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

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 Northwest Training and Testing (NWTT) activities

(2) National Marine Fisheries Services' promulgation of regulations pursuant to the Marine Mammal Protection Act for the Navy to "take" marine mammals incidental to NWTT activities from November 2015 through November 2020

(3) National Marine Fisheries Services' issuance of two Letters of Authorization to the Navy pursuant to regulations under the Marine Mammal Protection Act to "take" marine mammals incidental to NWTT activities from November 2015 through November 2020

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service

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1 INTRODUCTION

Section 7 (a)(2) of the ESA requires Federal agencies to insure 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 exercises and testing activities, and NMFS Office of Protected Resources - Permits and Conservation Division (Permits Division), which 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 activities in the Northwest Training and Testing (NWTT) Study Area that may affect several ESA-listed species. The regulations propose to authorize the issuance of two Letters of Authorization (LOAs) that will allow the Navy to "take" marine mammals incidental to its proposed action. The Federal action of issuing LOAs 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

This Opinion is based on information provided during pre-consultation and in the U.S. Navy's (Navy) January 9, 2015 request for ESA consultation package, including a draft EIS/OEIS, a supplement to the draft EIS/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 (80 FR 31737) and draft letters 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 including Northwest Training Range Complex (NWTRC) and Keyport biological opinions.

1.2 Consultation History

On January 24, 2014, the Navy provided NMFS with the Draft EIS/OEIS for NWTT.

On October 8, 2014, the Navy provided NMFS an updated timeline for the EIS/OEIS, NMFS' issuance of an incidental take authorization pursuant to the MMPA, and the ESA section 7 consultation schedule for NMFS review and concurrence.

On November 11, 2014, the Navy provided NMFS with the Navy's analysis of proposed biologically important areas (pursuant to the MMPA incidental take authorization) overlapping with the NWTT Study Area.

On December 12, 2014, the Navy withdrew the request for mortality in the MMPA incidental take authorization.

On January 9, 2015, NMFS received the Navy's biological evaluation and a request for formal consultation for proposed Navy NWTT activities.

On March 18, 2015, NMFS met with the Navy to review comments received on the Draft NWTT EIS/OEIS pursuant to the Cooperating Agency agreement as relevant to the pending MMPA incidental take authorization and ESA section 7 consultation.

On April 8, 2015, following initial review of the Navy's request for formal consultation, NMFS determined that there was sufficient information to initiate formal consultation.

Also on April 8, 2015, NMFS requested that the Navy agree to extend the consultation timeline beyond the statutory timeline (135 days) and to complete a biological opinion on or before September 21, 2015 (approximately 160 days).

On April 10, 2015, NMFS received the Navy's concurrence to extend the consultation beyond 135 days and to complete the biological opinion on or before September 21, 2015.

On May 28, 2015, NMFS received information from the USFWS on the NWTT description of the action and exposure profiles to ensure consistency between the Services' respective ESA section 7 consultations on the Navy's NWTT activities.

On July 7, 2015, NMFS' ESA Interagency Cooperation Division initiated intra-agency coordination on the NWTT biological opinion with NMFS' West Coast Regional Office to ensure the use of the use the best scientific and commercial data available.

On July 22, 2015, the Navy provided NMFS and the USFWS its proposed acoustic criteria for effects to fish species.

On July 30, 2015 NMFS and the USFWS met to discuss the Navy's proposed acoustic criteria for fish species. On July 31, 2015, NMFS provided the Navy comments on the proposed criteria. Receipt of this new information necessitated additional time to complete the biological opinion beyond the formerly agreed upon date of September 21, 2015. NMFS and the Navy agreed to extend the consultation to October 26 2015.

On September 15, 2015 the Navy provided NMFS final range to effects values for fish with swim bladders from underwater detonations.

On September 18, 2015 NMFS provided the Navy the draft biological opinion for NWTT activities.

On October 5, 2015, NMFS received comments from the Navy on the draft biological opinion.

Between October 5 and November 2, 2015, the NMFS and Navy communicated via email and telephone to resolve comments and remaining issues on the draft biological opinion.

On November 2, 2015, the Navy provided NMFS with supplemental information on the range to effects for fish without swim bladders from underwater detonations.

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 and testing activities conducted in the NWTT Study Area; (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 November 2015 through November 2020; and (3) NMFS Permits Division's proposed issuance of LOAs pursuant to the proposed regulations that will authorize the Navy to "take" marine mammals incidental to military readiness activities in the NWTT Study Area through November 2020. This Opinion supersedes the reinitiated biological opinion for the Northwest Training Range Complex (NWTRC) issued on August 1, 2014, and all previous biological opinions on the Keyport Range Complex and Southeast Alaska Acoustic Measurement Facility (SEAFAC).

The purpose of the activities the Navy conducts in the NWTT Study Area is to meet the requirements of the Navy's Fleet Response Training Plan and allow Navy personnel to remain proficient in anti-submarine warfare and mine warfare skills (i.e., military readiness activities). The purpose of the MMPA regulations and the Permits and Conservation Division's LOAs is to allow the Navy to "take" marine mammals incidental to military readiness activities in the NWTT Study Area conducted through November 2020 in a manner that is consistent with the requirements of the MMPA and implementing regulations.

NMFS recognizes that while Navy training and testing 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 and testing activities proposed by the Navy during the period of NMFS' proposed incidental take authorization pursuant to the MMPA (November 2015 through 2020) would continue into the reasonably foreseeable future at levels similar to that assessed in this Opinion and described in the NWTT DEIS/OEIS.

The tempo of training within the NWTT Study Area is subject to variation within the scope of the activities described in the Navy's NWTT Draft 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 NWTT; 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.

The Navy categorizes training and testing activities into functional warfare areas called primary mission areas. Most training and testing activities analyzed in the NWTT EIS/OEIS fall into the following six primary mission areas:

- Anti-Air Warfare
- Anti-Surface Warfare
- Anti-Submarine Warfare (ASW)
- Electronic Warfare
- Mine Warfare (MIW)
- Naval Special Warfare

The research and acquisition community (i.e., testing community) also conducts testing activities which do not readily fit into the primary mission areas. These additional testing activity categories are listed in this document as follows:

- Naval Undersea Warfare Center Division, Keyport (NUWC Keyport)Testing activities
- Naval Surface Warfare Center, Carderock Division (NSWCCD) Detachment Puget Sound activities
- NSWCCD, Southeast Alaska Acoustic Measurement Facility activities
- Life Cycle Activities
- Shipboard Protection Systems and Swimmer Defense Testing
- Unmanned Vehicle Testing
- New Ship Construction

Additionally, some miscellaneous activities are grouped under "Other Activities."

2.1 Navy Proposed Training Activities

The Navy's proposed training activities are briefly described in Table 1. The table is organized according to primary mission areas and includes the activity name and a short description. Appendix A (Navy Activities Descriptions) in the NWTT EIS/OEIS has more detailed descriptions of the training activities.

Table 1. Representative Training Activities Occurring in the NWTT Study 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.
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	Aircrews defend against threat aircraft with missiles.
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A])	Surface ship crews defend against threat aircraft or missiles with guns.
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	Surface ship crews defend against threat missiles and aircraft with missiles.
Anti-Surface Warfare (ASUW)	
Gunnery Exercise (Surface-to-Surface) – Ship (GUNEX [S-S] – Ship)	Ship 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.
Gunnery Exercise (Surface-to-Surface) – Boat (GUNEX [S-S] – Boat)	Small boat crews engage surface targets with small- and medium-caliber weapons. Only blank rounds are fired.
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	Fixed-wing aircrews simulate firing precision-guided missiles using captive air training missiles against surface targets. Some activities include firing a missile with a high-explosive warhead.
High-Speed Anti-Radiation Missile (HARM) Exercise (Non-firing)	Fixed-wing aircrews simulate firing HARM missiles, using captive air training missiles against surface targets. All missile firings are simulated; no actual missiles are fired.
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Fixed-wing aircrews deliver bombs against surface targets.
Anti-Submarine Warfare (ASW)	
Tracking Exercise – Submarine (TRACKEX – Sub)	Submarine crews search for, detect, and track submarines and surface ships.
Tracking Exercise – Surface (TRACKEX – Surface)	Surface ship crews search for, detect, and track submarines.
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 (Extended Echo Ranging Sonobuoys)	Maritime patrol aircraft crews search for, detect, and track submarines using a multistatic active coherent system.
Electronic Warfare (EW)	

Activity Name	Activity Description
Electronic Warfare Operations (EW OPS)	Aircraft, surface ship, and submarine crews attempt to control portions of the electromagnetic spectrum used by enemy systems to degrade or deny the enemy's ability to take defensive or offensive actions.
Mine Warfare (MIW)	
Mine Neutralization – Explosive Ordnance Disposal (EOD)	Personnel disable threat mines. Explosive charges may be used.
Submarine Mine Exercise	Submarine crews practice detecting non-explosive training mine shapes in a designated area.
Civilian Port Defense	Civilian Port Defense exercises are naval mine warfare activities conducted at various ports and harbors in support of maritime homeland defense/security.
Naval Special Warfare (NSW)	
Personnel Insertion/Extraction – Submersible	Military personnel train for covert insertion and extraction into target areas using submersibles.
Personnel Insertion/Extraction – Non-Submersible	Military personnel train for covert insertion and extraction into target areas using rotary wing aircraft, fixed-wing aircraft (insertion only), or small boats.
Other Training Activities	
Maritime Security Operations	Surface ship crews conduct a suite of Maritime Security Operations (MSO) events, including maritime security escorts for Navy vessels such as Fleet Ballistic Missile Submarines (SSBNs); Visit, Board, Search, and Seizure; Maritime Interdiction Operations; Force Protection; and Anti-Piracy Operations.
Precision Anchoring	Anchors are released in designated locations.
Small Boat Attack	Small boat crews engage pierside surface targets with small-caliber weapons. Only blank rounds are fired.
Intelligence, Surveillance, and Reconnaissance (ISR)	Aircraft crews and unmanned aircraft systems conduct searches and gather intelligence using visual, optical, acoustic, and electronic systems.
Search and Rescue	Helicopter crews conduct helicopter insertion and extraction.
Surface Ship Sonar Maintenance	Maintenance of sonar systems occurs while the ships are moored and at sea.
Submarine Sonar Maintenance	Maintenance of sonar systems occurs while the submarines are moored and at sea.

2.1.1 Training Activity Levels

Table 2 provides a summary of training activities (as described in the previous Section) including tempo and quantities of inert and live munitions that the Navy plans to expend during training that were analyzed by the Navy. Munitions that contain high explosives (highlighted in gray) have greater potential for impact to listed species.

Table 2. Proposed Training Activity Levels

Range Activity	Location	No. of events (per year)	Ordnance (Number per year)
Anti-Air Warfare			
Air Combat Maneuver (ACM)	Offshore Area (W-237, Olympic MOAs)	550	None
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	Offshore Area (W-237)	24	30 (AIM-7/9/120) 15 HE warheads
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A])	Offshore Area (W-237)	160	310 large-caliber rounds (230 HE) 16,000 medium-caliber rounds (6,320 HE)
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	Offshore Area (W-237)	4	8 HE warheads
Anti-Surface Warfare (ASUW)			
Gunnery Exercise (Surface-to-Surface) – Ship (GUNEX [S-S] – Ship)	Offshore Area	200	121,200 small-caliber rounds 33,540 medium-caliber rounds (48 HE) 2,880 large-caliber rounds (80 HE)
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	Offshore Area (W-237)	4	4 HE Missiles
High-Speed Anti-Radiation Missile (HARM) Exercise (Non-firing)	Offshore Area (W-237)	1,740	All non-firing Captive Air Training Missiles
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Offshore Area (W-237)	30	10 HE Bombs 110 NEPM Bombs
Anti-Submarine Warfare (ASW)			
Tracking Exercise – Submarine (TRACKEX – Sub)	Offshore Area	100	None
Tracking Exercise – Surface (TRACKEX – Surface)	Offshore Area	65	None
Tracking Exercise – Helicopter (TRACKEX – Helo)	Offshore Area	4	None
Tracking Exercise – Maritime Patrol Aircraft	Offshore Area	300	None

Range Activity	**Location**	**No. of events (per year)**	**Ordnance (Number per year)**
(TRACKEX – MPA)			
Tracking Exercise – Maritime Patrol Aircraft MAC (TRACKEX MPA MAC)	Offshore Area	24	720 SSQ-125 sonobuoys
Electronic Warfare (EW)			
Electronic Warfare Operations (EW OPS)	Offshore Area	5,000 (aircraft) 275 (ship)	None
Mine Warfare (MIW)			
Mine Neutralization – Explosive Ordnance Disposal (EOD)	Inland Waters (Crescent Harbor EOD Training Range)	3	Three 2.5 lb. HE charges
		3	18 SWAG
	Inland Waters (Hood Canal EOD Training Range)	3	Three 2.5 lb. HE charges
		3	18 SWAG
Submarine Mine Exercise	Offshore Area	8	None
Civilian Port Defense	Inland Waters	Every other year (three in 5 years)	None
Naval Special Warfare (NSW)			
Personnel Insertion/Extraction – Submersible	Inland Waters	35	None
Personnel Insertion/Extraction – Non-Submersible	Inland Waters (Crescent Harbor, R6701)	10	None
Other			
Maritime Security Operations	Inland Waters (NAVBASE Kitsap Bangor, Hood Canal, Dabob Bay, Puget Sound, Strait of Juan de Fuca)	286	Small- and medium-caliber blanks only
Other			
Precision Anchoring	Inland Waters (Naval Station Everett, Indian Island)	10	None
Small Boat Attack	Inland Waters (Naval Station Everett NAVBASE Kitsap Bangor NAVBASE Kitsap	1	3,000 small-caliber rounds (all blanks)

Range Activity	Location	No. of events (per year)	Ordnance (Number per year)
	Bremerton)		
Intelligence, Surveillance, Reconnaissance (ISR)	Offshore Area	200	None
Search and Rescue	Inland Waters (Crescent Harbor, Navy 7)	100	None
Surface Ship Sonar Maintenance	Inland Waters (NAVBASE Kitsap Bremerton, Naval Station Everett) and Offshore Area	13	None
Submarine Sonar Maintenance	Inland Waters (NAVBASE Kitsap Bangor) and Offshore Area	22	None

Notes: EOD = Explosive Ordnance Disposal, HE = High Explosive, lb. = pound(s), NEPM = Non-explosive Practice Munition, SWAG= Shock Wave Action Generator, MOA = Military Operations Area, MAC = Multistatic Active Coherent, NAVBASE = Naval Base

2.2 Navy Testing Activities

The Navy's research and acquisition community engages in a broad spectrum of testing activities in support of the fleet. These activities include, but are not limited to, basic and applied scientific research and technology development; testing, evaluation, and maintenance of systems (missiles, radar, and sonar), and platforms (surface ships, submarines, and aircraft); and acquisition of systems and platforms to support Navy missions and give a technological edge over adversaries.

The individual commands within the research and acquisition community are:

- Naval Sea Systems Command (NAVSEA). Within NAVSEA are the following field activities:
    - Naval Undersea Warfare Center (NUWC) Division, Keyport
    - Naval Surface Warfare Center, Carderock Division (NSWCCD), Detachment Puget Sound
    - NSWCCD Southeast Alaska Acoustic Measurement Facility (SEAFAC)
    - Puget Sound Naval Shipyard and Intermediate Maintenance Facility
    - Various NAVSEA program office-sponsored testing activities
- Naval Air Systems Command (NAVAIR)

The Navy operates in an ever-changing strategic, tactical, funding, and time-constrained environment. Testing activities occur in response to emerging science or fleet operational needs. Following identification of future needs, new systems are developed or existing systems are modified. These systems—whether new or modifications of existing systems—must be tested in the field to ensure they meet fleet needs and requirements.

Some testing activities are similar to training activities conducted by the fleet. For example, both the fleet and the research and acquisition community fire "test" torpedoes. While the firing of a torpedo might look identical to an observer, the difference is in the purpose of the firing. The fleet might fire the torpedo to practice the procedures for such a firing, whereas the research and acquisition community might be assessing a new torpedo guidance technology or ensuring that the torpedo meets performance specifications and operational requirements.

2.2.1 Naval Sea Systems Command Testing Activities

NAVSEA is responsible for engineering, building, buying, and maintaining the Navy's ships and submarines and associated combat systems. NAVSEA provides technical experts at field activities operated by the undersea and surface warfare centers (NUWC and NSWC) to support various aspects of RDT&E, and at shipyards to support lifecycle maintenance. NAVSEA has several field activities operating out of NAVBASE Kitsap, including NUWC Division, Keyport, NSWC Carderock Division, Detachment Puget Sound, and Puget Sound Naval Shipyard and Intermediate Maintenance Facility.

Each major category of NAVSEA activities in the Study Area is described below. NUWC Division Keyport and NSWC Carderock Division Detachment Puget Sound activities are grouped together in the discussion below to simplify review due to the diversity of activity types and locations they work in. Puget Sound Naval Shipyard and Intermediate Facility activities are grouped with the general activities conducted by NAVSEA. Numerous test activities and technical evaluations, in support of NAVSEA's systems development mission, often occur in conjunction with fleet activities within the NWTT Study Area.

2.2.1.1 *Naval Undersea Warfare Center Division, Keyport Testing Activities*

NUWC Division, Keyport's mission is to provide advanced technical capabilities for test and evaluation, in-service engineering, maintenance and industrial base support, fleet material readiness, and obsolescence management for undersea warfare. Naval Undersea Warfare Center Division, Keyport has historically provided facilities and capabilities to support testing of torpedoes, other unmanned vehicles, submarine readiness, diver training, and similar activities that are critical to the success of undersea warfare. Range support requirements for such activities include testing, training, and evaluation of system capabilities such as guidance, control, and sensor accuracy in multiple marine environments (e.g., differing depths, salinity levels, sea states) and in surrogate and simulated war-fighting environments. Technological advancements in the materials, instrumentation, guidance systems, and tactical capabilities of manned and unmanned vehicles continue to evolve in parallel with emerging national security priorities and threat assessments. However, NUWC Division, Keyport does not utilize explosives in any testing scenarios.

NUWC, Division Keyport operates the Keyport Range Complex, which includes the Keyport Range Site and the Dabob Bay Range Complex in the Inland Waters portion of the Study Area, and the Quinault Range Site including a small surf zone at Pacific Beach in the Offshore Area portion of the Study Area.

