mature salmon are returning to spawn. The authors documented very low abundances of mature Chinook in areas where Navy explosives will occur.

Migratory patterns of Chinook salmon can vary greatly within and among populations (PFMC 2014), but some general patterns have been described. For example, Chinook salmon originating from north of Cape Blanco in Oregon tend to migrate towards the Gulf of Alaska, whereas those originating south of Cape Blanco tend to migrate west and south to forage in waters off Oregon and California (PFMC 2014). Weitkamp (2010) examined coded wire-tag recovery data and found that Chinook salmon originating from a particular freshwater region share a common marine distribution. Chinook originating from Washington and Oregon were recovered, presumably while returning to spawn in their natal streams, within an area from their respective state coasts to southeast Alaska, and fish originating from southern Oregon and California were generally only recovered off the coast of Oregon and California. While these general patterns have been observed, Weitkamp and Neely (2002) suggested that Pacific salmon, including

Chinook, exhibit high diversity in ocean migration patterns, rivaling the variability that has been well demonstrated in freshwater life history. Celewycz et al. (2014) presented data analyzing Chinook salmon distribution in the eastern North Pacific Ocean from coded wire tag recoveries. Significant variability in ocean distribution was observed. For example, Chinook salmon from Washington and Oregon were recovered as far north as Bristol Bay, but as far south as northern California. Chinook salmon from Idaho were recovered as far north as the Gulf of Alaska and as far south as northern California. Chinook from the Puget Sound ESU are generally found in ocean environments from the Washington coast to the west coast of Vancouver Island, though a small percentage are recovered as far north as Alaska (NMFS 2008a). Bellinger et al. (2015) used genetic stock identification techniques in the ocean salmon troll fishery and found Puget Sound salmon as far south as Fort Bragg, California. Chinook from the Lower Columbia River ESU are found as far north as Alaska (NMFS 2008a), but as far south as offshore of San Francisco (Bellinger et al. 2015). Upper Willamette River Chinook are thought to be a far north migrating stock (NMFS 2008a), though Bellinger et al. (2015) observed individuals from this ESU caught as far south as the Klamath River region. The center of Snake River fall-run ESUs ocean distribution is thought to be located off the west coast of Vancouver Island (NMFS 2008a), though they have been found as far south as the Bay Area, California (Bellinger et al. 2015). The six Chinook ESUs considered in this opinion have all been documented in the Gulf of Alaska (e.g., Crane et al. 2000; Templin and Seeb 2004; Wahle and Pearson 1981; Wahle and Vreeland 1978).

A number of factors can drive variation in migratory pathways for Chinook salmon. Robinson and Sauer (2011) found that copepod community structure helps determine salmon distribution in oceanic waters because it provides useful information on ocean conditions (i.e., strength of upwelling). Burke et al. (2013) found that Columbia River yearling Chinook salmon have stock-specific spatial distributions in the marine environment that shift through time. The authors found that geospatial variation (e.g., latitude and distance from shore) drove habitat selection in the marine environment more than environmental variation (e.g., chlorophyll *a* and temperature), potentially leading individuals to select habitat areas with suboptimal environmental conditions. Bi et al. (2007) indicated that coho abundance was strongly correlated with variations in chlorophyll *a* concentrations (which vary annually), and observed large temporal variations in overall habitat usage. Sampling indicated that relatively large areas of habitat usage in some years with much lower areas of habitat usage in others. This study also highlighted the variability in habitat selection by life stage with subyearling and subadult Chinook found closer to shore than yearling Chinook.

Chinook are thought to be less surface oriented than other Pacific salmon, most abundant at depths of 30 to 70 m, and most often caught as adults in commercial troll fisheries at depths of 30 m or greater (PFMC 2014). However, juvenile Chinook salmon are known to be more abundant than adults near the surface, most frequently found at depths of less than 37 m (Fisher and Pearcy 1995). Walker et al. (2007) observed Chinook at an average depth of 42 meters.

Through observations of 883 purse seine net sets, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

6.7.2.1.2 Coho

In general, once coho leave their natal rivers, they use the cool, upwelled waters of the continental shelf for migration and feeding (Bellinger et al. 2015). Two general patterns have been described for coho salmon once they emigrate from freshwater with some spending several weeks in coastal waters before migrating north and offshore, and others remaining in coastal waters for at least the first summer before migrating north (PFMC 2014). In the Gulf of Alaska, juvenile coho salmon predominantly occur in coastal waters, throughout the continental shelf and slope (Echave et al. 2012b; Quinn 2005). Immature and mature coho in the Gulf of Alaska also occur along the continental shelf, but are distributed into offshore oceanic waters beyond the shelf break as well (Echave et al. 2012b).

Coho salmon distribution in the marine environment varies considerably among seasons, years, life stages, and populations. Weitkamp and Neely (2002) provided evidence that coho salmon exhibit high diversity in ocean migration patterns, rivaling the variability that has been well demonstrated in freshwater life history. The authors also showed that coho salmon from different freshwater regions are generally recovered, presumably while returning to spawn in their natal rivers, from different areas of the coastal ocean, identifying 12 distinct ocean distribution patterns from California to Alaska. However, despite these general patterns, fish from a given population were widely distributed in the coastal ocean (Weitkamp and Neely 2002). (Masuda et al. 2015b) presented data analyzing coho salmon distribution in the eastern North Pacific Ocean from coded wire tag recoveries. Significant variability in ocean distribution was observed. For example, coho salmon from Oregon were recovered as far north as Bristol Bay, but as far south as the Oregon/California border. Coho salmon from Washington were found to distribute in close proximity to Kodiak Island, but also as far south as the Oregon/California border. Morris et al. (2007) used coded wire tag recoveries to map the distribution of North American juvenile coho salmon and found that juvenile coho from the Lower Columbia River and from coastal Oregon were recovered in, or in close proximity to, the Gulf of Alaska TMAA, and as far west as near Kodiak Island. Bi et al. (2007) indicated that coho abundance was strongly correlated with variations in chlorophyll *a* concentrations (which vary annually), and observed large temporal variations in overall habitat usage. Sampling indicated that relatively large areas of habitat usage in some years with much lower areas of habitat usage in others. This study also highlighted the variability in habitat selection by life stage with subadult coho found closer to shore than yearling coho.

In marine waters, coho are generally found within the upper portion of the water column (PFMC 2014). Walker et al. (2007) found that the average depth of coho salmon in the North Pacific Ocean was 11 meters. Through observations of 883 purse seine net sets off the Oregon and

Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

6.7.2.1.3 Chum

The ocean distribution of chum is thought to be the broadest of any Pacific salmon (Neave et al. 1976), with the species found throughout the North Pacific Ocean north of the Oregon/Washington border. In general, chum move north and west along the coast upon entering saltwater, and have moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). Hartt and Dell (1986) observed that the vast majority of juvenile chum from Washington state migrate northward within a narrow coastal belt less than 20nm miles. Juvenile chum salmon are generally believed to migrate far to the north by the late summer after they have entered saltwater in the spring (Hartt and Dell 1986), though Pearcy and Fisher (1990) suggest that at least some individuals reside in coastal Washington waters for several months after they enter the marine environment. Some data suggests 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 and Dell 1986). (Myers et al. 1996a) 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 of sockeye from these areas were observed. Echave et al. (2012b) found that within the Gulf of Alaska, juvenile chum salmon are distributed throughout the inner and middle shelf along the Gulf coastline from Dixon entrance to the eastern Aleutian Islands, but that by the end of their first fall at sea, most fish have moved off the continental shelf into open waters (Quinn 2005). 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. 2012b).

Chum salmon are known to be surface-oriented, using the upper 20 m of the water column 78 percent of the time during the day and 95 percent of the time at night. The remaining time, they can be found down to depths of 60 meters (Ishida et al. 1997). Similarly, Walker et al. (2007) found the average depth of chum salmon to be 16 meters in the North Pacific Ocean. Through observations of 883 purse seine net sets off the U.S. west coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

6.7.2.1.4 Sockeye

In general, it is thought that sockeye follow a similar migration pattern as chum once they enter the ocean, moving north and west along the coast, and have moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). Previously, French et al. (1976b) summarized the general migration pattern of sockeye salmon originating in the various tributaries of the northeastern Pacific Ocean from the Alaska Peninsula to the Columbia River. Tag recovery data indicated a general mixing of these stocks during their residence in the

northeastern Pacific Ocean. These fish primarily occur east of 160° W and north of 48° N. It is thought that most fish originating from these areas have departed the high seas by early August of their second year at sea, to return to their natal rivers to spawn (French et al. 1976b). Tucker et al. (2009) did not observe juvenile sockeye originating from the Columbia River (inclusive of Redfish Lake sockeye) and the Washington coast (inclusive of Lake Ozette sockeye) north of southeast Alaska during any time of the year, indicating that if fish from these rivers occur in more northern areas of the Gulf of Alaska, they occur off the continental shelf. In the Gulf of Alaska, Echave et al. (2012b) documented that the distribution of juvenile sockeye salmon is generally contained to the continental shelf. Immature sockeye are distributed from the nearshore waters to the U.S. EEZ boundary throughout the entire Gulf (Echave et al. 2012b). Similarly, mature sockeye salmon occur in relatively low abundances extending from coastal waters to the U.S. EEZ boundary (Echave et al. 2012b). Myers et al. (1996a) documented maturing sockeye salmon from Washington and the Columbia River in offshore areas of the Gulf of Alaska.

Through observations of 883 purse seine net sets off the U.S. west coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids. Walker et al. (2007) recorded the vertical distribution of salmonids in North Pacific Ocean using data storage tags. The authors found that the average depth for sockeye was three meters, though the species was found down to 83 meters.

6.7.2.1.5 Steelhead

Steelhead are thought to rely heavily on offshore marine waters for feeding, with high seas tagging programs indicating steelhead make more extensive migrations offshore in their first year than any other Pacific salmonids (Quinn and Myers 2004). Commercial fisheries catch data indicate similar trends (Quinn and Myers 2004). The species spends approximately 1 to 3 years in freshwater, then migrates rapidly through estuaries, bypassing coastal migration routes of other salmonids, moving into oceanic offshore feeding grounds (Daly et al. 2014; Quinn and Myers 2004). Light et al. (1989) mapped the ocean distribution of steelhead in the North Pacific using catch per unit effort data from U.S., Canadian, USSR, and Japanese research vessels fishing with purse seines, gill nets, and longlines. Steelhead were distributed across the North Pacific throughout the year, but were in higher abundance closer to the US and Canadian coasts in spring and winter, and more evenly distributed in summer and fall. Steelhead trout hatched in freshwater streams in the Pacific Northwest are known to occur in Alaska marine waters during their juvenile or adult life stages (NMFS 2015a). McKinnell et al. (1997) assessed the distribution of North American hatchery steelhead stock in the Gulf of Alaska and Aleutian Islands using coded wire tag mark and recapture data collected by the NMFS Auke Bay Laboratories in Juneau, Alaska, and the Pacific Biological Station in Nanaimo, British Columbia, 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 the northern and southern Gulf of Alaska and the Aleutian Islands. These

studies indicate that although steelhead from the ESUs reviewed in this opinion are indeed present in Gulf of Alaska waters.

Tagging and diet studies indicate that adult and juvenile steelhead are surface oriented, spending most of their time in the upper portions of the water column (Daly et al. 2014). Walker et al. (2007) summarized information from a series of studies off British Columbia looking at the vertical distribution of steelhead and found the species spends 72 percent of its time in the top 1 m of the water column, with few movements below 7 m.

6.7.2.1.6 Estimation of Take

The preceding sections present the information we relied on to determine whether adult or juvenile ESA-listed salmonids are likely to co-occur with U.S. Navy explosive activities in the Gulf of Alaska TMAA. Table 34 summarizes this information and presents our conclusions on which salmonid species and life stages are likely to co-occur with Navy explosive activities. With the exception of a low number (n=6) of relatively small ordnance (5 to 10 lbs), Navy explosions occurring in the Gulf of Alaska TMAA will occur between 40 and 100 nm off the coast of Alaska, off the continental shelf.

Table 34. Expected co-occurrence of ESA-listed salmonids and Navy explosive activities in the GOA TMAA.

Species	Life Stage	Ordnance co-occurrence		Primary reference(s)
		E5 – E12 (off continental shelf)	E5 (on continental shelf)	
Chinook	Juvenile	No	No	Echave et al. 2012; Trudel et al. 2009
	Immature/maturing	No	Yes	Echave et al. 2012; Masuda et al. 2015
Coho	Juvenile	No	Yes	Echave et al. 2012
	Immature/maturing	Yes	Yes	Echave et al. 2012
Chum	Juvenile	No	Yes	Echave et al. 2012
	Immature/maturing	Yes	Yes	Echave et al. 2012
Sockeye	Juvenile	No	No	Echave et al. 2012; Tucker et al. 2012
	Immature/maturing	Yes	Yes	Echave et al. 2012
Steelhead	Juvenile	Yes	No	McKinnell et al. 1997; Light et al. 1989
	Immature/maturing	Yes	No	McKinnell et al. 1997; Light et al. 1989

For each ESA-listed salmon ESU and steelhead DPS, we estimated a density of animals in the action area based on information regarding the species' distribution and abundance. The 2015 biological opinion analyzing the effects of NMFS Southwest Fisheries Science Center (SWFSC) proposed research activities on ESA-listed species compiled recent information on salmon and steelhead abundance (i.e., outmigrants and adults) (NMFS 2015b). This information is presented in Table 35.

Table 35. Summary of estimated annual abundance of salmonids (NMFS 2015b). Abundance estimates for each ESU and DPS are divided into natural, listed hatchery intact adipose, and listed hatchery adipose clip[6].

Species	Life Stage	Natural Origin	Listed Hatchery Intact Adipose[1]	Listed Hatchery Adipose Clip[1]
Snake River fall Chinook	Adult[1]	14,438	30,475	-
Snake River spring/summer Chinook	Adult[1]	20,422	60,058	-
Lower Columbia River Chinook	Adult[1]	13,594	22,868	-
Upper Willamette River Chinook	Adult[1]	11,061	38,135	-
Upper Columbia River spring Chinook	Adult[1]	3,170	5,887	-
Puget Sound Chinook	Adult	18,127	11,089	-
Hood Canal summer run chum	Adult	17,556	3,452	-
	Smolt	3,072,420	275,000	-
Columbia River chum	Adult	12,239	428	-
	Smolt	2,978,550	391,973	-
Oregon Coast coho	Adult	192,431	1,753	-
	Parr	13,470,170	60,000	0
Lower Columbia River coho	Adult	10,957	208,192	-
	Smolt	839,118	299,928	8,637,196
Lake Ozette sockeye	Adult	1,683	33	-
Snake River sockeye	Adult	-	-	1,373*
Upper Columbia River steelhead	Adult	2,728	7,936	-
	Smolt	286,452	175,528	658,692
Snake River Basin steelhead	Adult	46,336	139,528	-
	Smolt	1,399,511	971,028	3,075,195
Lower Columbia River steelhead	Adult	11,117	23,000	-
	Smolt	447,659	2,428	1,025,729
Upper Willamette River steelhead	Adult	6,030	-	-
	Smolt	215,847	-	-
Middle Columbia River steelhead	Adult	24,127	2,724	-
	Smolt	540,850	426,556	347,113
Puget Sound steelhead	Adult	13,621	994	-
	Smolt	1,668,371	64,000	155,897

[1] **We do not have separate estimates for adult adipose fin-clipped and intact adipose fin hatchery fish.**

The 2015 SWFSC biological opinion only presented run-size estimates for fish returning to their natal rivers to spawn as a quantification of adults. The number of returning adults is an underestimate of the number of post-juvenile fish that will occur in the oceanic environment since most Chinook, chum, sockeye salmon and steelhead spend two to four years foraging and maturing in the ocean environment before returning to spawn. Coho salmon typically return to spawn at age three and thus spend approximately two years at sea before returning to freshwater to spawn. Information is not available for all ESA-listed salmon and steelhead DPSs to estimate the total oceanic abundance of these species (PFMC 2015). Therefore, we multiplied the number of returning adults for each ESU or DPS by the average number of years the species spends at sea before returning to spawn, in order to account for all age classes of fish that would be expected in the oceanic environment (i.e., three years for Chinook, chum, sockeye, and steelhead; two years for coho). We recognize that since this methodology is based on the number of returning adults, it does not account for individuals that die before returning to spawn. However, this does not inhibit our ability to accurately assess jeopardy and determine whether or not to expect any population level effects from this action because we are assessing jeopardy and the potential for any population level effects by comparing mortality from this action to the number of returning adults (which is generally how salmon and steelhead abundance and trends are tracked).

Additionally, we relied on smolt estimates (number of juveniles outmigrating from their natal rivers) as a proxy for juvenile abundance. However, actual abundance of juvenile salmonids in the ocean, particularly in areas far from their natal streams such as the Gulf of Alaska, would be much lower due to high mortality early in the marine phase of salmonid migrations. Because we did not have estimates for each ESU/DPS of survival from outmigration to arrival in the Gulf of Alaska, we did not refine this estimate further. This methodology results in an overestimate of juvenile salmonid abundance in the Gulf of Alaska. However, this does not inhibit our ability to accurately assess jeopardy and determine whether or not to expect any population level effects from this action because we are assessing jeopardy and the potential for any population level effects by comparing mortality from this action to the number of outmigrants (which is one way salmon and steelhead abundance and trends are tracked).

Once we estimated the ocean abundance of maturing/adult and juvenile fish from each ESU/DPS, we estimated a density based on expected distribution in the marine environment of each species. The following bullet points explain how we estimated the geographic distribution of each species and life stage.

- Chinook: The north-south oceanic distribution of Chinook was based on results presented in Masuda et al. (2015b).
    - Juvenile: As noted in Table 34, we determined that ESA-listed juvenile Chinook salmon are not likely to co-occur with Navy explosive activities in the Gulf of Alaska TMAA.

  - Maturing/Adult: Off the continental United States, Chinook primarily reside on the continental shelf (e.g.,(Bellinger et al. 2015), so we used the shelf break as the westward boundary for this species' distribution in this area. In the northern Gulf of Alaska, we used information presented in Echave et al. (2012b) to determine the geographic distribution of these species.
- Coho: The north-south oceanic distribution of coho was based on results presented in Masuda et al. (2015b).
  - Juvenile: Off the continental United States, coho primarily reside on the continental shelf (e.g.,(Bellinger et al. 2015), so we used the shelf break (the shelf break was defined as the 200 meter depth contour; (Landry and Hickey 1989)]) as the westward boundary for this species' distribution in this area. In the northern Gulf of Alaska, juvenile coho salmon predominantly occur in coastal waters, throughout the continental shelf and slope (Echave et al. 2012b; Quinn 2005), so we used the shelf break as the boundary for this species' distribution in this area.
  - Maturing/Adult: Off the continental United States, coho primarily reside on the continental shelf (e.g.,(Bellinger et al. 2015), so we used the shelf break as the westward boundary for this species distribution in this area. In the northern Gulf of Alaska, we used information presented in Echave et al. (2012b) to determine the geographic distribution of these species.
- Chum: The north-south distribution for chum was based on available tagging data for fish from Washington and Oregon which indicates that the majority of fish from these areas do not move north of the Gulf of Alaska (Myers et al. 1996a; Neave et al. 1976).
  - Juvenile: Chum migrate north and west once they leave their river of origin (Byron and Burke 2014; Quinn 2005) and are generally found on the continental shelf, inshore of 37 km from the coast (Pearcy and Fisher 1990). Echave et al. (2012b) found that juvenile chum are generally restricted to the continental shelf in the Gulf of Alaska. Therefore, the geographic distribution of juvenile chum was estimated to be the area of the continental shelf from each ESU's river of origin north through the continental shelf in the Gulf of Alaska.
  - Maturing/Adult: Chum geographic distribution for this species was estimated based on the ocean migration of the species from British Columbia, Washington, and Oregon, as determined from tagging data presented in Neave et al. (1976) and Myers et al. (1996a).
- Sockeye
  - Juvenile: As noted in Table 34, we determined that ESA-listed juvenile sockeye salmon are not likely to co-occur with Navy explosive activities in the Gulf of Alaska TMAA.
  - Maturing/Adult: We used the same geographic distribution for sockeye as we did for chum because in general, it is thought that sockeye follow a similar migration pattern once they enter the ocean (Byron and Burke 2014; Quinn 2005).

- Steelhead: For maturing adult and juvenile steelhead, we relied on the geographic ocean distribution of the species described in Light et al. (1989). We used the distribution of the species during autumn, the season during which they would be expected to occur in the action area at highest densities[7]. This resulted in a higher species density that would have been estimated in other seasons, and therefore likely overestimates impacts to steelhead for detonations that occur in spring or summer. Using this distribution is more conservative for the species since we are assessing impacts to the species should a worst-case scenario be implemented (i.e., training activities occurring in autumn when the species is more densely distributed).