2.2.1.2 ***Naval Surface Warfare Center, Carderock Division, Detachment Puget Sound Testing Activities***

The NSWCCD Detachment Puget Sound provides research, development, test and evaluation (RDT&E), analysis, acquisition support, in-service engineering, logistics and integration of surface and undersea vehicles and associated systems; develops and applies science and technology associated with naval architecture and marine engineering; and provides support to the maritime industry (e.g., NAVSEA, Research Laboratories, and other commercial, academic, and private research entities). Activities and support include engineering, technical, operations, diving, and logistics required for the RDT&E associated with:

- Advanced Technology Concepts, Engineering, and Proofing
- Experimental Underwater Vehicles, Systems, Subsystems, and Components
- Specialized Underwater Systems, Equipment, Tools, and Hardware
- Acoustic Data Acquisition, Analysis, and Measurement Systems (required to measure Navy Acoustic Signatures)

These activities can be categorized as two major types: System, Subsystem, and Component Acoustic Testing; and Proof-of-Concept Testing. System, Subsystem, and Component Acoustic Testing would occur in inland waters and at-sea environments to obtain static and short-distance operational performance and acoustic measurements. Development testing and training would also be exercised under this test category to validate equipment development and to provide operator training. Typical activity descriptions for each major category are provided below.

NSWCCD, Detachment Puget Sound operates the testing pier at NAVBASE Kitsap Bangor, conducts testing in the Inland Waters portion of the Study Area (Hood Canal, Dabob Bay and Carr Inlet, primarily), and operates the Southeast Alaska Measurement Facility (SEAFAC) in Ketchikan, Alaska.

2.2.1.3 ***Naval Surface Warfare Center, Carderock Division, Southeast Alaska Acoustic***

Measurement Facility Testing Activities

Naval Surface Warfare Center, Carderock Division, SEAFAC conducts high-fidelity passive acoustic signature measurements of submarines and ships. The SEAFAC site includes hydrophone arrays and data collection and processing systems for real-time data analysis and signature evaluation.

As the Navy's primary acoustic engineering measurement facility in the Pacific, SEAFAC provides the capability to perform RDT&E analyses to determine the sources of radiated acoustic noise, to assess vulnerability, and to develop quieting measures.

2.2.1.4 *Naval Sea Systems Command Program Office Sponsored Testing Activities*

Naval Sea Systems Command also conducts tests that are not associated with NUWC Keyport or NSWCCD. Some of these activities are conducted in conjunction with fleet activities in the Offshore Area off the coast of Washington, Oregon, and northern California, and some occur at Navy piers at NAVBASE Kitsap Bremerton, NAVBASE Kitsap Bangor, and Naval Station Everett. Tests within this category include, but are not limited to, anti-surface warfare, anti-submarine warfare, mine warfare, and force protection (maintaining security of Navy facilities, ships, submarines, and aircraft).

Table 3 provides descriptions of the NAVSEA activities included in the Action Area.

Table 3. Representative Naval Sea Systems Command Testing Activities

Activity Name		Activity Description
Naval Undersea Warfare Center Division, Keyport		
Torpedo Testing	Torpedo Non-Explosive Testing	Test of a non-explosive torpedo against a target.
Autonomous and Non-Autonomous Vehicles	Unmanned Underwater Vehicle Testing	UUVs are autonomous or remotely operated vehicles with a variety of different payloads used for various purposes.
	Unmanned Aircraft System	UASs are remotely piloted or self-piloted (i.e., preprogrammed flight pattern) aircraft that include fixed-wing, rotary-wing, and other vertical takeoff vehicles. They can carry cameras, sensors, communications equipment, or other payloads.
	Unmanned Surface Vehicle Testing	USVs are primarily autonomous systems designed to augment current and future platforms to help deter maritime threats. They employ a variety of sensors designed to extend the reach of manned ships.
Fleet Training Support	Cold Water Training	Fleet training for divers in a cold water environment and other diver training related to Navy divers supporting range operations.
	Post-Refit Sea Trial	Following periodic maintenance or repairs, sea trials are conducted to evaluate submarine propulsion, sonar systems, and other mechanical tests.
	Anti-Submarine Warfare Testing	Ships and their supporting platforms (e.g., helicopters, unmanned aerial vehicles) detect, localize, and prosecute submarines or other training targets.
Maintenance and Miscellaneous	Side Scan/Multibeam	Side Scan/Multibeam systems associated with a vessel or UUV are tested to ensure they can detect, classify, and localize targets in a real world environment.
	Non-Acoustic Tests	These tests involve non-acoustic sensors. Non-acoustic sensors may also gather other forms of environmental data.
Acoustic Component Test	Countermeasures Testing	Includes testing of two types of countermeasures: those that emit active acoustic energy of varying frequencies into the water to mimic the characteristics of a target so that the actual threat or target remains undetected; and those that would detect, localize, track, and attack incoming weapons.
	Acoustic Test Facility	Various acoustic component testing and calibration is conducted in a controlled experimental environment based on periodicity and is also conducted on modified, upgraded, and experimental devices.
	Pierside Integrated Swimmer Defense	Swimmer defense testing ensures that systems can effectively detect, characterize, verify, and engage swimmer and diver threats in harbor environments.
Naval Surface Warfare Center, Carderock Division Detachment Puget Sound		
System, Subsystem and Component Testing	Pierside Acoustic Testing	Operating AUV, ROV, UUV, submersibles/Concepts and Prototypes (including experimental vehicles, systems, equipment, tools and hardware) underwater in a static or dynamic condition within 500 yd. of an instrumented platform moored pierside.

Activity Name		**Activity Description**
	Performance Testing At-Sea	Operating AUV, ROV, UUV, submersibles/Concepts and Prototypes underwater at sea. Systems will be exercised to obtain operational performance measurements of all subsystems and components used for navigation and mission objectives.
	Development Training and Testing	Operating AUV, ROV, UUV, submersibles/Concepts and Prototypes underwater at Sea. Systems will be exercised to validate development and to provide operator familiarization and training with all subsystems and components used for navigation and mission objectives.
Proof of Concept Testing		Design, fabrication and installation of unique hardware and towing configurations in support of various surface and underwater demonstrations as proof-of-concept.
Naval Surface Warfare Center, Carderock Division, Southeast Alaska Acoustic Measurement Facility		
Surface Vessel Acoustic Measurement		Conduct acoustic trial measurements of surface vessels
Underwater Vessel Acoustic Measurement		Conduct acoustic trial measurements of underwater vessels
Underwater Vessel Hydrodynamic Performance Measurement		Conduct hydrodynamic performance trial measurements
Cold-water Training		Involves Navy personnel conducting insertion training in cold-water conditions. The training may include ingress and egress from subsurface vessels and small surface craft.
Component System Testing		Conduct testing on individual components of new defense acquisition systems
Countermeasures Testing		Conduct engineering and acceptance testing of countermeasures
Electromagnetic Measurement		Conduct new construction, post-PSA, and life cycle electromagnetic measurements
Measurement System Repair & Replacement		Conduct repairs, replacements and calibration of acoustic measurement systems
Project Operations (POPS)		Support testing of fleet assets
Target Strength Trial		Asset moored to static site. Acoustic projectors and receive arrays will be rotated around asset. Broadband waveforms will be transmitted. Underwater tracking system would be utilized to monitor relative positions.
Additional Naval Sea Systems Command Testing Activities		
Life Cycle Activities	Pierside Sonar Testing	Pierside testing of submarine and surface ship sonar systems occurs periodically following major maintenance periods and for routine maintenance.
Shipboard Protection Systems and Swimmer Defense Testing	Pierside Integrated Swimmer Defense	Swimmer defense testing ensures that systems can effectively detect, characterize, verify, and engage swimmer and diver threats in harbor environments.

Activity Name		Activity Description
Unmanned Vehicle Testing	Unmanned Vehicle Development and Payload Testing	Vehicle development involves the production and upgrade of new unmanned platforms on which to attach various payloads used for different purposes.
Anti-Surface Warfare (ASUW)/Anti-Submarine Warfare (ASW) Testing	Torpedo Testing	Air, surface, or submarine crews employ explosive torpedoes against artificial targets.
	Torpedo Non-Explosive Testing	Air, surface, or submarine crews employ non-explosive torpedoes against submarines or surface vessels.
	Countermeasure Testing	Countermeasure testing involves the testing of systems that would detect, localize, track, and attack incoming weapons.
New Ship Construction	Anti-Submarine Warfare Mission Package Testing	Ships and their supporting platforms (e.g., helicopters, unmanned aerial vehicles) detect, localize, and prosecute submarines.

Notes: AUV = Autonomous Underwater Vehicle, ROV = Remotely Operated Vehicle, UAS = Unmanned Aircraft System, USV = Unmanned Surface Vehicle, UUV = Unmanned Underwater Vehicle, PSA = Post Shakedown Availability (also known as post-delivery maintenance work)

2.2.2 Naval Air Systems Command Testing Activities

Naval Air Systems Command testing events generally fall into the primary mission areas used by the fleets. Naval Air Systems Command events include, but are not limited to, the testing of new aircraft platforms, weapons, and systems Many platforms (e.g., the P-8A) and systems (e.g., sonobuoys) currently being tested by NAVAIR will ultimately be integrated into fleet training activities. As all NAVAIR testing activities in the NWTT Study Area are similar to training events, it is difficult to discern between the two types of activities. The purpose of NAVAIR testing is to evaluate system performance in real-life scenarios rather than to train in system proficiency.

A comparison of NAVAIR's testing activities (Table 4) and the fleet's training activities (Table 1) highlights the commonalities between the two.

Table 4. Representative Naval Air Systems Command Testing Activities

Activity Name	Activity Description
Anti-Submarine Warfare (ASW)	
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (ASW TRACKEX – MPA) (Directional Command Activated Sonobuoy System [DICASS])	All NAVAIR ASW testing activities are similar to the training event ASW TRACKEX – MPA. This test evaluates the sensors and systems used by maritime patrol aircraft to detect and track submarines using the DICASS.
ASW Tracking Test – Maritime Patrol Aircraft (Mutistatic Active Coherent [MAC])	This test evaluates the sensors and systems used by maritime patrol aircraft to detect and track submarines using the MAC sonobuoy system.
ASW Tracking Test – Maritime Patrol Aircraft (Sound Underwater Signal [SUS])	This test evaluates the sensors and systems used by maritime patrol aircraft to communicate with submarines using any of the family of SUS systems.
ASW Tracking Test – Maritime Patrol Aircraft (Improved Extended Echo Ranging [IEER])	This test evaluates the sensors and systems used by maritime patrol aircraft to detect and track submarines using the IEER sonobuoy system.
ASW Tracking Test – Maritime Patrol Aircraft (High Duty Cycle [HDC])	This test evaluates the sensors and systems used by maritime patrol aircraft to detect and track submarines using the HDC sonobuoy system.
Electronic Warfare (EW)	
Flare Test	Flare tests evaluate newly developed or enhanced flares, flare dispensing equipment, or modified aircraft systems against flare deployment. Tests may also train pilots and aircrew in the use of newly developed or modified flare deployment systems. Flare tests are often conducted with other test events, and are not typically conducted as standalone tests.

Note: NAVAIR = Naval Air Systems Command

2.2.3 Testing Activity Levels

The following tables (Table 5 and Table 6) summarize the testing activities that comprise the Action. The tables identify the general locations within the NWTT Study Area where the events are typically conducted. Cells under the "Ordnance" column are shaded gray if that activity includes the use of explosives.

Table 5. Naval Sea Systems Command Testing Activities in the NWTT Study Area

Range Activity		**Location**	**Action**	
			No. of events (per year)	**Ordnance (Number per year)**
Naval Undersea Warfare Center Division, Keyport Testing Activities				
Torpedo Testing	Torpedo Non-Explosive Testing	Offshore Area (QRS)	20	101 NEPM torpedoes
		Inland Waters (DBRC Site)	41	189 NEPM torpedoes
Autonomous and Non-Autonomous Vehicles	Unmanned Underwater Vehicle Testing	Inland Waters (DBRC Site, Keyport Range Site)	131	134 NEPM torpedoes
		Offshore Area (QRS)	20	None
	Unmanned Aircraft System	Offshore Area (QRS)	20	None
		Inland Waters (DBRC Site)	20	None
	Unmanned Surface Vehicle	Offshore Area (QRS)	20	None
		Inland Waters (DBRC Site, Keyport Range Site)	20	None
Fleet Training/ Support	Cold Water Training	Offshore Area (QRS)	20	None
		Inland Waters (DBRC Site, Keyport Range Site)	65	None
	Post-Refit Sea Trial	Inland Waters (DBRC Site)	32	None
	Anti-Submarine Warfare (ASW) Testing	Offshore Area (QRS)	5	None
Maintenance and Miscellaneous	Side Scan/Multibeam Sonar	Inland Waters (DBRC Site, Keyport Range Site)	54	None
	Non-Acoustic Tests	Offshore Area (QRS)	6	None
		Inland Waters (DBRC Site, Keyport Range Site)	74	None
Acoustic Component Test	Countermeasures Testing	Offshore Area (QRS)	6	None

Range Activity		**Location**	**Action**	
			No. of events (per year)	**Ordnance (Number per year)**
		Inland Waters (DBRC Site, Keyport Range Site)	61	None
	Acoustic Test Facility	Inland Waters (DBRC Site, Keyport Range Site)	176	None
	Pierside Integrated Swimmer Defense	Inland Waters (DBRC Site, Keyport Range Site)	38	None
Naval Surface Warfare Center, Carderock Division Detachment Puget Sound				
System, Subsystem and Component Testing	Pierside Acoustic Testing	Inland Waters NAVBASE Kitsap Bangor and Bremerton	60	None
	Performance Testing At Sea	Inland Waters (DBRC Site, Carr Inlet)	60	None
	Development Training and Testing	Inland Waters (DBRC Site, Carr Inlet)	36	None
Proof of Concept Testing		Inland Waters (DBRC Site, Carr Inlet)	30	None
Naval Surface Warfare Center, Carderock Division, Southeast Alaska Acoustic Measurement Facility				
Surface Vessel Acoustic Measurement		Western Behm Canal, AK	12	None
Underwater Vessel Acoustic Measurement		Western Behm Canal, AK	26	None
Underwater Vessel Hydrodynamic Performance Measurement		Western Behm Canal, AK	3	None
Cold Water Training		Western Behm Canal, AK	1	None
Component System Testing		Western Behm Canal, AK	4	None
Countermeasures Testing		Western Behm Canal, AK	4	None
Electromagnetic Measurement		Western Behm Canal, AK	5	None
Measurement System Repair & Replacement		Western Behm Canal, AK	1	None
Project Operations (POPS)		Western Behm Canal, AK	3	None
Target Strength Trial		Western Behm Canal, AK	1	None
Additional Naval Sea Systems Command Testing Activities				
Life Cycle Activities	Pierside Sonar Testing	Inland Waters (Naval Station Everett, NAVBASE Kitsap Bangor, NAVBASE Kitsap Bremerton)	67	None

Range Activity		**Location**	**Action**	
			No. of events (per year)	**Ordnance (Number per year)**
Shipboard Protection Systems and Swimmer Defense Testing	Pierside Integrated Swimmer Defense	Inland Waters (NBK Bangor or Keyport)	1	None
Testing	Unmanned Vehicle Development and Payload Testing	Inland Waters (DBRC Site, Keyport Range Site)	4	None
Anti-Surface Warfare (ASUW)/Anti-Submarine Warfare (ASW) Testing	Torpedo (Explosive) Testing	Offshore Area	3	6 HE torpedoes 6 NEPM torpedoes
	Torpedo (Non-explosive) Testing	Offshore Area	3	18 NEPM torpedoes
	Countermeasure Testing	Inland Waters (DBRC Range Site, Pierside Naval Station Everett	13	21 NEPM torpedoes
		Offshore Area (QRS)	8	123 NEPM torpedoes
New Ship Construction	ASW Mission Package Testing	Offshore Area	8	16 NEPM

Notes: QRS = Quinault Range Site, DBRC = Dabob Bay Range Complex, AK = Alaska, NAVBASE = Naval Base, HE = High Explosive, NEPM = Non-explosive Practice Munition

Table 6. Naval Air Systems Command Testing Activity Levels

Range Activity	Location	Action	
		No. of events (per year)	**Ordnance (Number per year)**
Anti-Submarine Warfare (ASW)			
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (DICASS)	Offshore Area	28	170 DICASS sonobuoys
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (MAC)	Offshore Area	14	170 MAC sonobuoys
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (SUS)	Offshore Area	5	72 Impulsive SUS buoys (e.g., MK-61, MK-64, MK-82) 12 Non-impulsive SUS buoys (e.g., MK-84)
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (IEER)	Offshore Area	6	70 IEER sonobuoys
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (HDC)	Offshore Area	1	(16 HDC sonobuoys)
Electronic Warfare (EW)			
Flare Test	Offshore Area	10	600 flares

Notes: All of the following are types of sonobuoys to be tested: DICASS = Directional Command Activated Sonobuoy System;

2.3 Navy Mitigation Measures to Minimize or Avoid Exposure to Stressors

Proposed Navy mitigation measures are presented in a summary table below. These measures are discussed in detail in Chapter 5 (Standard Operating Procedures, Mitigation, and Monitoring) of the NWTT EIS/OEIS. The mitigation measures can be grouped into two categories: Lookout Procedural Measures and Mitigation Zone Procedural Measures and are considered as part of the proposed training and testing actions.

2.3.1 Lookout Procedural Measures

The use of Lookouts is a critical component of Navy procedural measures and implementation of mitigation zones. 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 job responsibilities that extend beyond looking at the water or air (such as supervision of other personnel). This section will introduce Lookouts who perform similar duties to watch personnel and whose duties satisfy safety of navigation and mitigation requirements.

The Navy will have two types of Lookouts for the purposes of conducting visual observations: those positioned on ships; and those positioned ashore, 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, 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, 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 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.

Some testing activities are conducted close enough to shore that observers located at shore sites have a clear view of the activities as they are conducted, and benefit from advanced systems (improved optics, acoustic detection) available for detection of animals. The procedural measures described below in Table 7 primarily consist of having Lookouts during specific training and testing activities.

Table 7. Summary of Navy Mitigation Measures for Training and Testing Activities

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Specialized Training		
Marine Species Awareness Training (Modules 1 through 4)	Training: Applicable personnel will complete the United States Navy Marine Species Awareness Training prior to standing watch or serving as a Lookout. Testing: Same as Training	Training: The mitigation zones observed by Lookouts are specified for each Mitigation Zone Procedural Measure below. Testing: Same as Training
Acoustic Stressors – Sonar and Other Active Acoustic Sources		
Low-Frequency and Hull-Mounted Mid-Frequency Active Sonar during Anti-Submarine Warfare and Mine Warfare	Training: 2 Lookouts (general), 1 Lookout (minimally manned, moored, or anchored) Testing: 2 Lookouts (general), 1 Lookout (small boats, minimally manned, moored, anchored, pierside, or shore-based)	Training: 1,000 yd. (920 m) and 500 yd. (460 m) power downs and 200 yd. (180 m) shutdown for cetaceans and sea turtles (excludes bow-riding dolphins), 100 yd. (90 m) mitigation zone for pinnipeds (excludes haulouts). Testing: 1,000 yd. (920 m) and 500 yd. (460 m) power downs for sources that can be powered down and 200 yd. (180 m) shutdown for cetaceans, 100 yd. (90 m) for pinnipeds
High-Frequency and Non-Hull-Mounted Mid-Frequency Active Sonar	Training: 2 Lookouts (general), 1 Lookout (minimally manned, moored, or anchored) Testing: 2 Lookouts (general), 1 Lookout (minimally manned, moored, anchored, and aircraft systems testing).	Training: 200 yd. (180 m) for marine mammals and concentrations of floating vegetation. Testing: 200 yd. (180 m) for marine mammals and (100 yd. [90 m] for pinnipeds) from intended track of the test unit (excludes haulouts).
Explosive and Impulse Sound		
Improved Extended Echo Ranging Sonobuoys	Training: (not applicable) Testing: 1 Lookout	Training: n/a Testing: 600 yd. (550 m) for marine mammals, sea turtles, and concentrations of floating vegetation
Explosive Signal Underwater Sound buoys using >0.5–2.5 lb. NEW	Training: 1 Lookout Testing: 1 Lookout	Training: 350 yd. (320 m) for marine mammals, sea turtles, and concentrations of floating vegetation. Testing: Same as Training
Mine Countermeasures and Mine Neutralization using Positive	Training: 2 Lookouts (1 each on 2	Training: 400 yd. (366 m) for >0.5–2.5 lb. charge for marine mammals,

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Control Firing Devices	survey boats) Testing: n/a	sea turtles. Testing: n/a
Gunnery Exercises – Small- or Medium-Caliber using a Surface Target	Training: 1 Lookout Testing: n/a	Training: 200 yd. (180 m) for marine mammals, sea turtles, and concentrations of floating vegetation. Testing: n/a
Gunnery Exercises – Large-Caliber Explosive Rounds using a Surface Target	Training: 1 Lookout Testing: n/a	Training: 600 yd. (550 m) around target for marine mammals, sea turtles, and concentrations of floating vegetation. Testing: n/a
Missile Exercises (Including Rockets) Up to 250 lb. NEW using a Surface Target	Training: 1 Lookout Testing: n/a	Training: 900 yd. (820 m) for marine mammals, sea turtles, and concentrations of floating vegetation. Testing: n/a
Missile Exercises up to 500 lb. NEW using a Surface Target	Training: 1 Lookout Testing: n/a	Training: 2,000 yd. (1.8 km) for marine mammals, sea turtles, and concentrations of floating vegetation. Testing: n/a
Explosive and Non-Explosive Bombing Exercises	Training: 1 Lookout Testing: n/a	Training: 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. Testing: n/a
Torpedo (Explosive) Testing	Training: n/a Testing: Surface ship – 2 Lookouts Aircraft – 1 Lookout	Training: n/a Testing: 2,100 yd. (1.9 km) for marine mammals, sea turtles, and concentrations of floating vegetation.

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Sinking Exercises	n/a	n/a
Weapons Firing Noise During Gunnery Exercises – Large-Caliber	Training: 1 Lookout Testing: n/a	Training: 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.
Physical Disturbance		
Vessel Movements	Training: 1 Lookout Testing: 1 Lookout	Training: 500 yd. (460 m) for whales. 200 yd. (183 m) for all other marine mammals (except bow-riding dolphins). Testing: Range craft shall not approach within 100 yd. (90 m) of cetaceans (bow-riding dolphins excluded, and pinnipeds excluded during test body retrieval).
Towed In-Water Device Use	Training: 1 Lookout Testing: 1 Lookout	Training: 250 yd. (229 m) for marine mammals Testing: Range craft shall not approach within 100 yd. (90 m) of marine mammals.

2.3.2 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 and testing 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. As a reference, aerial surveys are typically made by flying at 1,500 ft. (460 m) altitude or lower at the slowest safe speed.

Many of the Navy's proposed activities have mitigation measures that are currently being implemented, as required by previous environmental documents or consultations. Most of the current Phase I (e.g., Northwest Training Range Complex EIS/OEIS) mitigation zones for activities that involve the use of impulse and non-impulse sources were originally designed to reduce the potential for onset of temporary threshold shift (TTS). For the NWTT EIS/OEIS, 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 and Sea Turtles for the Northwest Training and Testing Environmental Impact Statement/Overseas Environmental Impact Statement technical report (Marine Species Modeling Team 2014).

As a result of the updates to the acoustic propagation modeling, in some cases the ranges to onset of TTS effects are much larger than those output by previous Phase I models. Due to the ineffectiveness and unacceptable operational impacts associated with mitigating these large areas, the Navy is unable to mitigate for onset of TTS for every activity. The Navy developed each recommended mitigation zone to avoid or reduce the potential for onset of the lowest level of injury, permanent threshold shift (PTS), 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 most cases, the predicted maximum range to PTS also consequently covers the predicted average range to TTS.

Table 8 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. The predicted ranges are based on local environmental conditions and are unique to the NWTT Action Area.

Table 8. The range to effects for activities using sonar and other active acoustic sources used in the Inland

Activity Category	Representative Source (Bin)	Predicted Average Range to TTS	Predicted Average Range to PTS	Predicted Maximum Range to PTS	Recommended Mitigation Zone
Sonar and Other Active Acoustic Sources					
Low-Frequency and Hull-Mounted Mid-Frequency Active Sonar	SQS-53 ASW hull-mounted sonar (MF1)	4,251 yd. (3,887 m)	281 yd. (257 m)	< 292 yd. (< 267 m)	Training: 1,000 yd. (920 m) and 500 yd. (460 m) power downs and 200 yd. (180 m) shutdown for cetaceans and sea turtles, 100 yd. (90 m) mitigation zone for pinnipeds Testing: 1,000 yd. (920 m) and 500 yd. (460 m) power downs for sources that can be powered down and 200 yd. (180 m) shutdown for cetaceans, 100 yd. (90 m) for pinnipeds
High-Frequency and Non-Hull-Mounted Mid-Frequency Active Sonar2	AQS-22 ASW dipping sonar (MF4)	226 yd. (207 m)	< 55 yd. (< 50 m)	< 55 yd. (< 50 m)	Training: 200 yd. (180 m) Testing: 200 yd. (180 m) for cetaceans, 100 yd. (90 m) for pinnipeds
Explosive and Impulse Sound					
Improved Extended Echo Ranging Sonobuoys	Explosive sonobuoy (E4)	237 yd. (217 m)	133 yd. (122 m)	235 yd. (215 m)	Training: n/a Testing: 600 yd. (550 m)
Signal Underwater Sound (SUS) buoys using >0.5–2.5 lb. NEW	Explosive sonobuoy (E3)	178 yd. (163 m)	92 yd. (84 m)	214 yd. (196 m)	Training: 350 yd. (320 m) Testing: 350 yd. (320 m)
Mine Countermeasure and Neutralization	2.5 lb NEW (E3)	495 yd. (453 m)	145 yd. (133 m)	373 yd. (341 m)	Training: 400 yd. (366 m) Testing: n/a

Activity Category	Representative Source (Bin)	Predicted Average Range to TTS	Predicted Average Range to PTS	Predicted Maximum Range to PTS	Recommended Mitigation Zone
Activities (positive control)					
Gunnery Exercises – Small- and Medium-Caliber (Surface Target)	25 mm projectile (E1)	72 yd. (66 m)	48 yd. (44 m)	73 yd. (67 m)	Training: 200 yd. (180 m) Testing: n/a
Gunnery Exercises – Large-Caliber (Surface Target)	5 in. projectiles (E5 at the surface)3	210 yd. (192 m)	110 yd. (101 m)	177 yd. (162 m)	Training: 600 yd. (550 m) Testing: n/a
Missile Exercises up to 500 lb. NEW (Surface Target)	Harpoon missile (E10)	1,164 yd. (1,065 m)	502 yd. (459 m)	955 yd. (873 m)	Training: 2,000 yd. (1.8 km) Testing: n/a
Bombing Exercises	MK-84 2,000 lb. bomb (E12)	1,374 yd. (1,256 m)	591 yd. (540 m)	1,368 yd. (1,251 m)	Training: 2,500 yd. (2.3 km) Testing: n/a
Lightweight Torpedo (Explosive) Testing	MK-46 torpedo (E8)	497 yd. (454 m)	245 yd. (224 m)	465 yd. (425 m)	Training: n/a Testing: 2,100 yd. (1.9 km)
Heavyweight Torpedo (Explosive) Testing	MK-48 torpedo (E11)	1,012 yd. (926 m)	472 yd. (432 m)	885 yd. (809 m)	Training: n/a Testing: 2,100 yd. (1.9 km)

For pierside maintenance and testing of hull-mounted mid-frequency sources in the Inland Waters, modeling provides an overestimate of the range to effects because it cannot adequately account for the complex interactions of the sound energy into very shallow water and associated shorelines, the loss into dampening structures (i.e., such as adjacent pilings, jetties, or seawalls), or occasions when a ship or submarine is moored bow in so that the sonar is transmitted toward the nearby shoreline. Therefore, the ranges in Table 8 are even more protective for activities in the Inland Waters.

For some activities, Lookouts may be 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 (kelp paddies) 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 and testing will not cease as a result of indicators entering the mitigation zone after activities have commenced. This measure is intended only for floating vegetation detached from the seafloor.

The activity-specific mitigation zones are based on the longest range for all the functional hearing groups (based on the hearing threshold metrics described in Section 3.4, Marine Mammals, and Section 3.5, Sea Turtles, in the NWTT EIS/OEIS). The mitigation zone for a majority of activities is driven by either the high-frequency cetaceans or the sea turtles functional hearing groups. Therefore, the mitigation zones are even more protective for the remaining functional 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.

2.3.3 Other Procedural Measures in the NWTT Study Area

The Navy has considered whether additional mitigation measures are warranted for certain activities or in certain locations in the NWTT Study Area. The following are particular additional mitigation measures the Navy intends to apply. These are further explained in the Navy's NWTT EIS/OEIS.

The Navy conducts pre-event planning and training to ensure environmental awareness of all exercise participants prior to Maritime Homeland Defense/Security Mine Countermeasure Integrated Exercises. When an event is proposed to be conducted in Puget Sound, Navy event planners will consult with Navy biologists who will contact NMFS during the planning process. NMFS and the Navy biologist will determine likelihood of gray whale or southern resident killer whale presence in the proposed exercise area. The Navy biologist will convey that information to event planners as planners consider specific details (timing, location, duration) of the event.

The Navy conducts pre-event planning and training to ensure environmental awareness of all exercise participants prior to Small Boat Attack Gunnery exercises. When an event is proposed to be conducted in and around Naval Station Everett, Naval Base Kitsap Bangor, or Naval Base Kitsap Bremerton in Puget Sound, Navy event planners will consult with Navy biologists who will contact NMFS early in the planning process. NMFS and the Navy biologist will determine the extent marine mammals may be present in the immediate vicinity of proposed exercise area. The Navy biologist will convey that information to event planners as planners consider the specific details (timing, location, duration) of the event.

The Navy will conduct Missile Exercises and Bombing Exercises using high explosives 50 nm or further from of the shore in the NWTRC Offshore Area.

The Navy will conduct Bombing Exercises (non-explosive practice munitions) greater than 20 nm from shore and not within the Olympic Coast National Marine Sanctuary.

The Navy will continue to require units to obtain approval from U.S. Third Fleet prior to any training or testing mine countermeasure and neutralization underwater detonations at Hood Canal or Crescent Harbor.

The Navy will continue to require units to obtain approval from U.S. Pacific Fleet's designated authority prior to use of hull-mounted mid-frequency active sonar during training while underway in Puget Sound and Strait of Juan de Fuca.

The Navy will continue to require units to obtain approval from U.S. Pacific Fleet's designated authority or Systems Command designated authority, as applicable, prior to conducting ship and submarine active sonar pierside maintenance/testing in Puget Sound or the Strait of Juan de Fuca.

NMFS is aware of and has considered additional mitigation measures the Navy intends to comply with as a result of ESA and other consultations as well. These include:

During February, March, and April (the juvenile migration period for Hood Canal summer-run chum), the Navy will not conduct any of the planned larger size (>0.5-2.5 lb) detonations in Hood Canal. During this time period the unit will only conduct training with the SWAG charges, which are 1 ounce or less in net explosive weight.

During August, September, and October (adult migration period for Hood Canal summer-run chum and Puget Sound Chinook), the Navy will also avoid to the maximum extent practicable (unless necessitated by readiness requirements), the use of the larger size charges (>0.5-2.5 lb) and only conduct training with 1 ounce or less charges.

2.4 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 testing and training activities within the NWTT Study Area. Several factors are considered under MMPA authorizations including: (1) specific activity and geographical region where "take" make occur, (2) dates when "take" may occur, (3) permissible methods of taking, (4) prohibitions on taking of marine mammals, (5) mitigation required, (6) monitoring and reporting requirements, (7) how LOAs may be applied for, (8) specifics of LOAs, and (9) renewals and modifications of LOAs and adaptive management processes.

In order to promulgate the Final Rule, 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 reiterated here.

2.4.1 Taking and Importing Marine Mammals; U.S. Navy's Northwest Training and Testing (NWTT) Study Area

The take of ESA-listed species by harassment incidental to the Navy's training and testing activities in the NWTT Study Area authorized pursuant to NMFS Permit Division's proposed MMPA rule is presented the following sections.

2.4.1.1 ***§ 218.140 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 NWTT Study Area, which is composed of established maritime operating and warning areas in the eastern North Pacific Ocean region, including areas of the Strait of Juan de Fuca, Puget Sound, and Western Behm Canal in southeastern Alaska. The Study Area includes air and water space within and outside Washington state waters, and outside state waters of Oregon and Northern California. The Study Area includes four existing range complexes and facilities: the Northwest Training Range Complex (NWTRC), the Keyport Range Complex, Carr Inlet Operations Area, and SEAFAC. In addition to these range complexes, the Study Area also includes Navy pierside locations where sonar maintenance and testing occurs as part of overhaul, modernization, maintenance and repair activities at NAVBASE Kitsap, Bremerton; NAVBASE Kitsap, Bangor; and Naval Station Everett.

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

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

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

(A) MF1 – an average of 166 hours per year.

(B) MF3 – an average of 70 hours per year.

(C) MF4 – an average of 4 hours per year.

(D) MF5 – an average of 896 items per year.

(E) MF11 – an average of 16 hours per year.

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

(A) HF1 – an average of 48 hours per year.

(B) HF4 – an average of 384 hours per year.

(C) HF6 – an average of 192 hours per year

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

(A) ASW2 – an average of 720 items per year per year.

(B) ASW3 – an average of 78 hours per year.

(2) Sonar and other Active Sources Used During Testing:

(i) Low-frequency (LF) Source Classes:

(A) LF4 – an average of 110 hours per year.

(B) LF5 – an average of 71 hours per year.

(ii) Mid-frequency (MF):

(A) MF1 – an average of 32 hours per year

(B) MF3 – an average of 145 hours per year.

(C) MF4 – an average of 10 hours per year.

(D) MF5 – an average of 273 items per year.

(E) MF6 – an average of 12 items per year.

(F) MF8 – an average of 40 hours per year.

(G) MF9 – an average of 1,183 hours per year.

(H) MF10 – an average of 1,156 hours per year.

(I) MF11 – an average of 34 hours per year.

(J) MF12 – an average of 24 hours per year.

(iii) High-frequency (HF) and Very High-frequency (VHF):

(A) HF1 – an average of 161 hours per year.

(B) HF3 – an average of 145 hours per year.

(C) HF5 – an average of 360 hours per year.

(D) HF6 – an average of 2,099 hours per year.

(iv) VHF:

(A) VHF2 - an average of 35 hours per year.

(v) ASW:

(A) ASW1 – an average of 16 hours per year.

(B) ASW2 – an average of 64 hours per year.

(C) ASW2 – an average of 170 items per year.

(D) ASW3 – an average of 444 hours per year.

(E) ASW4 – an average of 1,182 items per year.

(vi) Acoustic Modems (M):

(A) M3 - an average of 1,519 hours per year.

(vii) Torpedoes (TORP):

(A) TORP1 – an average of 315 items per year.

(B) TORP2 – an average of 299 items per year.

(viii) Swimmer Detection Sonar (SD):

(A) SD1 – an average of 757 hours per year.

(ix) Synthetic Aperture Sonar (SAS):

(A) SAS2 – an average of 798 hours per year.

(3) Impulsive Source Detonations During Training:

(i) Explosive Classes:

(A) E1 (0.1 to 0.25 pound [lb] NEW) – an average of 48 detonations per year.

(B) E3 (>0.5 to 2.5 lb NEW) – an average of 6 detonations per year.

(C) E5 (>5 to 10 lb NEW) – an average of 80 detonations per year.

(D) E10 (>250 to 500 lb NEW) – an average of 4 detonations per year.

(E) E12 (>650 to 1,000 lb NEW) – an average of 10 detonations per year.

(ii) [Reserved]

(4) Impulsive Source Detonations During Testing:

(i) Explosive Classes:

(A) E3 (>0.5 to 2.5 lb NEW) – an average of 72 detonations per year.

(B) E4 (>2.5 to 5 lb NEW) – an average of 140 detonations (70 sonobuoys) per year.

(C) E8 (>60 to 100 lb NEW) – an average of 3 detonations per year.

(D) E11 (>500 to 650 lb NEW) – an average of 3 detonations per year.

(ii) [Reserved]

2.4.1.2 *§ 218.141 Effective dates*

Regulations in this subpart are effective November 9, 2015 through November 8, 2020.

2.4.1.3 *§ 218.142 Permissible methods of taking.*

(a) Under Letters of Authorization (LOAs) issued pursuant to § 218.147, the Holder of, and those operating under, the LOA may incidentally, but not intentionally, take marine mammals within the area described in § 218.140, provided the activity is in compliance with all terms, conditions, and requirements of these regulations and the appropriate LOA.

(b) The activities identified in § 218.140(c) must be conducted in a manner that minimizes, to the greatest extent practicable, any adverse impacts on marine mammals and their habitat.

(c) The incidental take of marine mammals under the activities identified in § 218.140(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 – 25 (an average of 5 per year).

(B) Fin whale (*Balaenoptera physalus*), California, Oregon, and Washington (CA/OR/WA) – 125 (an average of 25 per year).

(C) Gray whale (*Eschrichtius robustus*), Eastern North Pacific – 30 (an average of 6 per year).

(D) Humpback whale (*Megaptera novaeangliae*), CA/OR/WA – 60 (an average of 12 per year).

(E) Minke whale (*Balaenoptera acutorostrata*), CA/OR/WA – 90 (an average of 18 per year).

(ii) Odontocetes:

(A) Baird's beaked whale (*Berardius bairdii*), CA/OR/WA – 2,955 (an average of 591 per year).

(B) Mesoplodont beaked whale (*Mesoplodon* spp.), CA/OR/WA – 7,085 (an average of 1,417 per year).

(C) Cuvier's beaked whale (*Ziphius cavirostris*), CA/OR/WA – 1,765 (an average of 353 per year).

(D) Dall's porpoise (*Phocoenoidea dalli*), CA/OR/WA – 18,178 (an average of 3,730 per year).

(E) Harbor porpoise (*Phocoena phocoena*), Northern OR/WA Coast – 175,030 (an average of 35,006 per year).