When estimating the geographic distribution of each species/life stage, we erred on the side of a more narrow geographic distribution when presented with uncertainty. This resulted in higher fish densities and take estimates than would have otherwise been estimated. This is a more conservative approach for the species as we assess the potential for any population level effects and jeopardy on a relatively high estimate of take. Based on the distribution information presented above, we used ArcMap version 10.2.1 (ESRI, Redlands, CA) to determine the area (km^2) of habitat in the offshore portion of the action area that each species ESU or DPS is known to occupy (Table 36). It is worth noting that when estimating densities of each ESU/DPS in the action area, we distributed individuals equally throughout the geographic range of each species. In reality, densities of ESA-listed salmonids in the TMAA would likely be lower than our estimates because the TMAA is at the northern edge of most of these species' range. Higher densities would be expected much closer to each species' natal watersheds.

[7] Light et al. (1989) indicates that steelhead distribution is most geographically condensed in winter. However, during this time of the year according to Light et al. (1989), species distribution is not expected to overlap with the action area. Further, pursuant to the MMPA incidental take rule and LOA, the Navy will only be authorized to conduct training activities between April and October.

Table 36. Habitat area (distribution) used for each ESU/DPS (km^2) in the offshore marine environment.

Species	Life Stage	Marine Habitat Area (km^2)
Snake River fall Chinook	Adult	3,017,298
Snake River spring/summer Chinook	Adult	3,017,298
Lower Columbia River Chinook	Adult	5,685,709
Upper Willamette River Chinook	Adult	5,685,709
Upper Columbia River spring Chinook	Adult	5,123,382
Puget Sound Chinook	Adult	5,123,382
Oregon Coast and Lower Columbia River coho	Adult	3,730,166
	Juvenile	1,205,661
Hood Canal summer-run chum	Adult	4,442,732
	Juvenile	1,162,534
Columbia River chum	Adult	4,442,732
	Juvenile	1,188,149
Lake Ozette and Snake River sockeye	Adult	4,442,732
Steelhead (all DPSs)	Adult/juvenile	8,821,318

We then used the range to effects values developed for this consultation to calculate an area around each detonation that would result in mortality (defined as 1 percent injury) and injury (defined as the distance from the detonation where no injury would occur; we consider this the point of onset injury). We multiplied this area of injury or mortality by the density of each species to determine the number of individual fish from each ESU or DPS that would be expected to die or be injured from each detonation (in order to estimate the number of fish injured, the area of mortality was subtracted from the area of injury estimate; this ensured we did not double count). We then multiplied this result by the number of detonations expected for each explosive bin to get a total number of fish (juvenile or adult) that would be expected to die or be injured annually from each explosive bin.

Results from these calculations are presented in Table 37, Table 38, and Table 39. Table 37 gives estimates for hatchery fish with the adipose fin intact, Table 38 gives estimates for hatchery fish with an adipose clip, and Table 39 gives estimates for the number of wild fish. All estimates are the annual number of injured or killed fish. If a species or life stage of a particular ESU/DPS is not included in the table, this indicates no injuries or mortalities were estimated to occur. For, example, we estimated that no ESA-listed Chinook or sockeye salmon would be injured or killed by Navy explosives, so estimates for these species are not included in the tables.

Table 37. Estimated annual number of ESA-listed salmonids (hatchery fish w/adipose fin intact) that would die or be injured by explosive activities in the action area.

Species	Life stage	ESU/DPS	Mortality	Injury
Coho	Adult	Oregon coast - T	0.01	0.01
	Juvenile		0.02	0.03
	Adult	Lower Columbia River - T	0.90	1.67
	Juvenile		0.09	0.16
Chum	Adult	Hood Canal - T	0.03	0.05
	Juvenile		0.11	0.19
	Adult	Columbia River - T	0.00	0.01
	Juvenile		0.16	0.27
Steelhead	Adult	Upper Columbia River - E	0.03	0.05
	Juvenile		0.48	0.80
	Adult	Snake River basin - T	0.54	0.95
	Juvenile		2.63	4.41
	Adult	Lower Columbia River - T	0.10	0.17
	Juvenile		0.01	0.01
	Adult	Middle Columbia River - T	0.01	0.02
	Juvenile		1.16	1.94
	Adult	Puget Sound - T	0.00	0.01
	Juvenile		0.17	0.29

Table 38. Estimated annual number of ESA-listed salmonids (hatchery fish w/adipose clip) that would die or be injured by explosive activities in the action area.

Species	Life stage	ESU/DPS	Mortality	Injury
Coho	Juvenile	Lower Columbia River - T	2.50	4.52
Steelhead	Juvenile	Upper Columbia River - E	1.78	2.99
	Juvenile	Snake River basin - T	8.33	13.96
	Juvenile	Lower Columbia River - T	2.78	4.66
	Juvenile	Middle Columbia River - T	0.94	1.58
	Juvenile	Puget Sound - T	0.42	0.71

Table 39. Estimated annual number of ESA-listed salmonids (natural fish) that would die or be injured by explosive activities in the action area.

Species	Life stage	ESU/DPS	Mortality	Injury
Coho	Adult	Oregon coast - T	0.83	1.54
	Juvenile		3.89	7.05
	Adult	Lower Columbia River - T	0.05	0.09
	Juvenile		0.24	0.44
Chum	Adult	Hood Canal - T	0.13	0.23
	Juvenile		1.27	2.17
	Adult	Columbia River - T	0.09	0.16
	Juvenile		1.21	2.06
Steelhead	Adult	Upper Columbia River - E	0.01	0.02
	Juvenile		0.78	1.30
	Adult	Snake River basin - T	0.18	0.32
	Juvenile		3.79	6.35
	Adult	Lower Columbia River - T	0.05	0.08
	Juvenile		1.21	2.03
	Adult	Upper Willamette River - T	0.03	0.05
	Juvenile		0.58	0.98
	Adult	Middle Columbia River - T	0.10	0.18
	Juvenile		1.47	2.45
	Adult	Puget Sound - T	0.06	0.10
	Juvenile		4.52	7.57

6.7.2.2 *Forage Fish Community*

Sound pressure waves from explosive activities will reach levels that would be expected to injure or kill forage fish. This creates an indirect effect to salmonids that prey on forage fish. While juvenile or sub-adult salmonids prey on larval or juvenile forage fish for some portion of their prey base, adult salmonids utilize forage fish as a major component of their prey base. Direct mortality of forage fish from explosive activities may cause a temporary and localized reduction in available prey for adult salmonids that may attempt to forage in the action area. However, adverse effects to listed fish are not expected to occur from this because the reduction in prey availability will occur in limited spatial and temporal scales, and prey resources will continue to be available to salmonids outside of the immediate area of injury and mortality zone of each explosion. Forage fish abundance will return to the area when explosive activities cease.

6.7.2.3 *Potential for TTS and Behavioral Reactions*

Some individual ESA-listed fish may experience TTS as a result of Navy impulsive acoustic stressors. Temporary threshold shift is short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006a). Further, the fish species considered in this opinion lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. To our knowledge, no studies have examined the fitness implications when a fish, without notable hearing specialization, experiences TTS. Popper et al. (2014b) suggested that fishes experiencing TTS may have a decreased ability to communicate, detect predators or prey, or assess their environment. However, these species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014b). Additionally, hearing is not thought to play a role in salmon or steelhead migration (e.g., (Putnam et al. 2013)). Because any TTS experienced would be temporary and the ESA-listed fish species considered in the opinion are able to rely on alternative mechanisms for these essential life functions, instances of TTS would not kill or injure any fish, nor would any such instances create the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. For these reasons, any TTS experienced by fish would not rise to the level of take and would not have fitness level consequences at the individual or population level. Therefore, the effect of exposure to acoustic stressors that may result in TTS is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

Additionally, based on the research presented in Section 6.4.5, behavioral effects resulting from reactions to sound created by the explosions will likely be temporary (e.g., a startle response lasting seconds). We do not expect these reactions to have any measurable effects on any individual's fitness because we expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each explosion. Similar to

instances of TTS, we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering, and such reactions would not rise to the level of take. Therefore, the effect of detonations that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

6.7.2.4 *Assessing the Effect of Take on ESA-listed Fish Populations*

It is important to note that the area of potential effect is the distance from the detonations to the point where no injury would occur. We expect the majority of fish injuries to be minor and recoverable, although some injuries may lead to internal bleeding, barotrauma, and death. Fish that are near the range to mortality may be more likely to incur a more severe injury that could lead to mortality with time (e.g., internal bleeding, barotrauma, higher susceptibility to predation). Because we do not have information to estimate what proportion of injured individuals will die, for the purposes of assessing jeopardy and potential population-level effects, we will assume all fish that may be injured may die. Additionally, it is important to emphasize that the mortality threshold is based on the distance from the detonation that would be expected to result in one percent of fish exposed dying. Therefore, this is a conservative method to estimate injury and mortality and resulting estimates of take are also conservatively high.

Many of the annual take estimates resulted in fractional numbers of fish being killed or injured. Since the Navy cannot kill or injure fractional numbers of fish, we evaluated the effect of this level of anticipated take over a 5 year period (e.g., if 0.2 fish were estimated to be killed in one year, 1 fish was estimated to be killed in 5 years). If take estimates over this 5 year period were still <1 fish killed or injured, but ≥0.1, this estimate was rounded up to one. If over the 5 year period any estimates were <0.1, we determined injury or mortality was not reasonably certain to occur and therefore, discountable. Expressing take over a 5 year period allows us to consider the effects of low levels of take that may not be expected to occur in a single year, but would be likely over a longer period of time. Expressing take over a five-year period does not risk overlooking effects that might appear only over a longer period of time because our analysis was structured such that the magnitude of the percentage of the population affected would not be expected to change over time unless Navy activity levels changed.

In order to evaluate the potential for this level of take from Navy training activities in the Gulf of Alaska to appreciably reduce the likelihood of the survival and recovery of any of the ESA-listed fish species considered in this opinion, we compared the level of take anticipated to population abundance estimates for each ESU or DPS. Results from this analysis are presented in terms of a percentage of each ESU or DPS that would be expected to die or be injured from Navy activities over a 5 year timeframe, assuming activity levels are maintained. Results are presented in Table 40, Table 41, and Table 42. If a species or life stage of a particular ESU/DPS is not included in

the table, this indicates no injuries or mortalities were estimated to occur over the 5 year time period. We estimated that no ESA-listed Chinook salmon from any ESA-listed ESUs would be injured or killed by Navy explosives, so estimates for these species are not included in the tables.

Table 40. Number and percent of each ESA-listed fish ESU/DPS (hatchery produced with intact adipose) that would be killed or injured in the action area over a 5-year period. --Indicates there are no fish from this ESU/DPS released from hatcheries with an intact adipose fin that would be affected.

Species	Life stage	ESU/DPS	Mortality (5 years)	Injury (5 years)	Percent of ESU/DPS (intact adipose) Killed or Injured
Coho	Juvenile	Oregon coast - T	0	1	<0.001
	Adult	Lower Columbia River - T	5	9	0.001
	Juvenile		1	1	<0.001
Chum	Adult	Hood Canal - T	1	2	0.006
	Juvenile		3	4	0.001
	Juvenile	Columbia River - T	3	5	<0.001
Steelhead	Adult	Upper Columbia River - E	1	2	0.003
	Juvenile		3	4	0.001
	Adult	Snake River basin - T	3	5	<0.001
	Juvenile		14	22	0.001
	Adult	Lower Columbia River - T	1	1	0.001
	Adult	Middle Columbia River - T	0	1	0.002
	Juvenile		6	10	0.001
	Juvenile	Puget Sound - T	1	2	0.001

Table 41. Number and percent of each ESA-listed fish ESU/DPS (hatchery produced with adipose fin-clip) that would be killed or injured in the action area over a 5-year period. --Indicates there are no fish from this ESU/DPS released from hatcheries with an adipose fin-clip that would be affected.

Species	Life stage	ESU/DPS	Mortality (5 years)	Injury (5 years)	Percent of ESU/DPS (adipose fin-clip) Killed or Injured
Coho	Juvenile	Lower Columbia River - T	13	23	<0.001
Steelhead	Juvenile	Upper Columbia River - E	9	15	0.001
	Juvenile	Snake River basin - T	42	70	0.001
	Juvenile	Lower Columbia River - T	14	24	0.001
	Juvenile	Middle Columbia River - T	5	8	0.001
	Juvenile	Puget Sound - T	3	4	0.001

Table 42. Number and percent of each ESA-listed fish ESU/DPS (naturally produced) that would be killed or injured in the action area over a 5-year period. --Indicates there are no naturally produced fish from this ESU/DPS that would be affected.

Species	Life stage	ESU/DPS	Mortality (5 years)	Injury (5 years)	Percent of ESU/DPS (naturally produced) Killed or Injured
Coho	Adult	Oregon coast - T	5	8	0.001
	Juvenile		20	36	<0.001
	Adult	Lower Columbia River - T	1	1	0.002
	Juvenile		2	3	<0.001
Chum	Adult	Hood Canal - T	1	2	0.001
	Juvenile		7	11	<0.001
	Adult	Columbia River - T	1	1	0.001
	Juvenile		6	11	<0.001
Steelhead	Juvenile	Upper Columbia River - E	4	7	0.001
	Adult	Snake River basin - T	1	2	<0.001
	Juvenile		19	32	0.001
	Adult	Lower Columbia River - T	1	1	0.001
	Juvenile		7	11	0.001
	Adult	Upper Willamette River - T	1	1	0.002
	Juvenile		3	5	0.001
	Adult	Middle Columbia River - T	1	1	0.001
	Juvenile		8	13	0.001
	Adult	Puget Sound - T	1	1	0.001
	Juvenile		23	38	0.001

We determined that no greater than 0.006 percent (i.e., hatchery adult Hood Canal summer-run chum with an intact adipose fin) of any ESA-listed salmonid ESU or DPS would be injured or killed from Navy activities. Most salmon ESUs and steelhead DPSs would be affected at much lower levels. Therefore, even if the Navy conducts the maximum amount of training and testing analyzed in this opinion, only a tiny fraction of the population of each ESA-listed salmonid fish species would be affected by this action. Also important for our analysis and conclusions is that true densities of ESA-listed salmonids in the TMAA would likely be lower than the estimates we used in our analysis because the TMAA is at the northern edge of most of these species' range and we assumed equal distribution of all individuals throughout their geographic range. In reality, higher densities would be expected much closer to each species' natal watersheds (i.e., off the coast of Oregon or Washington) and lower densities would be expected near the edges of each species' distribution (e.g., the Gulf of Alaska). This likely results in an overestimate of the number of ESA-listed salmonids injured or killed by the proposed action. Also of note, some of the fish killed or injured would be from hatcheries, which are considered to be of less conservation value for the ESU compared to the natural origin population (NMFS 2015b).

We assume for the purposes of this analysis that the current annual level of activities and impacts will continue into the reasonably foreseeable future, and we have made our jeopardy determination on that basis. Our estimate of the number of individuals affected is a function of juvenile and adult abundance estimates, habitat area affected by Navy explosives, and the oceanic distribution of each ESU or DPS. While the habitat area affected and the oceanic distribution of each ESU or DPS are not expected to change (unless the Navy uses fewer explosives in which case effects would be less), fish abundance may change significantly over even a short period of time due to a variety of factors including changes in fisheries harvest strategies, habitat conditions, and other changes in the environmental baseline that may impact reproductive success. For this opinion, we relied on adult and juvenile salmonid abundance estimates as compiled in NMFS' 2015 biological opinion on Southwest Fisheries Science Center activities. We expect that these abundance estimates will be updated on a continuous basis as monitoring information is collected each year. However, our analysis of the potential effect of the proposed action on ESA-listed salmonid populations (i.e., percent of the population affected) is a function of the amount of oceanic habitat within the geographic distribution of each species that would be affected. Our jeopardy determination is based on this analysis. Therefore, even though salmonid abundance estimates change through time, unless Navy activity levels increase or new information suggests higher-level impacts than were analyzed in this opinion (e.g., from updated criteria for predicting explosive impacts on fish), both of which would likely be triggers for reinitiation of consultation, the effects of the proposed action on ESA-listed salmonid populations is not expected to change over time. Interannual changes in ESA-listed salmonid abundance estimates would not alter our conclusions on the effect of the proposed action on ESA-listed salmonid populations.

Additionally, because MMPA take authorization is limited to five years, we will need to reassess the Navy's proposed action under both the MMPA and ESA prior to the expiration of the proposed take authorization, and not less than every five years thereafter. These periodic reassessments will allow us to update the environmental baseline and assess any changes in ESA-listed fish species status. In this manner, we will be able to ensure that our assessment of the effect of the Navy's ongoing activities, and the resultant impacts on listed fish species are up to date and that no impacts are omitted from consideration.

In addition to quantifying the number of individuals that we expect to be injured or killed and the corresponding proportion of each population that would be impacted, we assessed the likelihood that U.S. Navy underwater detonations could injure or kill a significant proportion of individuals from a single (potentially small) population during a single detonation or series of detonations. This discussion relates to whether or not Navy activities would be expected to impact salmonid species structure and diversity. However, as stated throughout this opinion, including in the *Status of Listed Resources* section, the ESA-listed salmonids that are likely to be impacted by these activities exhibit significant life history variation, even within populations (e.g., Bellinger et al. 2015; Burke et al. 2013; Weitkamp and Neely 2002). For example, Chinook salmon from the Skagit River, a portion of the Puget Sound ESU, are categorized into six populations (Zimmerman et al. 2015). The six populations are named according to the time of river entry (spring, summer, and fall) and location of spawning. Additionally, out-migrant life history varies with individuals out-migrating as fry, sub-yearling parr, or yearling smolts (Zimmerman et al. 2015). Because of this life history variability, we would expect Chinook salmon from the Skagit River to be widely dispersed in space and time in the marine environment. Additionally, with the exception of chum, salmonids are not known to school (Pearcy and Fisher 1990) and occur in relatively low densities, decreasing the likelihood that a significant number of individuals from a would be impacted. Therefore, we do not expect a large proportion of individuals from a single population to be grouped together in space and time such that they would be killed or injured by a single detonation or series of detonations. This conclusion applies specifically to the structure and diversity of populations within any given ESU or DPS, as we would not expect impacts to be focused on individuals from any one population from a specific river of origin.

In summary, we conclude that the level of incidental take of ESA-listed fish species during Navy training activities each year that has been proposed and considered in this opinion represents a very small reduction in abundance that is not likely to appreciably impact any ESA-listed fish populations over time. Because this level of incidental take is not expected to appreciably impact ESA-listed fish species population levels over time, we can also conclude that the proposed action will not effect productivity of any ESA-listed salmonid species. We also conclude that the structure and diversity of ESA-listed fish populations will not be affected by this limited amount of take that should be distributed across populations across their ranges and through time. We have generally identified and considered the worst case scenario of potential injury and mortality for each ESA-listed fish population considered in this opinion, where applicable, leading to the

most conservative estimates of expected take. We have concluded that the proposed action will have a very small effect on the species' abundance, and will not affect spatial structure, diversity, or prudctivity at all. As a result, we conclude that the proposed action would not reasonably be expected to appreciably reduce the likelihood of the survival and recovery of any of the ESA-listed fish species that may be affected by this proposed action.

6.8 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. Most of the action area includes federal military reserves or is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of Google and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require federal authorization or funding and is reasonably certain to occur. As a result, NMFS is not aware of any actions of this kind that are likely to occur in the action area during the reasonably foreseeable future.

7 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 6) to the environmental baseline (Section 4) and the cumulative effects (Section 6.8) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 4) and the *Environmental Baseline* in the action area (Section 5).

The following discussions separately summarize the probable risks the proposed action poses to threatened and endangered species and critical habitat that are likely to be exposed. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this opinion.

Our effects analyses identified the probable risks the Navy training activities and issuance of an MMPA rule and LOA to authorize take of marine mammals would pose to ESA-listed individuals that will be exposed to these actions. We measure risks to individuals of endangered or threatened species using changes in the individuals' "fitness" or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed animals exposed to an action's effects to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise. As a result, if we conclude that listed animals are not likely to experience reductions in their fitness, we would conclude our assessment. If, however, we conclude that listed animals are likely to experience reductions in their fitness, we would assess the consequences of those fitness reductions for the population or populations the individuals in an action area represent.

The activities the Navy conducts in the Gulf of Alaska TMAA introduce stressors into the marine and coastal ecosystems annually during a 21 day exercise period between April and October. The stressors include: low, mid, and high-frequency active sonar from surface vessels, torpedoes, and dipping sonar; shock waves and sound fields associated with underwater detonations, acoustic and visual cues from surface vessels as they move through the ocean's surface, and sounds transferred into the water column from fixed-wing aircraft and helicopters. Repeatedly exposing endangered and threatened marine animals to each of these individual stressors could pose additional risks as the exposures accumulate over time. Repeated exposures are discussed in more detail in sections 6.3.11 and 6.4.6. Also, exposing endangered and threatened marine animals to this suite of stressors could pose additional risks as the stressors interact with one another or with other stressors that already occur in those areas. More importantly, endangered and threatened marine animals that occur in the action area would be exposed to combinations of stressors produced by Navy activities at the same time they are exposed to stressors from other human activities and natural phenomena. We recognize these interactions might have effects on endangered and threatened species that we have not considered; however, the data available do not allow us to do more than acknowledge the possibility.