(F) Harbor porpoise (*Phocoena phocoena*), Northern CA/Southern OR – 262,545 (an average of 52,509 per year).

(G) Harbor porpoise (*Phocoena phocoena*), WA Inland Waters – 4,409 (an average of 1,417 per year).(H) Killer whale (*Orcinus orca*), West Coast Transient – 39 (an average of 9 per year).

(I) Killer whale (*Orcinus orca*), Eastern North Pacific Offshore – 65 (an average of 13 per year).

(J) Killer whale (*Orcinus orca*), Eastern North Pacific Southern Resident – 6 (an average of 2 per year).

(K) *Kogia* spp., CA/OR/WA – 365 (an average of 73 per year).

(L) Northern right whale dolphin (*Lissodelphis borealis*), CA/OR/WA – 6,660 (an average of 1,332 per year).

(M) Pacific white-sided dolphin (*Lagenorhynchus obliquidens*), CA/OR/WA – 17,408 (an average of 3,482 per year).

(N) Risso's dolphin (*Grampus griseus*), CA/OR/WA – 3,285 (an average of 657 per year).

(O) Short-beaked common dolphin (*Delphinus delphis*), CA/OR/WA – 3,670 (an average of 734 per year).

(P) Sperm whale (*Physeter macrocephalus*), CA/OR/WA – 405 (an average of 81 per year).

(Q) Striped dolphin (*Stenella coerulealba*), CA/OR/WA – 110 (an average of 22 per year).

(iii) Pinnipeds:

(A) California sea lion (*Zalophus californianus*), U.S. – 4,038 (an average of 814 per year).

(B) Steller sea lion (*Eumetopias jubatus*), Eastern U.S. – 1,986 (an average of 404 per year).

(C) Guadalupe fur seal (*Arctocephalus townsendi*), Mexico – 35 (an average of 7 per year).

(D) Harbor seal (*Phoca vitulina*), WA Northern Inland Waters – 1,855 (an average of 427 per year).

(E) Harbor seal (*Phoca vitulina*), Southern Puget Sound – 252 (an average of 58 per year).

(F) Harbor seal (*Phoca vitulina*), Hood Canal – 2,054 (an average of 452 per year).

(G) Northern elephant seal (*Mirounga angustirostris*), CA Breeding – 6,353 (an average of 1,271 per year).

(H) Northern fur seal (*Callorhinus ursinus*), Eastern Pacific – 12,475 (an average of 2,495 per year).

(I) Northern fur seal (*Callorhinus ursinus*), California – 185 (an average of 37 per year).

(2) Level A Harassment for all Training Activities:

(i) Mysticetes:

(A) [Reserved]

(B) [Reserved]

(ii) Odontocetes:

Dall's porpoise (*Phocoenoidea dalli*), CA/OR/WA – 20 (an average of 4 per year).

Harbor porpoise (*Phocoena phocoena*), WA Inland Waters – 5 (an average of 1 per year).

(iii) Pinnipeds:

Harbor seal (*Phoca vitulina*), WA Northern Inland Waters – 20 (an average of 4 per year).

Harbor seal (*Phoca vitulina*), Hood Canal – 10 (an average of 2 per year).

[Reserved]

(3) Level B Harassment for all Testing Activities:

(i) Mysticetes:

(A) Blue whale (*Balaenoptera musculus*), Eastern North Pacific – 30 (an average of 6 per year).

(B) Fin whale (*Balaenoptera physalus*), CA/OR/WA – 170 (an average of 34 per year).

(C) Fin whale (*Balaenoptera physalus*), Northeast Pacific – 10 (an average of 2 per year).

(D) Gray whale (*Eschrichtius robustus*), Eastern North Pacific – 60 (an average of 12 per year).

(E) Humpback whale (*Megaptera novaeangliae*), Central North Pacific – 5 (an average of 1 per year).

(F) Humpback whale (*Megaptera novaeangliae*), CA/OR/WA – 220 (an average of 44 per year).

(G) Minke whale (*Balaenoptera acutorostrata*), CA/OR/WA – 90 (an average of 18 per year).

(H) Sei whale (*Balaenoptera borealis*), Eastern North Pacific – 10 (an average of 2 per year).

(ii) Odontocetes:

(A) Baird's beaked whale (*Berardius bairdii*), Alaska – 125 (an average of 25 per year).

(B) Baird's beaked whale (*Berardius bairdii*), CA/OR/WA – 745 (an average of 149 per year).

(C) Mesoplodont beaked whale (*Mesoplodon* spp.), CA/OR/WA – 1,845 (an average of 369 per year).

(D) Cuvier's beaked whale (*Ziphius cavirostris*), Alaska – 75 (an average of 15 per year).

(E) Cuvier's beaked whale (*Ziphius cavirostris*), CA/OR/WA – 455 (an average of 91 per year).

(F) Dall's porpoise (*Phocoenoidea dalli*), Alaska – 6,000 (an average of 1,200 per year).

(G) Dall's porpoise (*Phocoenoidea dalli*), CA/OR/WA – 50,785 (an average of 10,157 per year).

(H) Harbor porpoise (*Phocoena phocoena*), Southeast Alaska – 4,630 (an average of 926 per year).

(I) Harbor porpoise (*Phocoena phocoena*), Northern OR/WA Coast – 86,060 (an average of 17,212 per year).

(J) Harbor porpoise (*Phocoena phocoena*), Northern CA/Southern OR – 129,095 (an average of 25,819 per year).

(K) Harbor porpoise (*Phocoena phocoena*), WA Inland Waters – 27,045 (an average of 5,409 per year).

(L) Killer whale (*Orcinus orca*), Alaska Resident – 10 (an average of 2 per year).

(M) Killer whale (*Orcinus orca*), West Coast Transient – 1,035 (an average of 207 per year).

(N) Killer whale (*Orcinus orca*), Eastern North Pacific Offshore – 110 (an average of 22 per year).

(O) *Kogia* spp., CA/OR/WA – 530 (an average of 106 per year).

(P) Northern right whale dolphin (*Lissodelphis borealis*), CA/OR/WA – 10,190 (an average of 2,038 per year).

(Q) Pacific white-sided dolphin (*Lagenorhynchus obliquidens*), North Pacific – 15 (an average of 3 per year).

(R) Pacific white-sided dolphin (*Lagenorhynchus obliquidens*), CA/OR/WA – 24,345 (an average of 4,869 per year).

(S) Risso's dolphin (*Grampus griseus*), CA/OR/WA – 5,770 (an average of 1,154 per year).

(T) Short-beaked common dolphin (*Delphinus delphis*), CA/OR/WA – 8,140 (an average of 1,628 per year).

(U) Sperm whale (*Physeter macrocephalus*), CA/OR/WA – 390 (an average of 78 per year).

(V) Striped dolphin (*Stenella coeruleoalba*), CA/OR/WA – 70 (an average of 14 per year).

(iii) Pinnipeds:

(A) California sea lion (*Zalophus californianus*), U.S. – 10,730 (an average of 2,146 per year).

(B) Steller sea lion (*Eumetopias jubatus*), Eastern U.S. – 2,605 (an average of 521 per year).

(C) Guadalupe fur seal (*Arctocephalus townsendi*), Mexico – 15 (an average of 3 per year).

(D) Harbor seal (*Phoca vitulina*), Southeast Alaska (Clarence Sound) – 110 (an average of 22 per year).

(E) Harbor seal (*Phoca vitulina*), OR/WA Coast – 8,275 (an average of 1,655 per year).

(F) Harbor seal (*Phoca vitulina*), WA Northern Inland Waters – 9,115 (an average of 1,823 per year).

(G) Harbor seal (*Phoca vitulina*), Southern Puget Sound – 980 (an average of 196 per year).

(H) Harbor seal (*Phoca vitulina*), Hood Canal – 296,085 (an average of 59,217 per year).

(I) Northern elephant seal (*Mirounga angustirostris*), CA Breeding – 6,625 (an average of 1,325 per year).

(J) Northern fur seal (*Callorhinus ursinus*), Eastern Pacific – 9,150 (an average of 1,830 per year).

(K) Northern fur seal (*Callorhinus ursinus*), California – 135 (an average of 27 per year).

(4) Level A Harassment for all Testing Activities:

(i) Mysticetes:

(A) Gray whale (*Eschrichtius robustus*), Eastern North Pacific – 5 (an average of 1 per year).

(B) [Reserved]

(ii) Odontocetes:

(A) *Kogia* spp., CA/OR/WA – 5 (an average of 1 per year).

(B) Dall' porpoise (*Phocoenoidea dalli*), CA/OR/WA – 215 (an average of 43 per year).

(C) Harbor porpoise (*Phocoena phocoena*), Northern OR/WA Coast – 75 (an average of 15 per year).

(D) Harbor porpoise (*Phocoena phocoena*), Northern CA/Southern OR – 115 (an average of 23 per year).

(E) Harbor porpoise (*Phocoena phocoena*), WA Inland Waters – 30 (an average of 6 per year).

(iii) Pinnipeds:

(A) Harbor seal (*Phoca vitulina*), OR/WA Coast – 20 (an average of 4 per year).

(B) Harbor seal (*Phoca vitulina*), WA Northern Inland Waters – 110 (an average of 22 per year).

(C) Harbor seal (*Phoca vitulina*), Southern Puget Sound – 5 (an average of 1 per year).

(D) Harbor seal (*Phoca vitulina*), Hood Canal – 335 (an average of 67 per year).

(E) Northern elephant seal (*Mirounga angustirostris*), CA Breeding – 10 (an average of 2 per year).

(F) [Reserved]

2.4.1.4 ***§ 218.143 Prohibitions.***

Notwithstanding takings contemplated in § 218.142 and authorized by an LOA issued under §§ 216.106 and 218.147 of this chapter, no person in connection with the activities described in § 218.140 may:

(a) Take any marine mammal not specified in § 218.142(c);

(b) Take any marine mammal specified in § 218.142(c) other than by incidental take as specified in § 218.142(c);

(c) Take a marine mammal specified in § 218.142(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 218.147.

2.4.1.5 ***§ 218.144 Mitigation.***

(a) When conducting training and testing activities, as identified in § 218.140, the mitigation measures contained in the LOA issued under §§ 216.106 and 218.147 of this chapter must be implemented. These mitigation measures include, but are not limited to:

(1) *Lookouts* – The following are protective measures concerning the use of Lookouts.

(i) Lookouts positioned on surface ships will be dedicated solely to diligent observation of the air and surface of the water. Their observation objectives will 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) Lookouts positioned ashore, in aircraft or on boats will, to the maximum extent practicable and consistent with aircraft and boat safety and training and testing requirements, comply with the observation objectives described in paragraph (a)(1)(i) of this section.

(iii) Lookout measures for non-impulsive sound:

(A) With the exception of vessels less than 65 ft (20 m) in length or minimally manned vessels, ships using low-frequency or hull-mounted mid-frequency active sonar sources associated with anti-submarine warfare and mine warfare activities at sea will have two Lookouts at the forward position of the vessel. For the purposes of this rule, low-frequency active sonar does not include surface towed array surveillance system low-frequency active sonar.

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

(C) Ships conducting active sonar activities while moored or at anchor (including pierside or shore-based testing or maintenance) will maintain one Lookout.

(D) Minimally manned vessels conducting hull-mounted mid-frequency testing will employ one Lookout.

(E) Ships, small boats, range craft, or aircraft conducting non-hull-mounted mid-frequency active sonar, such as helicopter dipping sonar systems, will maintain one Lookout.

(F) Surface ships or aircraft conducting high-frequency or non-hull-mounted mid-frequency active sonar activities associated with anti-submarine warfare and mine warfare activities at sea will have one Lookout.

(iv) Lookout measures for impulsive sound (e.g., explosives):

(A) Aircraft conducting improved extended echo ranging sonobuoy activities will have one Lookout.

(B) Aircraft conducting explosive sonobuoy activities using >0.5 to 2.5-lb net explosive weight (NEW) will have one Lookout.

(C) General mine countermeasure and neutralization activities involving positive control diver placed charges using >0.5 to 2.5 lb NEW will have a total of two Lookouts (one Lookout positioned in each of the two support vessels). All divers placing the charges on mines will support the Lookouts while performing their regular duties. The divers and Lookouts will report all marine mammal sightings to their dive support vessel.

(D) Surface vessels or aircraft conducting small-,medium-, and large-caliber gunnery exercises will have one Lookout. Towing vessels, if applicable, will also maintain one Lookout.

(E) Aircraft conducting missile exercises against a surface target will have one Lookout.

(F) Aircraft conducting explosive bombing exercises will have one Lookout and any surface vessels involved will have trained Lookouts.

(G) During explosive torpedo testing from aircraft one Lookout will be used and positioned in an aircraft. During explosive torpedo testing from a surface ship the Lookout procedures implemented for hull-mounted mid-frequency active sonar activities will be used.

(H) To mitigate effects from weapon firing noise, ships conducting explosive and non-explosive large-caliber gunnery exercises will have one Lookout. This may be the same Lookout used for small, medium, and large-caliber gunnery exercises using a surface target when that activity is conducted from a ship against a surface target.

(v) Lookout measures for physical strike and disturbance:

(A) While underway, surface ships and range craft will have at least one Lookout.

(B) During activities using towed in-water devices towed from a manned platform, one Lookout will be used. During activities in which in-water devices are towed by unmanned platforms, a manned escort vessel will be included and one Lookout will be employed.

(C) Activities involving non-explosive practice munitions (e.g., small-, medium-, and large-caliber gunnery exercises) using a surface target will have one Lookout.

(D) During non-explosive bombing exercises one Lookout will be positioned in an aircraft and trained Lookouts will be positioned in any surface vessels involved.

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

(i) Mitigation zones will 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 will 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. (91 m) of the sonar dome. Transmissions will 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. beyond the location of the last detection, or the Lookout 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. The pinniped mitigation zone does not apply to pierside sonar in the vicinity of pinnipeds hauled out on or in the water near man-made structures and vessels.

(D) The Navy shall ensure that low-frequency active sonar transmission levels are ceased if any detected cetaceans (or sea turtles) are within 200 yd. (183 m) and pinnipeds are within 100 yd. (91 m) of the source. Transmissions will 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, or the vessel has transited more than 2,000 yd. beyond the location of the last detection. The pinniped mitigation zone does not apply for pierside sonar testing in the vicinity of pinnipeds hauled out on or in the water near man-made structures and vessels.

(E) For training, the Navy shall ensure that high-frequency and non-hull-mounted mid-frequency active sonar transmission levels are ceased if any detected marine mammals are within 200 yd. (183 m) of the source. For testing, 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. (91 m) of the source. Transmissions will 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). The pinniped mitigation zone does not apply for pierside or shore-based testing in the vicinity of pinnipeds hauled out on or in the water near man-made structures and vessels.

(iv) Mitigation zones and procedures for explosive and impulsive sound:

(A) For activities using IEER sonobuoys, mitigation will include pre-exercise aerial observation and passive acoustic monitoring, which will begin 30 minutes before the first source/receiver pair detonation and continue throughout the duration of the exercise. IEER sonobuoys will not be deployed if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone around the intended deployment location. Explosive detonations will cease if a marine mammal, sea turtle, or concentrations of floating vegetation are sighted within a 600-yd. (549 m) mitigation zone. Detonations will recommence if 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.

(B) 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 net explosive weight. Mitigation will include pre-exercise aerial monitoring of the mitigation zone during deployment. Explosive SUS buoys will not be deployed if concentrations of floating vegetation (kelp paddies) are observed within the mitigation zone around the intended deployment location. A SUS detonation will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Detonations will recommence if 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.

(C) A mitigation zone with a radius of 400 yd. (366 m) shall be established for mine countermeasures and neutralization activities using positive control firing devices. For Demolition and Mine Countermeasures Operations, pre-exercise surveys shall be conducted within 30 minutes prior to the commencement of the scheduled explosive event. The survey may be conducted from the surface, by divers, or from the air, and personnel shall be alert to the presence of any marine mammal or sea turtle. Should a marine mammal or sea turtle be present within the survey area, the explosive event shall not be started until the animal voluntarily leaves the area. The Navy will ensure the area is clear of marine mammals for a full 30 minutes prior to initiating the explosive event. Explosive detonations will cease if a marine mammal is sighted in the water portion of the mitigation zone (i.e., not on shore). Detonations will recommence if 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 200 yd. (183 m) shall be established for small- and medium-caliber gunnery exercises with a surface target. Vessels will observe the mitigation zone from the firing position. When aircraft are firing, the aircrew will maintain visual watch of the mitigation zone during the activity. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed within the mitigation zone. Firing will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Firing will recommence if 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.

(E) A mitigation zone with a radius of 600 yd. (549 m) shall be established for large-caliber gunnery exercises with a surface target. Ships will observe the mitigation zone from the firing position. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Firing will recommence if 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.

(F) A mitigation zone with a radius of 2,000 yd. (1.8 km) shall be established for missile exercises up to 500 lb NEW using a surface target. When aircraft are involved in the missile firing, mitigation will include visual observation by the aircrew prior to commencement of the activity within a mitigation zone of 2,000 yd. (1.8 km) around the intended impact location. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing will not commence or will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Firing will recommence if 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 or 30 minutes (depending on aircraft type).

(G) A mitigation zone with a radius of 2,500 yd. (2.3 km) for explosive bombs and a mitigation zone of 1,000 yd (914 m) for non-explosive bombs around the intended impact location shall be established for bombing exercises. Aircraft shall visually survey the target and buffer zone for marine mammals prior to and during the exercise. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Bombing will not commence or will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Bombing will recommence if 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.

(H) A mitigation zone with a radius of 2,100 yd. (1.9 km) shall be established for torpedo (explosive) testing. Mitigation will include visual observation by aircraft immediately before, during, and after the event of the mitigation zone. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are sighted within the mitigation zone. Firing will not commence or will cease if a marine mammal, sea turtle, or aggregation of jellyfish is sighted within the mitigation zone. Firing will recommence if 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 or 30 minutes (depending on aircraft type). In addition to visual observation, passive acoustic monitoring shall be conducted by Navy assets, such as passive ship sonar systems or sonobuoys already participating in the activity. These assets would only detect vocalizing marine mammals within the frequency band 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 shall be reported to the Lookout posted in the aircraft in order to increase vigilance of the visual surveillance, and to the person in control of the activity for their consideration in determining when the mitigation zone is determined free of visible marine mammals.

(I) A mitigation zone with a radius of 70 yd. (46 m) within 30 degrees on either side of the gun target line on the firing side shall be established for weapons firing noise during large-caliber gunnery exercises. Mitigation shall include visual observation immediately before and during the exercise. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing will cease if a marine mammal or sea turtle is sighted within the mitigation zone. Firing will recommence if 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) For all training activities and for testing activities involving surface ships, 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, and pinnipeds hauled out on man-made navigational and port structures and vessels) during vessel movements. 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) For testing activities not involving surface ships (e.g. range craft) vessels shall maneuver to keep at least 100 yd. (91 m) away from marine mammals (except bow-riding dolphins, pinnipeds hauled out on man-made navigational and port structures and vessels, and pinnipeds during test body retrieval) during vessel movements. This requirement 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.