For the purposes of this opinion, we assume that all of these activities in the Gulf of Alaska TMAA and associated impacts will continue into the reasonably foreseeable future at the levels set forth in the 2016 Final SEIS/OEIS and MMPA rule (see section 2.5 of this opinion). To address the likelihood of long-term additive or accumulative effects, we first considered (1) stressors that accumulate in the environment, and (2) effects that represent either the response of individuals, populations, or species to that accumulation of stressors.

Sound does not permanently accumulate in the environment; therefore, an accumulative effects analysis of this stressor is not warranted. However, repeated exposure of individuals to acoustic stress can cause auditory fatigue, hearing loss, or other more long-term behavioral effects. If

sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks), it would be possible for individuals confined to a specific area to be exposed to acoustic stressors (e.g., MFA sonar) multiple times during a relatively short time period. However, we do not expect this to occur as we would expect individuals to move and avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120dB) and in the open water environment of the TMAA, animals will not be geographically constrained from doing so. For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect equivalent foraging to be available in close proximity. Furthermore, Navy training in the Gulf of Alaska would only persist up to a maximum of one exercise period of 21 days between April and October on an annual basis. Consequently, the Navy's Gulf of Alaska activities do not create conditions of chronic, continuous underwater noise and are unlikely to lead to habitat abandonment or long-term hormonal or physiological stress responses in marine mammals.

Although Goldbogen et al. (2013) speculates that "frequent exposures to mid-frequency anthropogenic sounds may pose significant risk to the recovery rates of endangered blue whale populations," the authors acknowledge that the actual responses of individual blue whales to simulated mid-frequency sonar documented in the study "typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Moreover, the most significant response documented in the study occurred not as a result of exposure to simulated mid-frequency sonar but as a result of exposure to pseudo-random noise. Therefore, the overall weight of scientific evidence indicates that substantive behavioral responses by mysticetes, if any, from exposure to mid-frequency active sonar and other active acoustic sources evaluated in this opinion are likely to be temporary and are unlikely to have any long-term adverse impact on individual animals or affected populations. Even if sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks during major training activities) such as the 21 day training period between April and October each year, we would expect that some individual whales would avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect equivalent foraging habitat to be available in close proximity. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

Further, establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013; Read et al. 2014). The difficulty in assessing the effects of sounds individually and cumulatively on marine species is the confounding nature of indirect effects, age class, prior experience, behavioral state at the time of exposure, and that responses may be influenced by other non-sound related factors (Ellison et al. 2011; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014; Williams et al. 2014a).

Our assessment that the continuation of the Navy activities into the reasonably foreseeable future is unlikely to have any adverse additive or long-term impacts on the affected threatened or endangered species (assuming current levels of activity and no significant changes in the status of species or to the *Environmental Baseline*) is also consistent with the absence of any documented population-level or adverse aggregate impacts resulting from Navy activities to date, despite years of training in the Gulf of Alaska TMAA using many of the same systems (the Navy has trained in the action area since the early 1990s; it began using MFA sonar in 2011). Most of the training activities the Navy conducts in the Gulf of Alaska TMAA are similar, if not identical, to activities that have been occurring in the same locations for years.

Our regulations require us to consider, using the best available scientific data, effects of the action that are "likely" and "reasonably certain" to occur rather than effects that are speculative or uncertain. See 50 C.F.R. § 402.02 (defining to "jeopardize the continued existence of" and "effects of the action"). Our analysis and conclusions in this opinion are based on estimates of exposures and take assuming that the Navy conduct the maximum number of authorized training activities for the maximum number of authorized hours. The effects of the action in relation to the *Status of Listed Resources* and the *Environmental Baseline* are presented by each species below.

7.1 Blue Whale

Blue whales are present in the action area. Blue whales may be exposed to acoustic stressors associated with training activities between April and October in any given year. Blue whales found in the action area are recognized as part of the Eastern North Pacific stock. The acoustic analysis predicts that blue whales of the Eastern North Pacific stock may be exposed to sonar and other non-impulsive acoustic sources associated with training activities that may result in 9 instances of TTS and 38 behavioral reactions per year. As presented above for mysticetes in general, long-term consequences for individuals or the population would not be expected.

Blue whales could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of blue whales being exposed to impulsive sources associated with training activities that rise to the level of take.

While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on blue whale hearing so we assume that blue whale vocalizations are partially representative of their hearing sensitivities. Blue whales are not likely to respond to high-frequency sound sources associated with the proposed training activities because of their hearing sensitivities. Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007b; Stafford and Moore 2005a), recent research has indicated blue whales not only hear mid-frequency active sonar transmissions, in some cases they respond to those transmissions (Southall et al. 2011a).

Blue whales may hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context (Goldbogen et al. (2013) and Melcon et al. (2012)). However, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational MFA sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). As stated in Goldbogen et al. (2013) when summarizing the response of blue whales to simulated MFA sonar, "We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Goldbogen et al. (2013) also speculated that if this temporary behavioral response interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication that this is the case, particularly since unconsumed prey would still be available in the environment following the cessation of acoustic exposure and resumption of normal behaviors following instances of behavioral response including responses associated with TTS. Additionally, in instances of TTS, individuals would likely fully recover within 24 hours of exposure and resume normal behavioral activities.

During the Controlled Exposure Experiments (CEEs) used in Goldbogen et al. (2013), sound sources were deployed from a stationary source vessel positioned approximately 1,000 m from the focal animals, with one transmission onset every 25 seconds (Southall et al. 2012b). In contrast, most Navy sonar systems are deployed from highly mobile vessels or in-water devices which do not directly target marine mammals. Further, the typical duty cycle with most tactical anti-submarine warfare is lower than used in the CEEs described above, transmitting about once per minute (DoN 2015b). For example, a typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance

behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. (see section 6.3.11 of this opinion). Given the nature of training, any periodic or episodic exposure and response scenarios (including behavioral responses and TTS) would allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. In the event an individual is exposed to multiple sound sources that elicit a behavioral response (e.g., disruption of feeding) in a short amount of time, including instances of TTS, we do not expect these exposures to have fitness consequences as individuals will resume feeding upon cessation of the sound exposure and unconsumed prey will still be available in the environment.

Further, even if sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks during major training activities) such as the case with a 21 day training period each year, we would expect that some individual whales would avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, even temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect equivalent foraging to be available in close proximity. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

As described previously, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts. Further, recent evidence indicates that the Eastern North Pacific blue whale population has likely reached carrying capacity (Monnahan et al. 2014a). Navy training activities, including the use of MFA sonar, have been ongoing in the Gulf of Alaska since 2011 and in other areas where this population occurs (i.e., Navy training ranges off west coast of continental United States) for decades. Therefore, any potential impacts from Navy training, including the use of MFA sonar, on blue whales do not appear to have inhibited growth of the Eastern North Pacific blue whale population.

The 1998 blue whale recovery plan does not outline downlisting or delisting criteria. The recovery plan does list several stressors potentially affecting the status of blue whales in the North Pacific Ocean that are relevant to Gulf of Alaska activities including: vessel strike, vessel disturbance, and military operations (including sonar). At the time the recovery plan was published, the effects of these stressors on blue whales in the Pacific Ocean were not well documented, their impact on recovery was not understood, and no attempt was made to prioritize

the importance of these stressors on recovery. As described previously, anthropogenic noise associated with Gulf of Alaska activities is not expected to impact the fitness of any individuals of this species. No mortality of blue whales is expected to occur from Gulf of Alaska activities.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA Rule from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual blue whales in ways or to a degree that would reduce their fitness. We anticipate temporary behavioral responses and TTS, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

7.2 North Pacific Right Whale

North Pacific right whales will likely occur in portions of the action area and are predicted to be exposed to sonar or other active acoustic stressors that may result in one TTS and two behavioral reactions per year between April and October. However, as discussed further in section 6.6, we do not expect instances of harassment (i.e., behavioral or TTS) to result in fitness consequences for any affected individuals. Because we do not expect fitness consequences for any individual animals, we also do not expect fitness consequences for populations. These three instances of take could be to the same animal on subsequent days, or be the result of exposures to two or more animals. Recovery from a threshold shift (i.e., partial hearing loss) can take a few minutes to a few days, depending on the severity of the initial shift. Threshold shifts do not necessarily affect all hearing frequencies equally, so some threshold shifts may not interfere with an animal's hearing of biologically relevant sounds. North Pacific right whales could also be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero

instances of North Pacific right whales being exposed to impulsive sources associated with training activities that rise to the level of take. No injury or mortality is anticipated from the proposed action.

Additionally, it is worth noting that the Navy will not use surface ship hull mounted mid-frequency sonar or explosives during training within the portion of the NMFS-identified North Pacific right whale feeding area overlapping the TMAA in the June to September timeframe (see section 2.3.3.2 of this opinion). This mitigation measure is expected to minimize the potential effect of any North Pacific right whale take that may occur because it reduces the likelihood that take will occur in a known feeding area for the species. The typical location of Navy sonar and explosive use in the TMAA (i.e., in deep waters, away from the borders of the TMAA) further reduces the likelihood of a take occurring in the NMFS-identified North Pacific right whale feeding area.

The 2013 North Pacific right whale recovery plan sets criteria for the downlisting and delisting of this species. Both downlisting and delisting criteria include abatement of the threats that limit the continued growth of North Pacific right whale populations including environmental contaminants; reduced prey abundance or location due to climate change; ship collisions; and exposure to anthropogenic noise (NMFS 2013b). Of these, anthropogenic noise and ship collision are relevant to Navy Gulf of Alaska training activities. However, as described previously in section 6.3, 6.6.1, and the introduction to the Integration and Synthesis, anthropogenic noise associated with Gulf of Alaska activities is not expected to impact the fitness of any individuals of this species, or any other mysticetes. Further, we do not anticipate Navy vessels will strike any North Pacific right whales. Downlisting criteria for North Pacific right whales includes the maintenance of at least 1,000 mature, reproductive individuals with at least 250 mature females and 250 mature males in each population (eastern and western). To qualify for downlisting, each recovery population must also have no more than a one percent chance of extinction in 100 years.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA rule from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of North Pacific right whales in the wild by reducing the reproduction, numbers, or distribution of that species as currently listed, or currently proposed for listing, under the ESA. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of North Pacific right whales in the wild by reducing the reproduction, numbers, or distribution of that species as currently listed under the ESA. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual North Pacific right whales in ways or to a

degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated.

7.3 Humpback Whale – Mexico and Western North Pacific DPSs

Humpback whales could be in all portions of the action area and may be exposed to sonar and other active acoustic sources associated with training activities between April and October in any given year. Humpback whales from the threatened Mexico DPS, endangered Western North Pacific DPS, and Hawaii DPS, which is not listed under the ESA, could all occur in the action area.

The acoustic analysis predicts that humpback whales may be exposed to sonar or other active acoustic stressors that may result in 16 TTS and 53 behavioral reactions per year. Based on the information presented in Wade et al. (2016), the majority of humpbacks exposed to sonar or other stressors would be from the Hawaii DPS which is not listed under the ESA. We anticipate one instance of behavioral take and zero instances of TTS for the endangered Western North Pacific DPS and 5 instances of behavioral take and 2 instances of TTS for the threatened Mexico DPS. All other instances of behavioral harassment and TTS would be assigned to the Hawaii DPS.

Humpback whales could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of humpback whales being exposed to impulsive sources associated with training activities that rise to the level of take.

Recent observations of blue whale responses to the mid-frequency sonar sounds support the possibility that humpback whales, an ecologically, physiologically, and taxonomically similar species, may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on humpback whales. Considering information presented in this opinion, we consider humpback whales to be able to hear and respond to mid-frequency sonar similar to blue whales.

Discussion of the potential for repeated exposures is included in section 6.3.11 of this opinion. In this section we concluded that repeat exposures capable of eliciting a behavioral response to an individual over a brief period of time would be rare. Given the nature of training, any periodic or episodic exposure and response scenarios (including behavioral response and TTS) would allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding, especially since a large portion of the primary feeding grounds are outside of the TMAA. As described previously in this opinion, the available scientific

information does not provide evidence that exposure to acoustic stressors from Navy training activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

The 1991 humpback whale recovery plan recommended that populations grow to at least 60% of their pre-hunting abundance to be considered recovered, but did not identify specific numerical targets due to uncertainty surrounding historical abundance levels. As an interim goal, the plan suggested a doubling of population sizes within 20 years, which corresponds to an annual growth rate of 3.5%. Most humpback whale DPSs where trend data are available have an estimated annual population growth rate of > 3.5%, including the Western North Pacific and Mexico DPSs (Betteridge et al. 2015). The general increase in the number of humpback whales in the North Pacific (approximately 4.9%) suggests that the stress regime these whales are exposed to, including Navy activities in the Gulf of Alaska TMAA, have not prevented these whales from increasing their numbers. Humpback whales have previously been exposed to Navy training activities in the Gulf of Alaska TMAA, including vessel traffic, aircraft traffic, active sonar, and underwater detonations.

Threats known or suspected of impacting humpback whale recovery include hunting, commercial fishing stressors, habitat degradation, loss of prey species, ship collision, and acoustic disturbance. Of these, ship collision and acoustic disturbance are relevant to Gulf of Alaska activities. As described previously, anthropogenic noise associated with Navy Gulf of Alaska activities (e.g., from vessel traffic, sonar, and explosions) is not expected to impact the fitness of any individual humpback whales from the Western North Pacific or Mexico DPSs. There have been no documented humpback whale ship collisions with Navy vessels in the Gulf of Alaska and we do not believe that a Navy vessel strike of a humpback whale in the action area is reasonably likely to occur.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA rule from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Western North Pacific or Mexico DPS humpback whales by reducing the reproduction, numbers, or distribution of these species as listed under the ESA. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Western North Pacific or Mexico DPS humpback whales by reducing the reproduction, numbers, or distribution of those species as listed under the ESA. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual humpback whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual

whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of these species as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated.

7.4 Fin Whale

Fin whales could be present in all portions of the Gulf of Alaska TMAA and may be exposed to acoustic sources associated with training activities between April and October. The acoustic analysis predicts that fin whales may be exposed to sonar or other active acoustic stressors that may result in 350 TTS and 941 behavioral reactions per year during the April to October timeframe.

Fin whales could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of fin whales being exposed to impulsive sources associated with training activities that rise to the level of take.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of fin whales. However, recent observations of blue whale responses to mid-frequency sonar sounds support the possibility that this ecologically, physiologically, and taxonomically similar species may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on fin whales. Considering information presented in this opinion, we consider fin whales to be able to hear and respond to mid-frequency sonar as blue whales appear to.

Discussion of the potential for repeated exposures is included in section 6.3.11 of this opinion. In this section we concluded that repeat exposures capable of eliciting a behavioral response to an individual over a brief period of time would be rare. Given the nature of training, any periodic or episodic exposure and response scenarios (including behavioral responses and TTS) would allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Individuals exposed to acoustic stressors at levels resulting in TTS will likely fully recover within 24 hours of the exposure and resume normal behaviors including feeding. As described previously in this opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors is not expected to have population or species level impacts.

The 2010 fin whale recovery plan defines three populations by ocean basin (the North Atlantic, North Pacific, and Southern Hemisphere) and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated

with vessel collisions, reduced prey abundance due to overfishing and/or climate change, the possibility that illegal whaling or resumed legal whaling will cause removals at biologically unsustainable rates and, possibly, the effects of increasing anthropogenic ocean noise. Of these, anthropogenic noise and ship collision are relevant to Gulf of Alaska activities. As discussed previously, anthropogenic noise associated with Gulf of Alaska activities is not expected to impact the fitness of any individuals of this species, and we do not anticipate vessels associated with Navy training in the TMAA to strike any fin whales. Downlisting criteria for fin whales includes the maintenance of at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a 1 percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on fin whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA Rule from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual fin whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

7.5 Sei Whale

Sei whales may be exposed to sonar or other active acoustic stressors that may result in one TTS and five behavioral reactions between April and October of a given year. Sei whales could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of sei whales being exposed to impulsive sources associated with training activities that rise to the level of take.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of sei whales. However, recent observations of blue whale responses to the mid-frequency sonar sounds support the possibility that this ecologically, physiologically, and taxonomically similar species may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on sei whales. Considering information presented in this opinion, we consider sei whales to be able to hear and respond to mid-frequency sonar similar to blue whales.

Discussion of the potential for repeated exposures is included in section 6.3.11 of this opinion. In this section we concluded that repeat exposures capable of eliciting a behavioral response to an individual over a brief period of time would be rare. Given the nature of training, any periodic or episodic exposure and response scenarios (including behavioral response and TTS) would allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Individuals experiencing TTS are likely to fully recover within 24 hours of exposure and return to normal behaviors. As described previously in this opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

The 2011 sei whale recovery plan defines three recovery populations by ocean basin (the North Atlantic, North Pacific, and Southern Hemisphere) and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with vessel collisions, entanglement in active or derelict fishing gear, reduced or displaced prey abundance due to climate change, the possibility that illegal whaling or resumed legal whaling will cause removals at biologically unsustainable rates, and the effects of increasing anthropogenic ocean noise. Of these, anthropogenic noise and ship collision are relevant to Gulf of Alaska activities. As described previously, anthropogenic noise associated with Gulf of Alaska activities will not impact the fitness of any individuals of this species and we do not anticipate vessels associated with Navy training in the TMAA to strike any sei whales. Downlisting criteria for sei whales includes the maintenance of 1,500 mature, reproductive individuals with at least 250 mature females and 250 mature males in each ocean basin. To qualify for downlisting, each recovery population must also have no more than a 1 percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sei whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA regulations from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would

not be expected to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual sei whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

7.6 Sperm Whale

Sperm whales may be exposed to sonar or other active acoustic stressors associated with training activities that may result in zero TTS and 98 behavioral reactions from April to October of any given year. Sperm whales could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of sperm whales being exposed to impulsive sources associated with training activities that rise to the level of take.

Discussion of the potential for repeated exposures is included in section 6.3.11 of this opinion. In this section we concluded that repeat exposures capable of eliciting a behavioral response to an individual over a brief period of time would be rare. Given the nature of training, any periodic or episodic exposure and response scenarios (including behavioral responses and TTS) would allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Individuals experiencing TTS will likely fully recover within 24 hours and resume normal behaviors. As described previously in this opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

The 2010 sperm whale recovery plan defines three recovery populations by ocean basin (the Atlantic Ocean/Mediterranean Sea, Pacific Ocean, and Indian Ocean) and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with vessel collisions, reduced prey abundance due to climate change, the possibility that illegal whaling or resumed legal whaling will cause removals at biologically unsustainable rates, contaminants and pollutants, and, possibly, the effects of increasing anthropogenic ocean noise. Of these, anthropogenic noise and ship collision are relevant to Gulf of Alaska activities. As discussed previously, anthropogenic noise associated

with Gulf of Alaska activities will not impact the fitness of any individuals of this species and we do not anticipate vessel collisions. Downlisting criteria for sperm whales includes the maintenance of 1,500 mature, reproductive individuals with at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a 1 percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sperm whale recovery populations.

Based on the evidence available, including the environmental baseline and cumulative effects, we conclude that impulsive and non-impulsive stressors resulting from training activities the Navy plans to conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA regulations from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the status of the species or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of sperm whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of sperm whales in the wild by reducing the reproduction, numbers, or distribution of that species. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual sperm whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

7.7 Steller Sea Lion – Western DPS

Research and observations show that pinnipeds in the water are tolerant of anthropogenic noise and activity. If sea lions are exposed to sonar or other active acoustic sources they may react in a number of ways depending on their experience with the sound source and what activity they are engaged in at the time of the acoustic exposure. Sea lions may not react at all until the sound source is approaching within a few hundred meters and then may alert, approach, ignore the stimulus, change their behaviors, or avoid the immediate area by swimming away or diving (Finneran et al. 2003; Götz and Janik 2011; Kvadsheim et al. 2010). Significant behavioral reactions would not be expected in most cases, and long-term consequences for individuals or the population are unlikely.

Acoustic modeling predicts that the western DPS of Steller sea lion (western U.S. stock) could be exposed to sound from sonar and other active acoustic sources that may result in 286 behavioral reactions. We do not anticipate any instances of TTS, PTS, or other injury or mortality. Ranges to some behavioral impacts could take place at distances exceeding 100 km (62 mi.), although significant behavioral effects are much more likely at higher received levels within a few kilometers of the sound source. Behavioral reactions would be short term, likely lasting the duration of the exposure, and long-term consequences for individuals or populations are unlikely.

Steller sea lions from the western DPS could be exposed to sound from impulsive sources (i.e., explosives) during training activities in the action area. While the NAEMO provides predictions that are conservative, the NAEMO predicts zero instances of Western DPS Steller sea lions being exposed to impulsive sources associated with training activities that rise to the level of take.