(C) The Navy shall ensure that towed in-water devices being towed from manned platforms avoid coming within a mitigation zone of 250 yd. (230 m) for all training events and testing activities involving surface ships, and a mitigation zone of 100 yd (91 m) for testing activities not involving surface ships (e.g. range craft) around any observed marine mammal, 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. Mitigation will include visual observation from a vessel or aircraft immediately before and during the exercise within the mitigation zone of the intended impact location. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed in the mitigation zone. Firing will cease if a marine mammal is sighted within the mitigation zone. Firing will recommence if 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. (914 m) shall be established for non-explosive bombing exercises. Mitigation shall include visual observation from the aircraft immediately before the exercise and during target approach within the mitigation zone around the intended impact location. The exercise will not commence if concentrations of floating vegetation (kelp paddies) are observed within the mitigation zone. Bombing will not commence or will cease if a marine mammal is sighted within the mitigation zone. Bombing will recommence if 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) *NWTT-Specific Mitigation* – The following are additional measures the Navy shall comply with when conducting training or testing activities in the NWTT Study Area:

(i) Maritime Homeland Defense/Security Mine Countermeasure Integrated Exercises – The Navy shall conduct pre-event planning and training to ensure environmental awareness of all exercise participants. When this event is proposed to be conducted in Puget Sound, Navy event planners shall consult with Navy biologists who shall contact NMFS during the planning process in order to determine likelihood of gray whale or southern resident killer whale presence in the proposed exercise area as planners consider specifics of the event.

(ii) Small Boat Attack Gunnery Exercises - The Navy shall conduct pre-event planning and training to ensure environmental awareness of all exercise participants. When this event is proposed to be conducted in and around Naval Station Everett, Naval Base Kitsap Bangor, or Naval Base Kitsap Bremerton in Puget Sound, Navy event planners shall consult with Navy biologists who shall contact NMFS early in the planning process in order to determine the extent marine mammals may be present in the immediate vicinity of the proposed exercise area as planners consider the specifics of the event.

(iii) Missile Exercise - The Navy shall conduct Missile Exercises using high explosives at least 50 nm from shore in the NWTT Offshore Area.

(iv) BOMBEX – The Navy shall conduct BOMBEX (high explosive munitions) greater than 50 nm from shore.

(v) BOMBEX (non-explosive practice munitions) - The Navy shall conduct BOMBEX (non-explosive practice munitions) events at least 20 nm from shore and shall not conduct BOMBEX events within the Olympic Coast National Marine Sanctuary.

(vi) Mine Countermeasure and Neutralization Underwater Detonations – The Navy shall require approval from U.S. Third Fleet prior to conducting mine countermeasure and neutralization underwater detonations at Hood Canal or Crescent Harbor.

(vii) Hull Mounted Mid-Frequency Active Sonar Training - The Navy shall require approval from U.S. Pacific Fleet's designated authority prior to conducting hull-mounted mid-frequency active sonar on vessels while training underway in Puget Sound and the Strait of Juan de Fuca.

(viii) Pierside Maintenance or Testing of Sonar Systems – The Navy shall require approval from U.S. Pacific Fleet's designated authority or Systems Command designated authority (as applicable to ship and submarine active sonar use) prior to conducting pierside maintenance or testing in Puget Sound or the Strait of Juan de Fuca.

2.4.1.6 ***§ 218.145 Requirements for monitoring and reporting.***

(a) The Navy is required to cooperate with the NMFS, and any other Federal, state or local agency monitoring the impacts of the activity on marine mammals.

(b) General Notification of Injured or Dead Marine Mammals - Navy personnel shall ensure that NMFS is notified immediately (or as soon as clearance procedures allow) if an injured, stranded, or dead marine mammal is found during or shortly after, and in the vicinity of, any Navy training exercise utilizing MFAS, HFAS, or underwater explosive detonations. The Navy will 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 will report the same information as listed above as soon as operationally feasible and clearance procedures allow.

(c) 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)

(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

(d) Event Communication Plan - The Navy 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 as described in the proposed notification measures above.

(e) The Navy must conduct all monitoring and/or research required under the Letter of Authorization including abiding by the NWTT monitoring plan. (*http://www.nmfs.noaa.gov/pr/permits/incidental/military.htm*).

(f) Annual NWTT Monitoring Report - The Navy shall submit an annual report of the NWTT monitoring describing the implementation and results of the NWTT monitoring efforts from the previous calendar year. Data collection methods will be standardized across range complexes and study areas to allow for comparison in different geographic locations. Although additional information will be gathered, the protected species observers collecting marine mammal data pursuant to the NWTT monitoring plan shall, at a minimum, provide the same marine mammal observation data required in § 218.145. 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 NWTT 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 NWTT Exercise and Testing Reports - The Navy shall submit preliminary reports detailing the status of authorized sound sources within 21 days after the anniversary of the date of issuance of the LOA. The Navy shall submit detailed reports 3 months after the annual anniversary of the date of issuance of the LOA. The detailed annual reports shall describe the level of training and testing conducted during the reporting period, and a summary of sound sources used (total annual hours or quantity [per the LOA] of each bin of sonar or other non-impulsive source; total annual number of each type of explosive exercises; total annual expended/detonated rounds [missiles, bombs, etc.] for each explosive bin; and improved Extended Echo-Ranging System (IEER)/sonobuoy summary, including total number of IEER events conducted in the Study Area, total expended/detonated rounds (buoys), and total number of self-scuttled IEER rounds. The analysis in the detailed reports will be based on the accumulation of data from the current year's report and data collected from previous reports.

The annual classified exercise reports will also include the amount of hull-mounted mid-frequency and high frequency active sonar use during training and testing activities in the months specified for the following three feeding areas, (to the extent that active sonar training or testing does occur in these feeding areas): the Humpback Whale Northern Washington feeding area (May through November); the Stonewall and Heceta Bank feeding area (May through November) and the Gray Whale Northern Puget Sound Feeding Area (March through May).

(h) 5-year Close-out Exercise and Testing Report—This report will be included as part of the 2020 annual exercise or testing report. This report will 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 will 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 will be submitted 3 months after the expiration of the rule. NMFS will submit comments on the draft close-out report, if any, within 3 months of receipt. The report will 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.

2.4.1.7 ***§ 218.146 Applications for Letters of Authorization.***

To incidentally take marine mammals pursuant to the regulations in this subpart, the U.S. citizen (as defined by § 216.106) conducting the activity identified in § 218.140(c) (the U.S. Navy) must apply for and obtain either an initial LOA in accordance with § 218.147 or a renewal under § 218.148.

2.4.1.8 ***§ 218.147 Letters of Authorization.***

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

(b) Each LOA will 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, modification, or renewals of LOAs will be based on a determination that the total number of marine mammals taken by the activity as a whole will have no more than a negligible impact on the affected species or stock of marine mammal(s).

2.4.1.9 *§ 218.148 Renewals and Modifications of Letters of Authorization and Adaptive Management.*

(a) A Letter of Authorization issued under §§ 216.106 and 218.147 of this chapter for the activity identified in § 218.140(c) will 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 adequately 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 § 218.147 of this chapter for the activity identified in § 218.144 of this chapter may be modified by NMFS under the following circumstances:

(1) Adaptive Management – NMFS may modify (including add to, change, or remove) 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 set forth in the preamble for these regulations.

(i) Possible sources of data that could contribute to the decision to modify the mitigation, monitoring, and reporting measures in an LOA include (but are not limited to):

(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 LOAs.

(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 § 218.142(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.

2.5 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 encompasses the NWTT Study Area, transit corridor, and the area outside of the Study Area where direct and effects of stressors from Navy training and testing activities could be experienced. Figure 1 provides an overview of the NWTT Action Area which includes the inshore and offshore portions of the Northwest Training Range Complex (NWTRC) in Northern California, Oregon, and Washington State, the Naval Undersea Warfare Center (NUWC) Keyport Range Complex in and around Puget Sound, and the Southeast Alaska Acoustic Measurement Facility (SEAFAC) in the Western Behm Canal, Alaska. Figure 2, Figure 3, and Figure 4 illustrates the offshore portion of the NWTRC, the inshore portion of the NWTRC (Keyport Range Complex), and SEAFAC in greater detail.

2.5.1 Offshore Area of the Action Area

The offshore component of the Action Area includes air, surface, and subsurface operating areas extending generally west from the coastline of Washington, Oregon, and Northern California for a distance of approximately 250 nm into international waters. The eastern boundary of the Offshore Area lays 12 nm off the coastline for most of the Action Area, including southern Washington, Oregon, and Northern California. The Offshore Area includes the ocean all the way to the coastline only along the Washington coast beneath the airspace of W-237 and the Olympic MOA and the Washington coastline north of the Olympic MOA. The Offshore Area also includes approximately 1 mile of beach area located at Pacific Beach.

Commander Submarine Force, U.S. Pacific Fleet Pearl Harbor uses the water space of the Offshore Area as transit lanes for U.S. submarines. The sea space is ample for all levels of Navy training, and its location is ideal for ships, submarines, and aircraft based in the Pacific Northwest. The size of the area and its extension south off the coast of Northern California provides valuable training and testing space for ships and submarines transiting between the Pacific Northwest and Southern California.

Within the boundaries of the Offshore Area lies the Quinault Range Site (Figure 2), a defined area of sea space where training and testing is conducted. The Quinault Range Site coincides with the boundaries of W-237A and also includes a surf zone component. The surf zone component extends north to south for 5 nm along the eastern boundary of W-237A, approximately 3 nm to shore along the mean lower low water line, and encompasses 1 mile (mi.) (1.6 kilometers [km]) of shoreline at Pacific Beach, Washington. Surf-zone activities would be conducted from an area on the shore and seaward.

2.5.1.1 ***Special Use Airspace***

The Special Use Airspace (SUA) in the Offshore Area (Figure 1) is comprised of Warning Area 237 (W-237), which extends westward off the coast of northern Washington State and is divided into nine sub-areas (A–H and J). The eastern boundary of W-237 lays 3 nm off the coast of Washington. The floor of W-237 extends to the ocean surface, and the ceiling of the airspace varies between 27,000 feet (ft.) mean sea level (MSL) (8,200 meters [m]) in areas E, H, and J; 50,000 ft. (15,200 m) (MSL) in areas A and B; and unlimited in areas C, D, F, and G.

The Olympic Military Operations Areas (MOAs) overlay both land (the Olympic Peninsula) and sea (extending to 3 nm off the coast of Washington into the Pacific Ocean). The MOAs lower limit is 6,000 ft. (1,800 m) (MSL) above mean sea level but not below 1,200 ft. above ground level, and the upper limit is up to but not including 18,000 ft. (5,500 m) (MSL), with a total area coverage of 1,614 square nautical miles (nm2). Above the Olympic MOA is the Olympic ATCAA, which has a floor coinciding with the Olympic MOAs ceiling. The ATCAA has an upper limit of 35,000 ft. (10,700 m) (MSL).

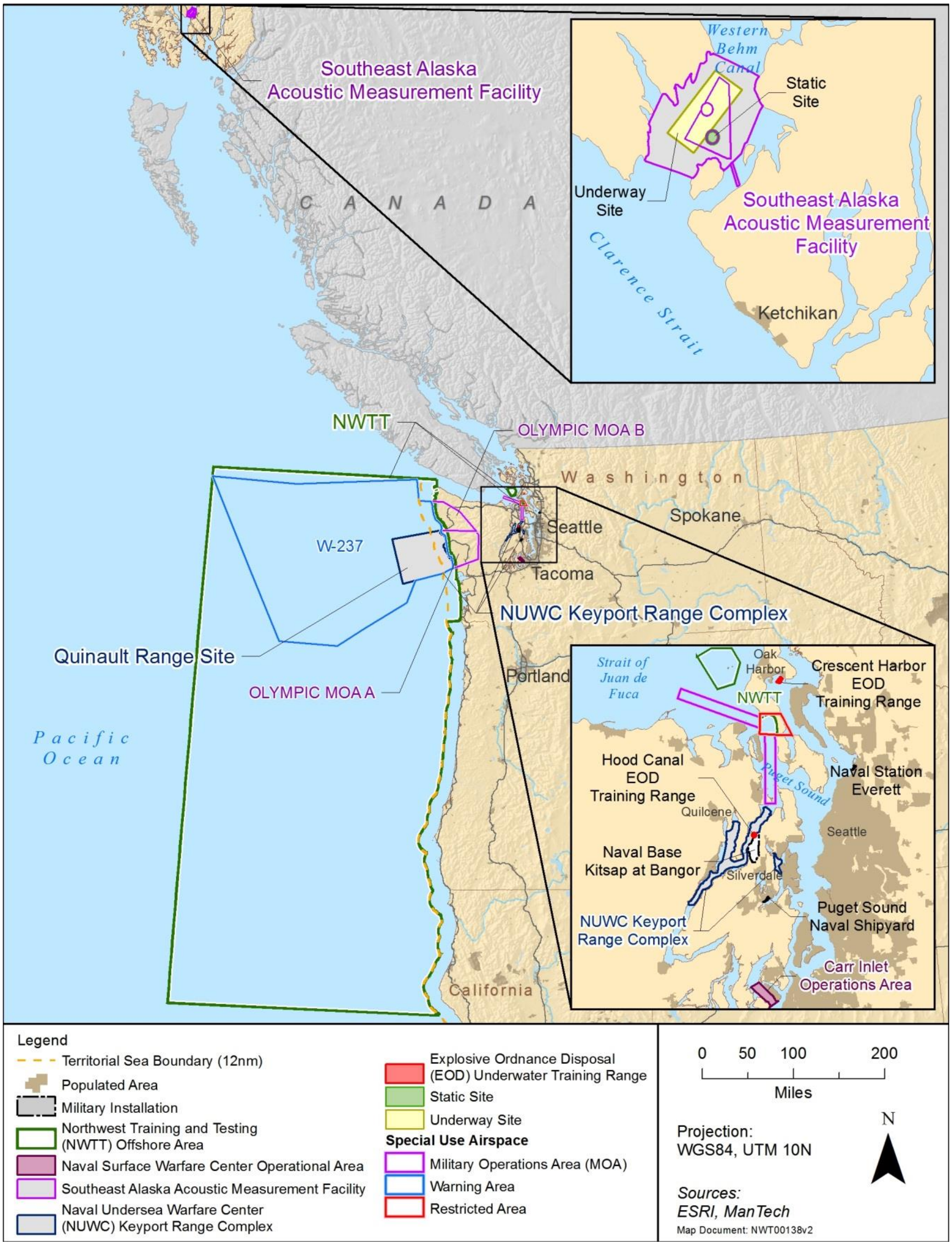

Figure 1. Northwest Training and Testing Action Area

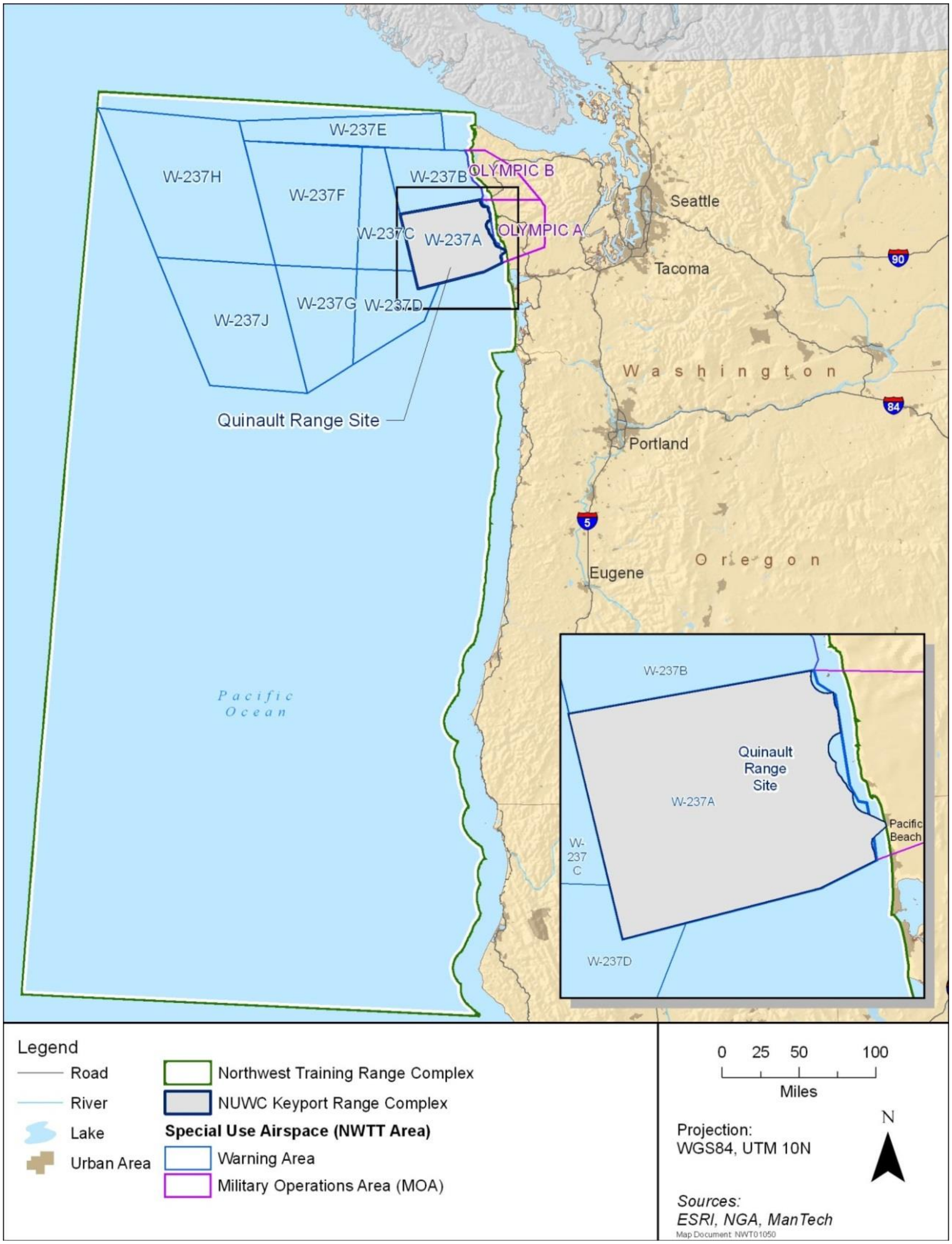

Figure 2. Action Area – Northwest Training Range Complex – Offshore Portion

2.5.1.2 *Sea and Undersea Space*

The Offshore Area includes sea and undersea space approximately 510 nm in length from the northern boundary at the mouth of the Strait of Juan de Fuca to the southern boundary at 40 degrees (°) north (N) latitude, and 250 nm in length from the coastline to the western boundary at 130° west (W) longitude. The southern boundary of 40° N latitude corresponds to the northern boundary of Mendocino County in Northern California. Total surface area of the Offshore Area is approximately 121,000 nm_2. The Offshore Area extends to the shoreline only along the northern portion of the Washington coast, and is 12 nm from the coastline from the southern boundary of the Olympic MOA to Northern California, including the entire coastline of Oregon and Northern California.