The 2008 Steller sea lion recovery plan for the western DPS set criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include eliminating or controlling the threats to the Western DPS including changes in the environment, predation, direct takes by humans, and competition for prey use with fisheries. None of these are relevant to Gulf of Alaska activities. Additionally, downlisting criteria for western DPS Steller sea lions will be met when the population for the U.S. region has increased for 15 years (statistically significant increase) based on counts of juvenile and adult animals. To qualify for delisting, the population for the U.S. region of this DPS must have increased for 30 years (at an average annual growth rate of 3%) based on counts of juvenile and adult animals. Although data vary for the major rookeries, as a whole, the western DPS in Alaska has increased in size by an average of 1.45% y^{-1} of pups and 1.67% y^{-1} of non-pups (95 percent credible interval) from 2000 to 2012, and has been increasing annually since 2002 (Allen and Angliss 2014).

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the Gulf of Alaska TMAA on an annual basis, cumulatively over the five year period of the MMPA rule from April 2017 through April 2022, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the western DPS of Steller sea lions in the wild by reducing the reproduction, numbers, or distribution of that species as currently listed, or currently proposed for listing, under the ESA. We also conclude that effects from ongoing Navy training activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Western DPS Steller sea lions in the wild by reducing the reproduction, numbers, or distribution of that species as currently listed, or currently proposed for listing, under the ESA. Gulf of Alaska stressors will not affect the population dynamics, behavioral ecology, and social

dynamics of individual Steller sea lions in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual sea lions would not be likely to reduce the viability of the populations those individual sea lions represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA, or as currently proposed pursuant to the ESA, that would be sufficient to be readily perceived or estimated.

7.8 Threatened and Endangered Fishes

As described in Section 6, the only activity we determined would likely adversely affect ESA-listed fish species was the use of explosive ordnances, resulting in injury and/or death of some individuals. The highest percentage of any ESU or DPS that is expected to be injured or killed from these activities is 0.006 percent. Most ESA-listed fish species ESUs and DPSs would be affected at much lower levels. This level of mortality and injury represents a very small reduction in abundance that is not likely to appreciably reduce the likelihood of survival and recovery of any ESA-listed Pacific salmon or steelhead. It is also worth noting that, as described in Section 6.7.2, the methodology used to quantify injury and mortality was conservative. Additionally, we conclude that the diversity of ESA-listed fish populations will not be affected by this limited amount of take because it is expected to be distributed across populations through species' ranges in the ocean. As a result, the activities the Navy plans to conduct annually in the GOA TMAA would not appreciably reduce the likelihood of ESA-listed Pacific salmon or steelhead, surviving and recovering in the wild.

Some individual ESA-listed fish may experience TTS as a result of Navy impulsive acoustic stressors. However, the fish species considered in this opinion lack notable hearing specialization, minimizing the likelihood of each instance of TTS affecting an individual's fitness. These species are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014b). Additionally, hearing is not thought to play a role in salmon or steelhead migration (e.g., (Putnam et al. 2013)). Temporary threshold shift is also short term in duration with fish being able to replace hair cells when they are damaged (Lombarte et al. 1993; Smith et al. 2006a). Because these species are able to rely on alternative mechanisms for these essential life functions, instances of TTS would not create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering and would not rise to the level of take.

Additionally, behavioral effects resulting from reactions to sound created by the explosions will be temporary (e.g., a startle response), and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to pre-detonation behavior immediately following each

explosion. Similar to instances of TTS, we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering and would not rise to the level of take.

We also assessed the potential for non-impulsive sources to induce TTS in fish, but determined this was unlikely because fish would have to be within one meter of the sonar source in order for this to occur and at this close of range, fish would likely perceive the sonar source as a predator and move. In the unlikely scenario that a fish was to remain within one meter of the source, we determined that instances of TTS would not create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering and would not rise to the level of take. Further, we assessed the potential for exposure to non-impulsive acoustic stressors to result in behavioral responses, but determined that should an ESA-listed fish elicit a behavioral reaction from exposure to low-frequency sonar (ESA-listed fish considered in this opinion cannot hear mid or high-frequency sonar), we expect these reactions to be temporary, with the individual returning to pre-exposure behavioral patterns immediately following each exposure. Similar to instances of TTS, we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering and such reactions would not rise to the level of take.

As documented in the *Status of Listed Resources* section of this opinion, a number of factors have been identified as limiting ESA-listed salmonid recovery. Most notably those include fisheries harvest, competition from fish raised in hatcheries, hydropower systems, and freshwater and estuarine habitat degradation. The primary stressors associated with Navy training activities in the Gulf of Alaska TMAA, explosive ordnance use and sonar, do not contribute to these factors and have not been documented as being among the factors limiting ESA-listed salmonid recovery.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training activities the Navy will conduct in the GOA TMAA on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Pacific salmon or steelhead in the wild by reducing the reproduction, numbers, or distribution of those ESUs, or DPSs. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of these species as listed pursuant to the ESA.

8 CONCLUSION

During the consultation, we reviewed the current status of blue whales, North Pacific right whales, fin whales, humpback whales (Western North Pacific and Mexico DPSs), sei whales, sperm whales, Steller sea lion - western DPS, coho salmon (Lower Columbia River and Oregon Coast ESUs), chum salmon (Hood Canal summer-run and Columbia River ESUs), and steelhead (Upper Columbia River, Snake River Basin, Lower Columbia River, Upper Willamette River, Middle Columbia River, and Puget Sound DPSs). Additionally, we assessed the *Environmental Baseline* for the action area including ongoing Navy training in the Gulf of Alaska TMAA along with the potential effects of Navy proposed Gulf of Alaska from April 2017 through April 2022 (and into the foreseeable future) along with the National Marine Fisheries Service's Permit Division's proposed rule on the take of marine mammals incidental to training activities and proposed letter of authorization for the five-year period.

We conclude that Navy training activities in the Gulf of Alaska TMAA and NMFS' issuance of the MMPA regulations and LOA are likely to adversely affect but will not appreciably reduce the ability of these threatened and endangered species under NMFS' jurisdiction to survive and recover in the wild by reducing the reproduction, numbers, or distribution of these species. Therefore, we conclude that these activities are not likely to jeopardize the continued existence of any endangered or threatened species. The actions also will not result in the destruction or adverse modification of critical habitat during the five-year period of the MMPA rule or continuing into the reasonably foreseeable future. These conclusions will remain valid assuming that the type, amount and extent of training do not exceed levels assessed in this opinion and/or the status of the species affected by these actions does not change significantly from that assessed in this opinion.

9 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species without special exemption. ESA § 9 statutory prohibitions are limited to "endangered" species unless extended to "threatened" species. In the case of threatened species, section 4(d) of the ESA leaves it to the Secretary's discretion whether and to what extent to extend the statutory 9(a) ''take'' prohibitions, and directs the agency to issue regulations it considers necessary and advisable for the conservation of the species. Take prohibitions have been extended to natural-origin salmonids from threatened ESUs/DPSs and hatchery produced salmonids with intact adipose fins from threatened ESUs/DPSs (70 FR 37160; 71 FR 5178; 73 FR 55451). Take prohibitions have also been extended to Mexico DPS humpback whales which are listed as threatened under the ESA (81 FR 62259). Take prohibitions have not been extended to hatchery produced salmonids with clipped adipose fins from threatened ESUs/DPSs. However, consistent with *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we assessed the amount or extent of take to these threatened species that is anticipated incidental to Navy training activities and include this information in the ITS.

Inclusion of these species in the incidental take statement serves to assist the action agency with monitoring of take and provides a trigger for reinitiation if levels of estimated take are exceeded.

The ESA defines "take" as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct." 16 U.S.C. § 1532(19). Harm is further defined by regulation to include "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering." 50 C.F.R. 222.102. NMFS has not yet defined "harass" under the ESA in regulation. On December 21, 2016, NMFS issued interim guidance on the term "harass," defining it as an action that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering." Under the MMPA, Level B harassment for military readiness activities, such as the activities analyzed in this opinion, is defined as "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered." 16 U.S.C. § 1362(18)(B)(ii). For purposes of this consultation, we relied on NMFS' interim definition of harassment to evaluate whether the proposed activities are likely to harass ESA-listed fish species. For marine mammals, we relied on the MMPA definition of Level B harassment in the context of military readiness activities to estimate the number of instances of harassment. For further explanation, see section 6 of the opinion.

Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

9.1 Amount or Extent of Take

The section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 CFR §402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by proposed actions. If we cannot assign numerical limits for animals that could be incidentally taken during the course of an action, we use the extent of take or "the extent of land or marine area that may be affected by an action" (51 FR 19953). The amount of take resulting from the Navy's activities was estimated based on the best information available.

Section 7(b)(4)(C) of the ESA provides that if an endangered or threatened marine mammal is involved, the taking must first be authorized by Section 101(a)(5) of the MMPA. Accordingly, the terms of this incidental take statement and the exemption from Section 9 of the ESA become effective only upon the issuance of MMPA authorization (i.e., five year regulations and LOA) to

take the marine mammals identified here. Absent such authorization, this statement is inoperative for marine mammals.

The effects analysis contained in this opinion concluded that individual blue whales, North Pacific right whales, fin whales, humpback whales (Mexico and Western North Pacific DPSs), sei whales, sperm whales, Steller sea lions (Western DPS), and listed fish species are likely to be exposed to active sonar, sound fields associated with underwater detonations, or noise and other environmental cues associated with the movement of surface vessels. In some instances, for marine mammals, we concluded that this exposure is likely to result in evasive behavior or changes in behavioral state which we would consider "harassment" for the purposes of this Incidental Take Statement.

The instances of harassment for marine mammals identified in Table 43 would generally represent changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures and, therefore, would represent disruptions of the normal behavioral patterns of the marine mammals that have been exposed. As discussed throughout this opinion, these disruptions are not expected to result in fitness consequences to the animals exposed. However, because the possibility remains that some takes in the form of harassment could occur as a result of exposures, we specify a level of take by harassment that is exempted from the take prohibitions. No marine mammals are likely to die or be wounded or injured as a result of their exposure to the Navy Gulf of Alaska activities. For the purposes of this biological opinion and Incidental Take Statement, we assume that the Navy Gulf of Alaska training activities are likely to result in incidental "take" shown in Table 43.

Table 43. The number of threatened or endangered marine mammals that are likely to be "taken" as a result of their exposure to Navy Training activities conducted in the Gulf of Alaska TMAA on an Annual Basis (April through October each year).

Species	Annual Estimated Take		
	Harrassment		Harm
	Behavioral	TTS	Injury/ Mortality
North Pacific right whale	2	1	0
Humpback whale – Mexico DPS	5	2	0
Humpback whale – Western North Pacific DPS	1	0	0
Blue whale	38	9	0
Fin whale	941	350	0
Sei whale	5	1	0
Sperm whale	98	0	0
Steller sea lion – Western DPS	286	0	0

In Section 6 of this opinion, we estimated the number of ESA-listed salmonids that would be injured or die from Navy Gulf of Alaska training activities annually and over a longer (five-year) period. Expressing take annually and over a 5-year period allows us to consider the effects of low levels and exempt take that may not be expected to occur in a single year, but would be likely over a longer period of time. Table 44, Table 45, and Table 46 show the number of ESA-listed fish that would be killed or injured by Navy Gulf of Alaska training activities annually and over a representative 5-year period in the action area. "Take" of these species will be exceeded if the number or the Net Explosive Weight of the detonations are greater than we expected in our analyses or if the location of these detonations is different from what we expected in our analysis.

Table 44. The number of threatened or endangered fish (hatchery fish with adipose fin-clip) that are likely to be killed or injured as a result of their exposure to Navy Training Activities conducted in the Gulf of Alaska. Take numbers are presented annually and over a representative 5-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Mortality		Injury		% of ESU/DPS (adipose fin-clip) killed or injured in 5 years
			Annual	5 years	Annual	5 years	
Coho	Juvenile	Lower Columbia River - T	3	13	5	23	<0.001
Steelhead	Juvenile	Upper Columbia River - E	2	9	3	15	<0.001
	Juvenile	Snake River basin - T	9	42	14	70	<0.001
	Juvenile	Lower Columbia River - T	3	14	5	24	<0.001
	Juvenile	Middle Columbia River - T	1	5	2	8	<0.001
	Juvenile	Puget Sound - T	1	3	1	4	<0.001

Table 45. The number of threatened or endangered fish (hatchery fish with intact adipose) that are likely to be killed or injured as a result of their exposure to Navy Training Activities conducted in the Gulf of Alaska TMAA. Take numbers are presented annually and over a representative 5-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Mortality		Injury		% of ESU/DPS (intact adipose) killed or injured in 5 years
			Annual	5 years	Annual	5 years	
Coho	Juvenile	Oregon coast - T	0	0	1	1	<0.001
	Adult	Lower Columbia River - T	1	5	2	9	<0.001
	Juvenile		1	1	1	1	<0.001
Chum	Adult	Hood Canal - T	1	1	1	2	0.006
	Juvenile		1	3	1	4	<0.001
	Juvenile	Columbia River - T	1	3	1	5	<0.001
Steelhead	Adult	Upper Columbia River - E	1	1	1	2	0.003
	Juvenile		1	3	1	4	<0.001
	Adult	Snake River basin - T	1	3	1	5	<0.001
	Juvenile		3	14	5	22	<0.001
	Adult	Lower Columbia River - T	1	1	1	1	<0.001
	Adult	Middle Columbia River - T	0	0	1	1	0.002
	Juvenile		2	6	2	10	<0.001
	Juvenile	Puget Sound - T	1	1	1	2	<0.001

Table 46. The number of threatened or endangered fish (naturally produced) that are likely to be killed or injured as a result of their exposure to Navy Training Activities conducted in the Gulf of Alaska TMAA. Take numbers are presented annually and over a representative 5-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Mortality		Injury		% of ESU/DPS (natural) killed or injured in 5 years
			Annual	5 years	Annual	5 years	
Coho	Adult	Oregon coast - T	1	5	2	8	<0.001
	Juvenile		4	20	7	36	<0.001
	Adult	Lower Columbia River - T	1	1	1	1	0.002
	Juvenile		1	2	1	3	<0.001
Chum	Adult	Hood Canal - T	1	1	1	2	0.001
	Juvenile		2	7	3	11	<0.001
	Adult	Columbia River - T	1	1	1	1	0.001
	Juvenile		2	6	2	11	<0.001
Steelhead	Juvenile	Upper Columbia River - E	1	4	2	7	<0.001
	Adult	Snake River basin - T	1	1	1	2	<0.001
	Juvenile		4	19	7	32	<0.001
	Adult	Lower Columbia River - T	1	1	1	1	0.001
	Juvenile		2	7	2	11	<0.001
	Adult	Upper Willamette River - T	1	1	1	1	0.002
	Juvenile		1	3	1	5	<0.001
	Adult	Middle Columbia River - T	1	1	1	1	<0.001
	Juvenile		2	8	3	13	<0.001
	Adult	Puget Sound - T	1	1	1	1	<0.001
	Juvenile		5	23	8	38	<0.001

Activity Levels as Indicators of Take

As discussed in this opinion, the estimated take of ESA-listed marine mammals from acoustic stressors is based on Navy modeling, which represents the best available means of numerically quantifying take. As the level of modeled sonar or explosive use increases, the level of take is likely to increase as well. For non-lethal take from acoustic sources specified above, feasible monitoring techniques for detecting and calculating actual take at the scale of Gulf of Alaska activities do not exist. We are not aware of any other feasible or available means of determining when estimated take levels may be exceeded. Therefore, we must rely on Navy modeling, and the link between sonar or explosive use and the level of take, to determine when anticipated take levels have been exceeded. As such, we established a term and condition of this Incidental Take Statement that requires the Navy to report to NMFS any exceedance of activity specified in the preceding opinion and in the final MMPA rule before the exceedance occurs if operational security considerations allow, or as soon as operational security considerations allow after the relevant activity is conducted. Exceedance of an activity level will require the Navy to reinitiate consultation.

Detecting injury or mortality of fish during Navy training activities in the GOA TMAA will also be extremely difficult. Monitoring techniques to calculate actual take of fish including detection and collection of individuals and assessment of injuries or death is not feasible in the offshore marine environment where the Navy's training activities occur. We must rely on predicted take associated with levels of activities and any opportunistic observations of potential injured or dead fish during training activities to determine when anticipated take levels have been exceeded. "Take" of these species will be exceeded if the number or the Net Explosive Weight of the detonations are greater than we expected in our analyses or if the location of these detonations is different from what we expected in our analysis.

9.2 Effects of the Take

In the accompanying opinion, we determined that the anticipated take levels specified above are not likely to result in jeopardy to any of the affected species or destruction or adverse modification of critical habitat that has been designated for such species.

9.3 Reasonable and Prudent Measures

The measures described below are nondiscretionary, and must be undertaken by the Navy so that they become binding conditions for the exemption in section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures, and term and conditions to implement the measures, must be provided. Only incidental take

resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

The following reasonable and prudent measures are necessary and appropriate to minimize impacts of incidental take of the species listed in Table 43, Table 44, Table 45, and Table 46 in the incidental take statement of this biological opinion.

1. The Navy and NMFS' Permits Division shall implement mitigation and reporting measures to limit the potential for interactions with ESA-listed species (i.e., marine mammals and fish) that may rise to the level of take as a result of the proposed actions described in this opinion.

2. The Navy and NMFS' Permits Division shall report all observed interactions resulting in take with any ESA-listed species (i.e., marine mammals, fish) resulting from the proposed training activities and any observations of stranded or dead ESA-listed marine mammals that are not attributable to Navy training but are observed during the course of Navy training activities and while implementing monitoring requirements required by this opinion and the MMPA LOA.

3. The U.S. Navy must report to NMFS any exceedance of activity levels or planned training events specified in the preceding opinion and in the final MMPA rule before the exceedance occurs (if operational security considerations allow), or as soon as operational security considerations allow after the relevant activity is conducted.

4. The Navy and NMFS' Permits and Conservation Division shall submit reports that identify the general location, timing, number of sonar hours and other aspects of the training activities, and any potential to exceed levels of training analyzed in this opinion to help assess the actual amount or extent of take incidental to training activities.

9.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Navy and NMFS' Permits and Conservation Division must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above and outlines the mitigation, monitoring and reporting measures required by the section 7 regulations (50 CFR 402.14(i)). These terms and conditions are non-discretionary. If the Navy and NMFS's Permits and Conservation

Division fail to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

1. The Navy shall implement all mitigation and monitoring measures as proposed in the action described in the Final SEIS/OEIS and consultation initiation package, as specified in the final MMPA rule and LOA, and as described in this opinion in Section 2.

2. NMFS' Permits Division shall ensure that all mitigation and monitoring measures as prescribed in the final rule and LOAs, and as described in Section 2 of this opinion are implemented by the U.S. Navy.

3. The U.S. Navy and NMFS' Permits Division shall compile and summarize annual monitoring and exercise reports and describe interactions with ESA-listed species, as specified in the final MMPA rule and LOA.

4. If a dead or injured marine mammal is observed during the training activities, the Navy shall immediately contact NMFS and appropriate stranding networks.

5. The Navy shall monitor and coordinate with marine mammal stranding networks to help determine any potential relationship of any stranding with Navy training activities.
6. The Navy must report to NMFS any exceedance of activity levels (e.g. sonar hours and the type and numbers of explosives used) or planned training events specified in the preceding opinion and in the final MMPA rule before the exceedance occurs if operational security considerations allow, or as soon as operational security considerations allow after the relevant activity is conducted. Exceedance of an activity level will require the Navy to reinitiate consultation.

10 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 CFR 402.02).

1. As practicable, develop procedures to aid any ESA-listed marine mammals that have been impacted by Navy Gulf of Alaska activities and is in a condition requiring assistance to increase likelihood of survival.

2. Continue to model potential impacts to ESA-listed species using NAEMO and other relevant models; validate assumptions used in risk analyses; and seek new information and higher quality data for use in such efforts.

3. Continue to validate assumptions in the post-model analysis regarding animal avoidance behavior and effectiveness of mitigation to avoid or reduce acoustic exposures during Navy training activities.

4. Continue to support the recording of underwater ambient and anthropogenic sounds in the Gulf of Alaska TMAA. In particular, focus efforts to record sound levels during Navy training exercises at locations in the TMAA closest to designated critical habitat for North Pacific right whales and Western DPS Steller sea lions, as well as locations in the TMAA closest to the mouth of Cook Inlet.

5. Continue technical assistance/adaptive management efforts with NMFS to help inform future consultations on Navy training in the Gulf of Alaska TMAA.

6. Coordinate with NMFS to better understand acoustic effects to fish. This includes further refining acoustic criteria developed for this and other consultations. This can also include increased coordination with NMFS Fisheries Science Centers for the use of NMFS collected life history information and at-sea distribution, abundance, and density of ESA-listed fish.

11 REINITIATION OF CONSULTATION

This concludes formal consultation on proposed Gulf of Alaska training activities the Navy will conduct and NMFS's promulgation of regulations and issuance of incidental take authorizations pursuant to the MMPA from April 2017 through April 2022. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the Navy and NMFS' Permits Division must contact the ESA Interagency Cooperation Division, Office of Protected Resources immediately.

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