2.5.2 Inland Waters of the Action Area

The Inland Waters component of the Action Area includes air, sea, and undersea space inland of the Pacific coastline, from buoy "J" at 48° 29.6‘ N, 125° W, eastward to include all waters of the Strait of Juan de Fuca and the Puget Sound. None of this area extends into Oregon, California, or Alaska. Within the Inland Waters are specific geographic components in which training and testing occur. The Inland Waters and its component areas are described below and depicted in Figure 3.

2.5.2.1 *Air Space*

Restricted Area 6701 (R-6701, Admiralty Bay) is a Restricted Area over Admiralty Bay, Washington, with a lower limit at the ocean surface and an upper limit of 5,000 ft. This airspace covers a total area of 56 nm^2. Chinook A and B MOAs are 56 nm^2 of airspace south and west of Admiralty Bay. The Chinook MOAs extend from 300 ft. to 5,000 ft. above the ocean surface.

2.5.2.2 *Sea and Undersea Space – Explosive Ordnance Disposal Ranges*

Two active EOD ranges, also used for swimmer training in Mine Countermeasures, are located in the Inland Waters at the following locations, as depicted by Figure 3:

- NAVBASE Kitsap Bangor – Hood Canal EOD Range
- Naval Air Station (NAS) Whidbey Island (NASWI) – Crescent Harbor EOD Range

2.5.2.3 *Sea and Undersea Space - Surface and Subsurface Testing Sites*

There are three geographically distinct range sites in the Inland Waters where the Navy conducts surface and subsurface testing and some limited training. The Keyport Range Site is located in Kitsap County and includes portions of Liberty Bay and Port Orchard Reach (also known as Port

Orchard Narrows). The Dabob Bay Range Complex (DBRC) Site is located in Hood Canal, in Jefferson, Kitsap, and Mason counties. The Carr Inlet OPAREA is located in southern Puget Sound.

The Keyport Range Site is located adjacent to NAVBASE Kitsap, providing approximately 3.2 nm^2 for underwater testing, including in-shore shallow water sites and a shallow lagoon to support integrated undersea warfare systems and vehicle maintenance and engineering activities. Water depth at the Keyport Range Site is less than 100 ft. (30 m).

Underwater tracking of test activities can be accomplished by using temporary or portable range equipment. The Navy has conducted underwater testing at the Keyport Range Site since 1914.

The DBRC Site includes Dabob Bay and Hood Canal from 1 mi. (1.6 km) south of the Hood Canal Bridge to the Hamma Hamma River, a total area of approximately 45.7 nm^2. The Navy has conducted underwater testing at the DBRC Site since 1956, beginning with a control center at Whitney Point. The control center was subsequently moved to Zelatched Point.

Dabob Bay is a deep-water area in Jefferson County approximately 14.5 nm^2 in size, which contains an underwater acoustic tracking range. The acoustic tracking space within the range is approximately 7.3 nm by 1.3 nm (9 nm^2) with a maximum depth of 600 ft. (182.9 m). The Dabob Bay tracking range, the only component of the DBRC Site with extensive acoustic monitoring instrumentation installed on the seafloor, provides for object tracking, communications, passive sensing, and target simulation. Many activities conducted within Dabob Bay are supported by land-based facilities at Zelatched Point.

The Carr Inlet OPAREA is a quiet, deep-water inland range approximately 12 nm^2 in size. It is located in an arm of water between Key Peninsula and Gig Harbor Peninsula. Its southern end is connected to the southern basin of Puget Sound. Northward, Carr Inlet OPAREA separates McNeil Island and Fox Island as well as the Key and Gig Harbor peninsulas. The acoustic tracking space within the range is approximately 6 nm by 2 nm with a maximum depth of 545 ft. (166 m). The Navy previously performed underwater acoustic testing at Carr Inlet from the 1950s through 2009, at which time activities were relocated to NAVBASE Kitsap Bangor. While no permanently installed structures are present in the Carr Inlet OPAREA, the waterway remains a Naval Restricted Area (33 C.F.R. § 334.1250).

2.5.2.4 ***Sea and Undersea Space - Pierside Testing Facilities***

Most of the NWTT training and testing activities occur in established training and testing ranges; however, the Navy conducts some testing at or near Navy piers. Most pierside testing is sonar maintenance or other type of testing conducted while ships are in port for maintenance or system re-fitting. The Navy piers within the Action Area are all within Puget Sound and include

NAVBASE Kitsap Bremerton in Sinclair Inlet, NAVBASE Kitsap Bangor Waterfront in Hood Canal, and NAVSTA Everett.

2.5.2.5 *Sea and Undersea Space - Navy Surface Operations Areas*

In addition to the areas described above, there are two surface and subsurface operations areas used for Navy training and testing within the Inland Waters. Navy 3 OPAREA is a surface and subsurface area off the west coast of northern Whidbey Island. Navy 7 OPAREA is the surface and subsurface area that lies beneath R-6701. This area covers a total area of 56 nm^2.

2.5.3 Southeast Alaska Acoustic Measurement Facility – Western Behm Canal, Alaska

The Western Behm Canal is located in Southeast Alaska, near the city of Ketchikan, Alaska. Southeast Alaska Acoustic Measurement Facility (SEAFAC) is located in the Western Behm Canal and covers an area of 48 nm^2 (Figure 4). The Navy has been conducting testing activities at SEAFAC since 1992. The facility replaced the Santa Cruz Acoustic Range Facility in Southern California and is now the location for some acoustic testing previously conducted at the NSWC Carr Inlet Acoustic Range in Washington State.

SEAFAC is comprised of land-based facilities and in-water assets. The land-based facilities located within 5.5 acres on Back Island are not a part of the Action Area, and Navy activities occurring in this location will be or have been addressed under separate regulatory documentation.

The in-water assets include two sites: the underway site and the static site. Bottom-moored acoustic measurement arrays are located in the middle of the site. These instrumented arrays are established for measuring vessel signatures when a vessel is underway (underway site) and is at rest and moored (static site). Active acoustic sources are used for communications, range calibration, and to provide position information for units operating submerged on the range.

The SEAFAC at-sea areas are:

- Restricted Areas 1 through 5. The five restricted areas are located within Western Behm Canal. The main purposes of the restricted areas are to provide for vessel and public safety, lessen acoustic encroachment from non-participating vessels, and prohibit certain activities that could damage SEAFAC's sensitive in-water acoustic instruments and associated cables. Area 5 encompasses the entire SEAFAC operations area.

- Underway Measurement Site. The underway measurement site is in the center of Western Behm Canal and is 5,000 yards (yd.) (4,572 m) wide and 12,000 yd. (10,973 m) long. The acoustic arrays are located at the center of this area (Area 1).

- Static Site. The static site is approximately 2 nm northwest of Back Island. During testing, a vessel is tethered between two surface barges. In most scenarios, the vessel submerges to conduct acoustic measurements. The static site is located at the center of Area 2.

- Area 3 and Area 4. These restricted areas provide protection to underwater cables and the bottom-mounted equipment they encompass.

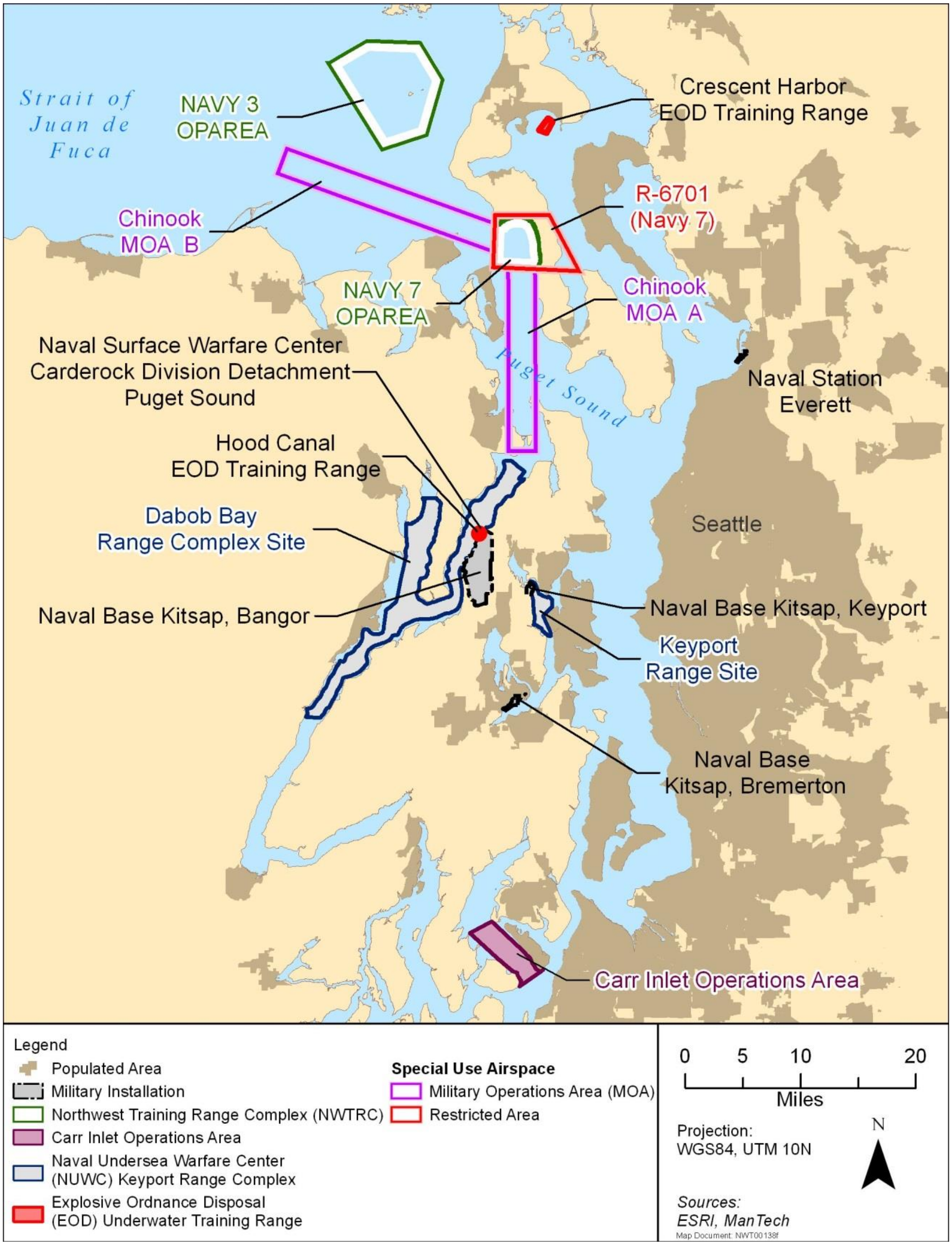

Figure 3. Action Area – Northwest Training Range Complex – Inshore Portion

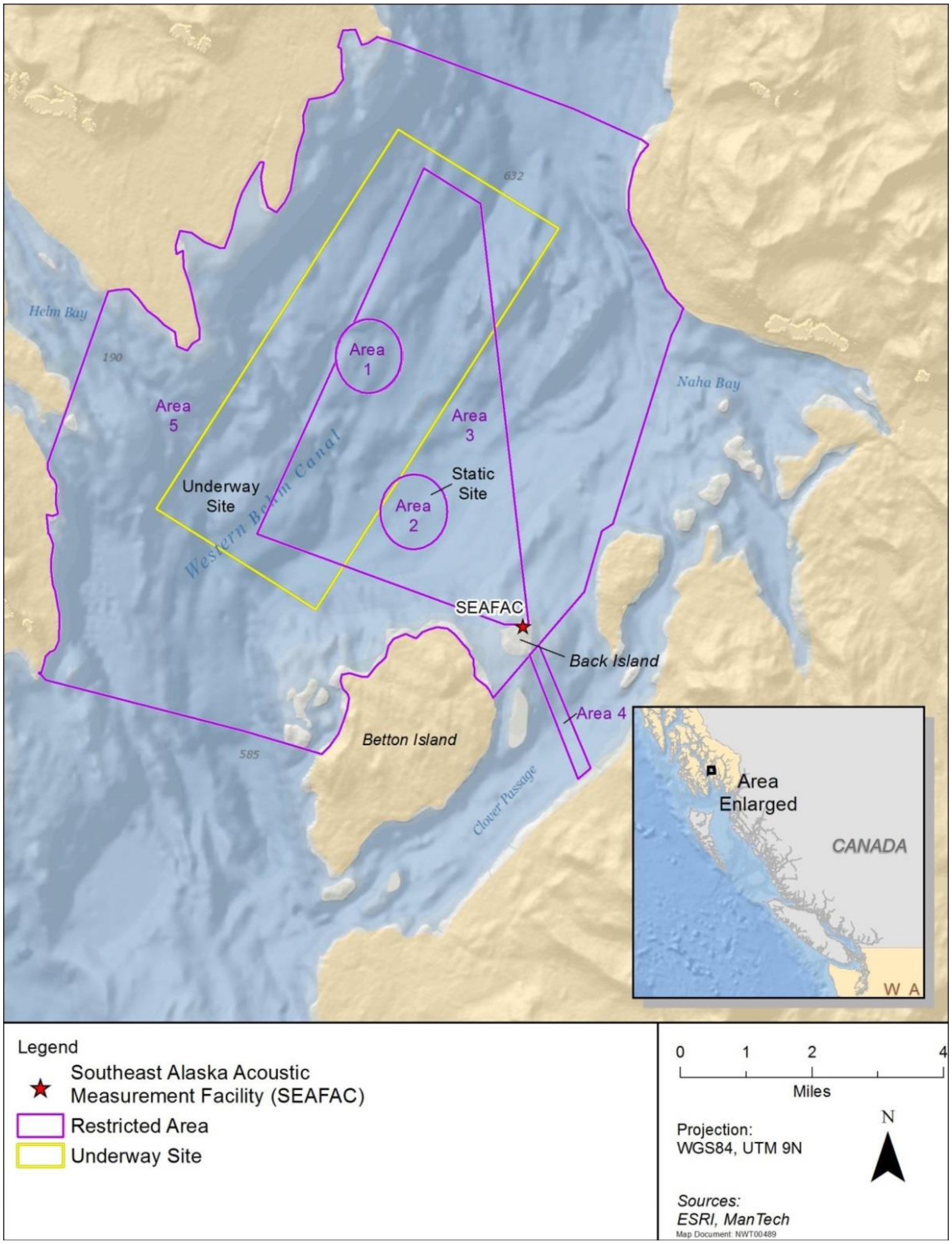

Figure 4. Action Area – Western Behm Canal, Alaska and the Southeast Alaska Acoustic Measurement Facility

2.6 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 actions outside the scope of Navy training and testing 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. This opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.[1]

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.

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[1] Memorandum from William T. Hogarth to Regional Administrators, Office of Protected Resources, NMFS (Application of the "Destruction or Adverse Modification" Standard Under Section 7(a)(2) of the Endangered Species Act) (November 7, 2005).

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 EIS/OEIS on training and testing activities that are proposed in the Action Area. 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 and testing 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 and testing.

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 Consideration of new literature and pending Acoustic Guidance on the Effects of Anthropogenic Sound on Marine Mammals

NOAA is currently in the process of developing Acoustic Guidance on thresholds for onset of auditory impacts from exposure to sound, which will be used to support assessments of the effects of anthropogenic sound on marine mammals. To develop this Guidance, NOAA is compiling, interpreting, and synthesizing the best information currently available on the effects of anthropogenic sound on marine mammals, and is committed to finalizing the Guidance through a systematic, transparent process that involves internal review, external peer review, and public comment.

In December 2013, NOAA released for public comment a "Draft Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammals: Acoustic Threshold Levels for Onset of Permanent and Temporary Threshold Shifts" (National Oceanic and Atmospheric Administration 2013c; 78 Federal Register [FR] 78822). The Draft Guidance was generally consistent with the Navy's PTS/TTS criteria used in the NWTT EIS/OEIS and detailed within Finneran and Jenkins (2012). Prior to the finalization of this guidance by NOAA, the Navy suggested revisions to the criteria (e.g., auditory weighting functions and PTS/TTS thresholds) based on a number of newly available studies. In January 2015, the Navy submitted a draft proposal (Finneran 2015) to NOAA staff for their consideration.

Finneran (2015) proposed new weighting functions and thresholds for predicting PTS/TTS in marine mammals. The methodologies presented within this paper build upon the methodologies used to develop the criteria used within the Navy's EIS/OEIS (Finneran and Jenkins 2012) and incorporate relevant auditory research made available since 2012. While Finneran and Jenkins (2012) presented a conservative approach to development of auditory weighting functions where data was limited, Finneran (2015) synthesizes a wide range of auditory data, including newly available studies, to predict refined auditory weighting functions and corresponding TTS thresholds across the complete hearing ranges of functional hearing groups. Finneran (2015) also developed updated threshold shift growth functions to facilitate the development of new PTS thresholds.

During the development process of NOAA's Draft Guidance, NOAA chose to incorporate Finneran (2015) into its Draft Guidance prior to its finalization. As a result, the Navy's proposal (Finneran 2015) was submitted for peer review by external subject matter experts, in accordance with the process previously conducted for NOAA's Draft Guidance. Peer review comments were received by NOAA in April 2015. NOAA subsequently developed a Peer Review Report, which was published on its website on 31 July 2015. The published report documents the Navy's proposal (Finneran 2015) that underwent peer review, the peer-review comments, and NOAA's responses to those comments. NOAA then incorporated this information into revised Draft Guidance which was recently published in the Federal Register for public review and comment (80 FR 45642). The auditory weighting functions and PTS/TTS thresholds provided in that revised Draft Guidance will not be adopted by NOAA or applied to applicants until the revised Draft Guidance is issued as final guidance. At the time of this consultation, a second public comment period has closed, though final guidance has not been issued. Therefore, the Navy has not adopted these proposed criteria in its Final EIS/OEIS. However, the underlying science contained within Finneran (2015) has been addressed qualitatively within the applicable sections of the Final EIS/OEIS and this Opinion.

If the proposed criteria in Finneran (2015) were adopted by NOAA, incorporated into its Final Guidance, and applied to the Navy in the future, predicted numbers of PTS/TTS would change for most functional hearing groups. However, because Finneran (2015) relies on much of the same data as the auditory criteria presented in the Navy's NWTT EIS/OEIS, these changes would not be substantial, and in most cases would result in a reduction in the predicted impacts. Predicted PTS/TTS would be reduced over much to all of their hearing range for low-frequency cetaceans and phocids. Predicted PTS/TTS for mid-frequency and high-frequency cetaceans would be reduced for sources with frequencies below about 3.5 kHz and remain relatively unchanged for sounds above this frequency. Predicted auditory effects on otariids would increase for frequencies between about 1 kHz and 20 kHz and decrease for frequencies above and below these points, although otariids remain the marine mammals with the least sensitivity to potential PTS/TTS. Overall, predicted auditory effects within this Opinion would not change significantly.

In summary, NOAA's continued evaluation of all available science for the Acoustic Guidance could result in changes to the acoustic criteria used to model the Navy's activities for this consultation, and, consequently, the enumerations of "take" estimates. However, consideration of the revised Draft Guidance and information contained in Finneran (2015) does not alter our assessment of the likely responses of affected ESA-listed species to acoustic sources employed by Navy in the NWTT Action Area, or the likely fitness consequences of those responses.

Since the Navy's Phase 2 modeling (the NAEMO model considered in this Opinion), there have been several studies on marine mammal TTS (e.g., (Finneran et al. 2005a; Finneran et al. 2010; Finneran and Schlundt 2013; Kastelein et al. 2012a; Kastelein et al. 2012d; Kastelein et al. 2014a; Kastelein et al. 2014d; Popov et al. 2013a; Popov et al. 2011)) (Table 9).

3.1.1.1 ***Mid-frequency Cetaceans***

The results from Finneran and Schlundt (2013) demonstrate there are frequency-specific differences by species in TTS onset and growth, and suggest increased susceptibility to auditory fatigue for frequencies between approximately 10 kHz and 30 kHz. This study supports the importance of using auditory weighting functions (i.e., functional hearing groups are more susceptible to noise-induced hearing loss at certain frequencies), but is not expected to change any of the thresholds to onset TTS between those the Navy used in this consultation and the pending Acoustic Guidance (A. Scholik-Schlomer, personal communication, March 26, 2015).

Popov et al. (2013a) and Popov et al. (2014) examined the impacts of half-octave band noise on hearing in two belugas. In their 2013 study, levels beyond TTS onset were induced (i.e., threshold shifts ranged from 7.5 to 62.5 dB) with recovery occurring within 24 h. In their 2014 study, TTS onset was identical to that of a beluga from another study (i.e., NOC from Schlundt et al. (2000a); i.e., these studies are not expected to result in any changes to the thresholds).

3.1.1.2 ***High-frequency Cetaceans***

All the recent TTS studies on harbor porpoises were completed by Kastelein using the same individual exposed to either octave band noise (Kastelein et al. 2012a), tones (Kastelein et al.

2013b; Kastelein et al. 2014c) , or 1 to 2 kHz sonar (Kastelein et al. 2014b). The results from Kastelein et al. (2014b) are expected to necessitate a change to TTS onset thresholds for non-impulsive sounds for HF cetaceans (i.e., changed from 160 dB SEL_{cum} in Draft NOAA Guidance to 152 dB SEL_{cum}). This potential change to the TTS onset threshold could impact the enumerations of "take" estimates. However, any changes and subsequent modeling by the Navy would not alter our assessment of the likely responses of affected ESA-listed species to acoustic sources employed by Navy in the NWTT Action Area, or the likely fitness consequences of those responses. Other points of note from these Kastelein et al. studies (all of which have been demonstrated from previous marine mammal studies), include showing 1) increasing exposure duration was more effective in elevating TTS compared to increasing level of exposure, 2) lower frequency sounds (i.e., 1.5 kHz) are unlikely to affect ability for this species to echolocate (since echolocation occurs at much higher frequency: 125 kHz), and 3) interpulse interval plays an important role in determining TTS onset (lower level of TTS onset for continuous exposure compared to intermittent exposure; longer interpulse interval results in less incidences of TTS and faster recovery times), even with exposures with identical SEL_{cum}.

Kastelein et al. (2015) exposed a harbor porpoise to 2,760 pile driving strikes resulting in an unweighted level of 180 dB SEL_{cum}, but threshold shifts of less than 6 dB were induced (i.e., mean threshold shift of 2.3 to 3.6 dB). Thus, these results are not expected to result in any changes to the thresholds the Navy used in this consultation to those proposed in the pending Acoustic Guidance.

3.1.1.3 *Phocid Pinnipeds*

Two more recent studies for phocid pinnipeds (harbor seal) were completed by Kastelein et al. (Kastelein et al. 2013a; Kastelein et al. 2012b). Neither of these studies are expected to result in changes in the TTS onset threshold from those the Navy used in this consultation to those proposed in the pending Acoustic Guidance. Of note, the results for Kastelein et al. (2012b) were excluded from consideration from NOAA's analysis because the exposure durations were greater than 1 h (i.e., durations were 120 and 240 minutes of continuous exposure) and considered unlikely to be encountered by wild individuals. The exposure in Kastelein et al. (2013a) resulted in a 44 dB threshold shift that took 4 days to recover (199 dB SEL_{cum}; note that this exposure level is higher than what is being proposed in the Guidance for PTS onset, which is 197 dB SEL_{cum}, and that the exposure in Kastelein did result in recovery).

Table 9. Available underwater marine mammal threshold shift peer reviewed studies (since 2011).

References in Chronologic Order^{+}	Sound Source (Sound Source Category)	Sound-Exposed Species (number of individuals^)
(Kastelein et al. 2012b)	Octave-band noise (non-impulsive)	Harbor seal (2)
(Kastelein et al. 2012c)	Octave-band noise (non-impulsive)	Harbor porpoise (1)

References in Chronologic Order+	Sound Source (Sound Source Category)	Sound-Exposed Species (number of individuals^)
(Finneran and Schlundt 2013)	Tones (non-impulsive)	Bottlenose dolphin (2)
(Popov et al. 2013b)	Half-octave band noise (non-impulsive)	Beluga (2)
(Kastelein et al. 2013a)	Octave-band noise (non-impulsive)	Harbor seal (1)
(Kastelein et al. 2013b)	Tone (non-impulsive)	Harbor porpoise (1)
(Popov et al. 2014)	Half-octave band noise (non-impulsive)	Beluga (2)
(Kastelein et al. 2014b)	1-2 kHz sonar (non-impulsive)	Harbor porpoise (1)
(Kastelein et al. 2014c)	6.5 kHz tone (non-impulsive)	Harbor porpoise (1)
(Kastelein et al. 2015)	Impact pile driving (impulsive)	Harbor porpoise (1)
Kastelein et al. unpublished+	Impact pile driving (impulsive)	Harbor seal (2)
Finneran et al. in press+	Multiple airgun shots (impulsive)	Bottlenose dolphin (3)

+Peer reviewed studies available and evaluated as of 4 April 2015. Note: Two papers expected to publish in the near future. However, for all these studies, a TS could not be induced. Thus, neither study affects the acoustic threshold levels and are instead included for completeness.

^Note: Some individuals have been used in multiple studies.

3.1.2 The Navy's Exposure Estimates

To estimate exposure of marine mammal and sea turtle to acoustic sounds, the Navy uses acoustic modeling and marine mammal and sea turtle 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.

3.1.2.1 *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 and testing 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 and testing 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 detailed discussion of the NAEMO is contained in chapter 3.4.3.1.6 of the EIS/OEIS, which is incorporated by reference. A full description of NAEMO can be accessed in the Naval Undersea Warfare Center Newport Technical Report 12,071a, August 23, 2013 (updated from March 12, 2012).

3.1.2.1.1 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 (REFMS).

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, 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.

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 and testing activities. Some scenarios are evaluated by platform and single locations, while others are evaluated in multiple locations within a single range complex or testing 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 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.

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 NWTT EIS/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 12,071a, August 23, 2013 (updated from March 12, 2012). 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 2012).

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 testing 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 testing 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 Study.

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 NWTT Study Area, sound may propagate beyond the boundary of the Study Area. Any exposures occurring outside the boundary of the Study Area are counted as if they occurred within the Study Area boundary or within the Action Area for this Opinion.

3.1.2.1.2 Model Assumptions

There are limitations to the data used in the NAEMO, and the results must be interpreted within these 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 assumed to 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 and testing 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.

3.1.3 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. (2014) "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. (2014) 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 10. Sound Exposure Criteria for Fishes Exposed to Navy 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.

3.1.3.1 *Low-Frequency Navy Sonar*

3.1.3.1.1 Mortality, Mortal Injury, and Recoverable Injury All Fish = > 218 dB SEL$_{cum}$ (cells A1, A2, A3, B1, B2 & B3 in Table 10 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 NWTT proposed activities. 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.

3.1.3.1.2 Temporary Threshold Shift, Fish-no SB = > 218 dB SEL$_{cum}$ (cell C1 in Table 10 above)

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

3.1.3.1.3 Temporary Threshold Shift, Fish w/ SB = 210 dB SELcum (cells C2 & C3 in Table 10 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 (2014) 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.

$$193\text{dB SPLrms} + 10\log(324 \text{ sec}) = 218 \text{ dB SEL}_{cum} \text{ (Popper et al. 2007)}$$

3.1.3.1.4 Masking, Fish w/out SB and Fish w/ SB not involved in hearing = (N)Low, (I)Low, (F)Low (cells D1 &, D2 in Table 10 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. 2014). The narrow bandwidth of most sonar would result in only a limited range of frequencies being masked (Popper et al. 2014). Furthermore most sonars are intermittent (i.e., low duty cycle) which further lowers the probability of any masking effects.

3.1.3.1.5 Masking, Fish w/ SB involved in hearing = (N)Mod, (I)Low, (F)Low (cell D3 in Table 10 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. 2014); however, the narrow bandwidth of most sonar would result in only a limited range of frequencies being masked (Popper et al. 2014). Furthermore most sonars are intermittent (i.e., low duty cycle) which further lowers the probability of any masking effects.

3.1.3.1.6 Behavior, Fish no SB and Fish w/ SB not involved in hearing = (N)Low, (I)Low, (F)Low (cells E1 & E2 in Table 10 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. 2014).

3.1.3.1.7 Behavior: Fish w/ SB involved in hearing = > 197 dB SPLrms (cell E3 in Table 10 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. (2014), >197 dB SPLrms (Doksaeter et al. 2009; Doksaeter et al. 2012).

3.1.3.2 *Mid-Frequency Navy Sonar*

3.1.3.2.1 Mortality, Mortal Injury & Recoverable Injury: >> 221 dB SEL$_{cum}$ (cells A4, A5, A6, B4, B5, & B6 in Table 10 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.

3.1.3.2.2 TTS: Fish-no SB and Fish w/SB not involved in hearing = NA (cells C4 & C5 in Table 10 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.

3.1.3.2.3 TTS: Fish w/ SB used in hearing = 220 dB SEL_{cum} (cell C6 in Table 10 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. (2014) 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).

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

3.1.3.2.4 Masking: NA (cells D4, D5, & D6 in Table 10 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. 2014). 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).

3.1.3.2.5 Behavior: Fish no SB and Fish w/ SB not involved in hearing = NA (cells E4 & E5 in Table 10 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.

3.1.3.2.6 Behavior: Fish w/ SB involved in hearing = 200 dB SPL_{rms} (cell E6 in Table 10

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).

3.1.4 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, eulachon, and rock fish. The most appropriate thresholds to assess are the onset of physical injury (including auditory injury from permanent threshold shift) and onset of mortality from impulsive stressors. These thresholds must be based on the best scientific and commercial data[2] pursuant to section 7 of the ESA.

3.1.4.1 *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).

3.1.4.1.1 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 11) 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.

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[2] 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)]

Table 11. 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.

3.1.4.1.2 Popper et al. 2014 Thresholds for Explosives

The authors of Popper et al. (2014) 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 12). Thus, Popper et al. (2014) 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 12. Popper et al. 2014 Explosive Thresholds for Mortality/Mortal Injury (highlighted).

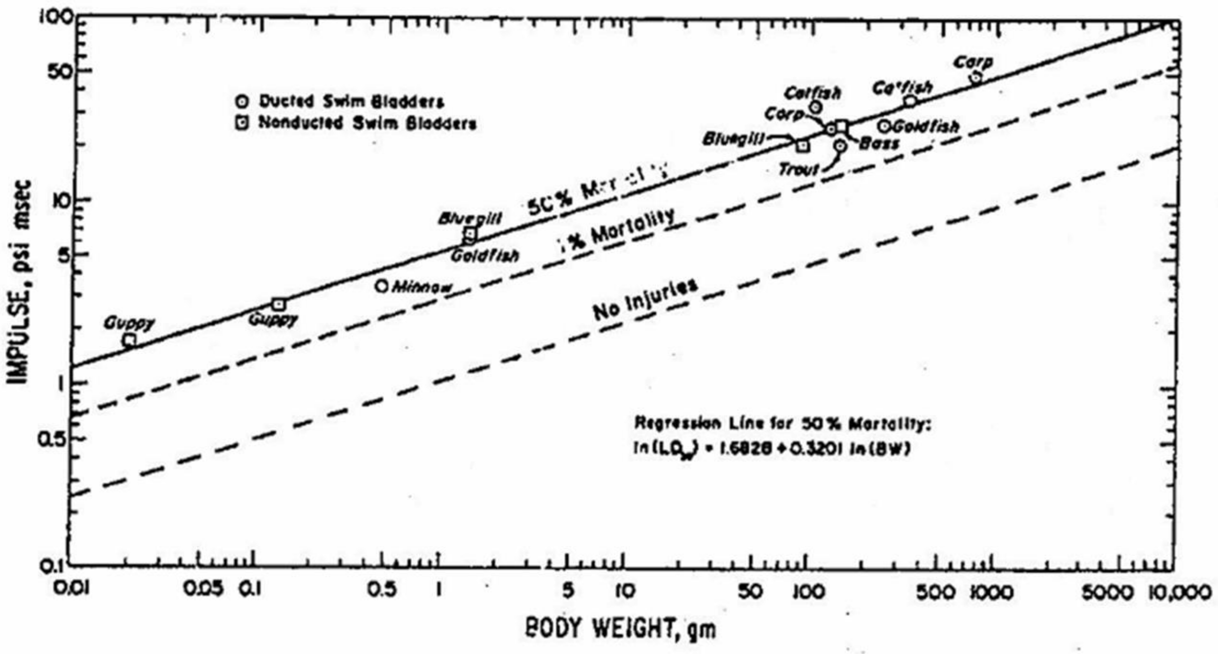
Guidelines for explosions. Levels other than for eggs and larvae from Hubbs and Rechnitzer (1952); levels for eggs and larvae from Wright and Hopky (1998). Guidelines are not provided for masking since the animals are not exposed to more than a few explosive events, and masking would not last beyond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

3.1.4.1.3 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 (psi-ms) (Figure 5). 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 5. Results from Yelverton et al. (1975) (caption from Hastings and Popper (2005)).

3.1.4.2 *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.

The intent of the Popper et al. (2014) 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 6), they lend additional support to this model. Thus, upon further evaluation, 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).

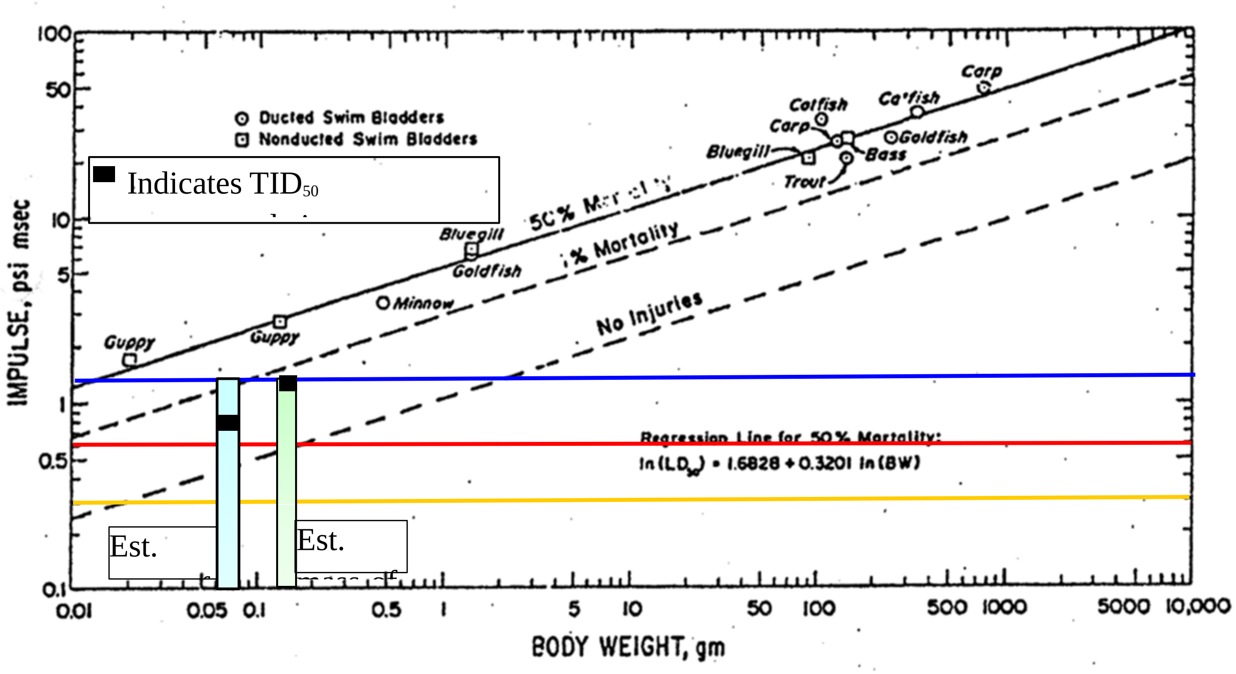


Figure 6. Average impulse values from Govoni et al. (2003, 2007; vertical aqua bars with black squares indicating total injury dose (TID_{50}) recommended by Govoni et al. 2007) compared with results of Yelverton et al. (1975) from Hasting 2007. Distance from the explosive source is indicated by the color of the line: blue – 3.6 m; red – 7.5 m; and yellow – 17 m.

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. (2014) 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 13 are appropriate to assess effects to salmonids and rockfish[3] resulting from impulsive stressors associated with Navy NWTT activities. These thresholds may also be appropriate for other actions involving explosions in water.

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[3] As discussed in several sections of this Opinion, fish without swim bladders are less susceptible to acoustic stressors, including underwater sound from explosions. Popper et al. (2014) suggested that little or no damage occurs to fishes without a swim bladder except at very short ranges from an in-water explosive event. The range from an explosive event over which damage may occur to fish without swim bladders is on the order of 100 times less than that for a fish with a swim bladder Goertner (1978), as cited in Popper et al. (2014). Therefore, the range to effects values for eulachon were calculated using the same methodology as was used for the other fish species considered in this Opinion, and then each range to effect value was divided by 100 to account for the fact that eulachon do not possess a swim bladder.

Table 13. 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)**		

3.1.5 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.

Finneran and Schlundt (2010)

Finneran and Schlundt (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.

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 testing 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 ~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).

3.1.6 Post Processing. Avoidance Behavior and Mitigation Measures as Applied to Sonar and Active Acoustic Sources

As described in the NWTT DEIS/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 and testing activities, regardless of how many vessels or low-flying aircraft (i.e., helicopters) are involved. The outputs of the model, therefore, present an unrealistically high estimate of acoustic impacts in close proximity to certain Navy training and testing activities.

Additionally, the modeling currently does not account for implementation of mitigation designed to avoid or reduce marine mammal and sea turtle exposures to explosives and high intensity sound, nor does it account for standard operating procedures (procedures designed for the safety of personnel and equipment) implemented to ensure safety and mission success, but which may have incidental environmental benefits. That is, the modeling assumes that any mitigations measures, such as sonar power-down or delay of a detonation, would not be implemented even if an animal could be sighted within the mitigation zone. 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. The outputs of the model (without mitigation), therefore, present an unrealistically high estimate of acoustic impacts within the mitigation zones of certain Navy training and testing activities.

In order to provide a holistic quantitative assessment of acoustic impacts, the post-model analysis quantitatively assessed the effect of animal avoidance behavior and implementation of mitigation, 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
- Ability to observe the mitigation zone around different platforms during different activities

The steps of the post-model analysis are briefly summarized in Table 14 and presented in the order they are expected to occur during an actual training or testing activity, which is also the order in which they were mathematically considered in the post-model analysis. When feasible for a given activity, mitigation begins prior to the actual production of underwater sound (e.g., 10 to 30 minutes, dependent upon platform, prior to most sonar and explosive activities); therefore, mitigation effectiveness is applied in the post-model analysis before animal avoidance is quantified.

Table 14. 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	E-2. Can Lookouts observe the activity-specific

Is the Sound Source Sonar/Other Active Acoustic Source or Explosives?	
Sonar and Other Active Acoustic Sources	Explosives
mitigation zone (see Chapter 5) up to and during the sound-producing activity?	mitigation zone (see Chapter 5) up to and during the sound-producing activity?
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.

3.1.6.1 ***Mitigation***

The Navy Acoustic Effects Model estimates acoustic effects without taking into account any shutdown or delay of the activity when marine mammals are present and detectable within the mitigation zone; therefore, the model overestimates impacts to marine mammals within mitigation zones. The post-model analysis considers and quantifies the potential for mitigation to reduce the likelihood or risk of PTS (due to sonar and other active acoustic sources) and injuries and mortalities (due to explosives). The NAEMO estimated no exposures of ESA-listed marine mammals or sea turtles that would be expected to result in PTS, injury, or mortality; therefore, there were no post-processing changes in estimated take numbers for ESA-listed species.

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.

3.1.6.2 *Avoidance*

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 for most cases. Because the Navy Acoustic Effects Model does not consider horizontal movement of animats, including avoidance of high-intensity sound exposures, it over-estimates the number of marine mammals and sea turtles that would be exposed to sound sources that could cause injury. 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. Therefore, the potential for avoidance is considered in the Navy's post-model analysis. The NAEMO estimated no exposures of ESA-listed marine mammals or sea turtles that would be expected to result in PTS, injury, or mortality; therefore, there were no post-processing changes in estimated take numbers for ESA-listed species.

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 and testing 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[4]") of the entire area contained in the designation.

4 STATUS OF LISTED RESOURCES

This section identifies the ESA-listed species that occur within the Action Area that may be affected by Navy NWTT 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 Action are listed in Table 15, along with their ESA listing status.

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

Table 15. Species listed under the Federal Endangered Species Act (ESA) under NMFS jurisdiction that may occur in the NWTT Action Area.

Species	ESA Status	Critical Habitat	Recovery Plan	Trend
Marine Mammals – Cetaceans				
Blue Whale (*Balaenoptera musculus*)	E - 35 FR 18319	-- --	07/1998	Stable
Fin Whale (*Balaenoptera physalus*)	E - 35 FR 18319	-- --	75 FR 47538	unknown
Humpback Whale (*Megaptera novaeangliae*)*	E - 35 FR 18319	-- --	55 FR 29646	↑
North Pacific Right Whale (*Eubalaena japonica*)	E - 73 FR 12024	73 FR 19000	-- --	unknown
Sei Whale (*Balaenoptera borealis*)	E - 35 FR 18319	-- --	12/2011	unknown
Sperm Whale (*Physeter macrocephalus*)	E - 35 FR 18619	-- --	75 FR 81584	unknown
Southern resident killer whale *(Orcinus orca*)	E - 70 FR 69903	71 FR 69054	73 FR 4176	stable
Marine Mammals – Pinnipeds				
Guadalupe Fur Seal (*Arctocephalus townsendi*)	T - 50 FR 51252	-- --	-- --	unknown
Sea Turtles				
Green Turtle (*Chelonia mydas*)	E - 43 FR 32800	63 FR 46693	63 FR 28359	varies
Loggerhead Turtle (*Caretta caretta*)	E - 76 FR 58868	-- --	63 FR 28359	↓
Olive Ridley Turtle (*Lepidochelys olivacea*)	T - 43 FR 32800	-- --	63 FR 28359	Stable
Leatherback Turtle (*Dermochelys coriacea*)	E - 61 FR 17	77 FR 4170	63 FR 28359	↓
Fish – Rockfish				
Bocaccio *(Sebastes paucispinus*)				
Puget Sound/Georgia Basin DPS	E - 75 FR 22276	79 FR 68041	-- --	↓
Canary rockfish (*Sebastes pinniger*)				
Puget Sound/Georgia Basin DPS	T - 75 FR 22276	79 FR 68041	-- --	↓

Species	ESA Status	Critical Habitat	Recovery Plan	Trend
Yelloweye rockfish (*Sebastes ruberrimus*)				
Puget Sound/Georgia Basin DPS	T - 75 FR 22276	79 FR 68041	-- --	↓
Fish – Sturgeon				
Green sturgeon (*Acipenser medirostris*)				
Southern DPS	T - 71 FR 17757	73 FR 52084	-- --	unknown
Fish – Pacific Eulachon / Smelt (*Thaleichthys pacificus*)				
Southern DPS	T - 75 FR 13012	76 FR 65323	-- --	↓
Fish – Salmonids				
Chinook Salmon (*Oncorhynchus tshawytscha)* Evolutionarily Significant Units (ESU)				
Puget Sound ESU	T - 64 FR 14308	70 FR 52630	2007	↓
Lower Columbia River ESU	T – 64 FR 14308	70 FR 52630	2013	↓
Upper Columbia River spring-run ESU	E – 64 FR 14308	70 FR 52630	2007	↓
Upper Willamette River ESU	T – 64 FR 14308	70 FR 52630	2011	↓
Snake River spring/summer-run ESU	T – 59 FR 42529	64 FR 57399	--	↑
Snake River fall-run ESU	T – 59 FR 42529	58 FR 68543	--	↑
Central Valley, spring-run ESU	T – 64 FR 50393	70 FR 52488	2014	↓
Central Valley, spring-run ESU	NEP – 78 FR 79622	--	--	--
California Coastal ESU	T – 64 FR 50393	70 FR 52488	2007	unknown
Sacramento River, Winter-run ESU	E – 59 FR 440	58 FR 33212	2014	↓
Chum Salmon (*Oncorhynchus keta*) ESUs				
Columbia River ESU	T – 64 FR 14507	70 FR 52630	2013	Temporally variable
Hood Canal Summer Run ESU	T - 64 FR 14507	70 FR 52630	2007	Stable
Coho Salmon (*Oncorhynchus kisutch*) ESUs				
Lower Columbia River ESU	T - 70 FR 37160	78 FR 2725 (proposed)	2013	↓
Oregon Coast ESU	T – 63 FR 42587	64 FR 24049	-- --	↑

Species	ESA Status	Critical Habitat	Recovery Plan	Trend
So. Oregon Northern California Coast ESU	T- 76 FR 50447	64 FR 24049	2014	↓
Central California Coast ESU	E – 61 FR 56138	65 FR 7764	2012	↓
Sockeye Salmon (*Oncorhynchus nerka*) ESUs				
Ozette Lake ESU	T – 64 FR 14528	70 FR 52630	2009	Not likely increasing
Snake River ESU	E – 56 FR 58619	58 FR 68543	2015	↑
Steelhead Trout (*Oncorhynchus mykiss*) Distinct Population Segments (DPS)				
Lower Columbia River DPS	T - 71 FR 834	70 FR 52630	2013	Temporally variable
Upper Columbia River DPS	T – 71 FR 834	72 FR 52630	2007	↑
Middle Columbia River DPS	T – 71 FR 834	70 FR 52630	2009	Likely stable
Middle Columbia River DPS	NEP – 78 FR 2893	-- --	-- --	-- --
Puget Sound DPS	T - 72 FR 26722	-- --	-- --	↓
Upper Willamette River DPS	T – 71 FR 834	70 FR 52630	2011	↓
Northern California DPS	T - 71 FR 834	70 FR 52488	-- --	unknown
Central California Coast DPS	T - 71 FR 834	70 FR 52488	-- --	unknown
Snake River Basin DPS	T – 71 FR 834	70 FR 52630	-- --	unknown
California Central Valley DPS	T – 71 FR 834	70 FR 52488	2014	↓
South-Central California Coast DPS	T – 71 FR 834	70 FR 52488	2013	↓
Southern California DPS	E – 71 FR 834	70 FR 52488	2012	Declining to stable

***ESA-listed humpback whales in the Action Area were proposed as a threatened Central America DPS on April 21, 2015. Although this DPS designation is not final, this Opinion will analyze the effects of the action on both the currently listed entity (range-wide) and the** proposed DPS.

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. We applied these criteria to the species listed at the beginning of this section; this subsection summarizes the results of those evaluations.

4.1.1 North Pacific Right Whale

North Pacific right whale remains one of the most endangered whale species in the world, likely numbering fewer than 1,000 individuals between the eastern and western populations. Historically, the endangered North Pacific right whale occurred in waters off the coast of British Columbia and the States of Washington, Oregon, and California (Clapham et al. 2004b; Scarff 1986b). 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 (Kraus et al. 1986); (Clapham et al. 2004a). 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. 2004a); (Shelden et al. 2005); (Gregr 2011); (Ivashchenko et al. 2013). Right whales were probably never common along the west coast of North America (Scarff 1986a); (Brownell Jr. et al. 2001). The rarity of reports for right whales in more southern coastal areas in winter in either historical or recent times suggests that their breeding grounds may have been offshore (Clapham et al. 2004a). 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. 2010). 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. 2011a); (Zerbini et al. 2010). In addition to sighting data, (Wade et al. 2011b), (Wade et al. 2011a), (Matsuoka et al. 2013a), passive acoustic data have indicated the presence of North Pacific right whales in the Gulf of Alaska (Mellinger et al. 2004b; Sirovic et al. 2015). No right whales were detected from more than 5,324 hours of passive acoustic data obtained from Navy-funded monitoring devices in the north-central Gulf of Alaska (Baumann-Pickering et al. 2012; Debich et al. 2013), 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).

4.1.1.1 *Occurrence in the Offshore Portion of the NWTT Action Area*

Various sightings of North Pacific right whales in the general vicinity of the Action Area have occurred on an irregular basis. Two right whales were sighted in 1983 on Swiftsure Bank at the entrance to the Strait of Juan de Fuca (Osborne et al. 1988b). In May 1992 there was a sighting of a single of North Pacific right whale over Quinault Canyon in the NWTT Action Area (Green et al. 1992a); (Rowlett et al. 1994). There were no sightings of North Pacific right whales during six ship surveys conducted in summer and fall off California, Oregon, and Washington from 1991 through 2008 (Barlow 2010b). Recently, two sightings and an acoustic detection of a North Pacific right whale have occurred in the vicinity of or in the Action Area. On June 20, 2013, a newspaper reported that a single right whale was sighted in waters off Haida Gwaii, British Colombia (located approximately 200 nautical miles (nm; 370 km) north of the Action Area; Hume (2013)). Right whales had not been detected by two bottom deployed passive acoustic monitoring devices in the offshore waters of Washington State from 2011 through 2012, but on June 29, 2013 (9 days after the sighting to the north), a Navy-funded passive acoustic monitoring device at Quinault Canyon (in the Action Area) detected two right whale calls within a two-hour period (Sirovic et al. 2015). Approximately 4 months later (October 2013) another (different) right whale was sighted in a group of humpback whales off the entrance to the Strait of Juan de Fuca and moving south (DoN 2015b; Pynn 2013). Because of the low population numbers (likely less than 1,000) in the North Pacific and the few individuals that have been observed (Brownell Jr. et al. 2001); (Wade et al. 2006), 2011b), even given these recent sightings/detections, this species is considered extremely rare in the Action Area. Based on this information, there is a very low probability of encountering this species anywhere in the coastal and offshore waters in the Action Area and their occurrence is therefore considered discountable and thus not considered further in this Opinion.

4.1.1.2 *Occurrence in Inland Waters of the NWTT Action Area*

As noted above, the rarity of coastal records suggests right whales would not be present in more inland areas. The occurrence of a North Pacific right whale within the Inland Waters is considered extralimital and thus not considered further in this Opinion.

4.1.1.3 *Occurrence in the Western Behm Canal, Alaska*

North Pacific right whales were not observed during the Alaska Fisheries Science Center's National Marine Mammal Laboratory 1991 through 2007 surveys of the inland waters of southeast Alaska (Dahlheim et al. 2009). Given their small population size (likely less than

1,000) and lack of sightings in southeast Alaska as noted above, North Pacific right whales are considered extralimital within the Behm Canal portion of the NWTT Action Area and therefore are not considered further in this Opinion.

4.1.1.4 ***Critical Habitat***

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 and the Bering Sea that support the species.

The Navy's training and testing activities would not occur in the designated critical habitat nor would the activities be expected to have any impacts to the critical habitat or the primary constituent elements. Therefore, Navy NWTT activities in the Action Area are not likely to adversely affect the designated critical habitat for North Pacific right whales. As a result, we will not consider this critical habitat in greater detail in the remainder of this Opinion.

4.1.1.5 ***Conclusion***

The extremely low population numbers of this species in the North Pacific Ocean over the past five decades and the rarity of reports from these waters suggests that these whales are not reasonably likely to be exposed to the activities considered in this consultation. As a result, this species will not be considered in the remainder of this Opinion. Due to the North Pacific right whale's discountable likelihood of exposure to NWTT training and testing activities, and the lack of designated critical habitat for this species in the Action Area, we have determined the North Pacific right whale is not likely to be adversely affected by NWTT activities and will not be considered further in this Opinion.

4.1.2 Western North Pacific Gray Whale

Gray whales (*Eschrichtius robustus*) 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.

Gray whales occur in two genetically distinct populations on the eastern and western sides of 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. 2006; Weller et al. 2007; Weller et al. 2004; Weller et al. 2006).

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. 2008; Weller et al. 2012b).

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 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. 2012a). 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. 2012b).

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).

4.1.2.1 ***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. 2012b; 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).

4.1.2.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

During visual surveys off the Washington coast from August 2004 through September 2008, there were a total of 55 gray whale sightings of 116 individuals (Oleson et al. 2009). Clear seasonal differences in gray whale distribution were noted based on three distinct time periods: (1) winter (December through January), corresponding to the timing of their southbound migration; (2) spring (February through April), corresponding to the timing of their northbound migration; and (3) summer/fall (May through October), a time when any gray whales present are primarily members of the Pacific Coast Feeding Group. Oleson et al. (2009) found significant differences in the sighting distributions between these three time periods, based on an analysis of distance from shore, distance from the shelf break, and water depth. During the winter southbound migration, gray whales were sighted mainly offshore, with an average distance of 18 mi. (29 km) from the coast. This compared to the spring northbound migration when the average distance was 6.2 mi. (10 km) from shore. During summer and fall, gray whale sightings were clustered in two areas, in and around the entrance to Grays Harbor, Washington and in an offshore area approximately 12 to 15 mi. (20 to 25 km) from shore (Oleson et al. 2009). These offshore sightings were unusual given that in the Pacific Northwest the Pacific Coast Feeding Group is typically close to shore (Calambokidis et al. 2002).

The occurrence of Eastern North Pacific gray whales and members of the Pacific Coast Feeding Group in the Action Area is considered to be seasonal and to most likely occur in the offshore portion of the Action Area during migrations between summer feeding grounds and winter breeding grounds (Carretta et al. 2013a). Given their small population size and limited number of sightings off the U.S. west coast, the occurrence of Western North Pacific gray whales in the offshore portion of the Action Area is rare and not considered further in this Opinion.

4.1.2.1.2 Occurrence in Inland Waters of the NWTT Action Area

As the majority of gray whales migrate past the Strait of Juan de Fuca en route to or from their feeding or breeding grounds, a few of them enter the inland waters to feed. Gray whales are observed in Washington inland waters in all months of the year (Calambokidis et al. 2010; Washington State Department of Fish and Wildlife 2012), with peak numbers from March through June (Calambokidis et al. 2010). Typically fewer than 20 gray whales have been documented annually in the inland waters of Washington and British Columbia based on a review of Orca Network (Washington State Department of Fish and Wildlife 2012). Calambokidis et al. (2009a) reported that Puget Sound (mudflats near the Whidbey Island and Camano Island area) is used as a springtime feeding area for a small, regularly occurring group of gray whales. Observed feeding areas are located in Saratoga Passage between Whidbey and Camano Islands including Crescent Harbor, and in Port Susan Bay located between Camano Island and the mainland in Possession Sound. These areas are between NASWI (Crescent Harbor) and NAVSTA Everett.

In the Rich Passage to Agate Passage area in the vicinity of NAVBASE Kitsap Bremerton and Keyport, 11 opportunistic sightings of gray whales were reported to Orca Network between January 2003 and July 2012. One stranding, unassociated with Navy activities, occurred at NAVBASE Kitsap Bremerton in January 2013. There are typically anywhere from 2 to 10 stranded gray whales per year in Washington (Cascadia Research 2012). Gray whales have been sighted in Hood Canal south of the Hood Canal Bridge on six occasions since 1999, including a stranded whale at Belfair State Park. The most recent report in Hood Canal was of characteristic "blows" (air exhaled through the whale's blowhole) in the waters near Lilliwaup in November 2010.

4.1.2.1.3 Occurrence in the Western Behm Canal, Alaska – NWTT Action Area

Gray whales were not observed during 1991 through 2007 surveys of the inland waters of southeast Alaska (Dahlheim et al. 2009), and they are considered extralimital in this region of the NWTT Action Area and therefore not considered further in this Opinion.

4.1.2.2 ***Critical Habitat***

NMFS has not designated critical habitat for Western North Pacific gray whale.

4.1.2.3 ***Conclusion***

Western North Pacific gray whales may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area of the Action Area; however, there are no predicted effects resulting from the Navy's modeling efforts and our analysis. 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.

Gray whales may also be exposed to sonar or other active acoustic stressors when their presence coincides with testing activities in the NWTT Action Area. Acoustic modeling predicts that the Western North Pacific stock of gray whales may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area; however, as with training, there are no predicted effects from these testing activities that rise to the level of "take."

Therefore, we conclude that stressors from training and testing activities may affect Western North Pacific gray whales, but those effects are expected to be minor and would not rise to the level of "take" as defined by the ESA and therefore would not adversely affect this species.

4.1.3 Green Sea Turtle

Four of the seven living species of sea turtles including leatherback (*Dermochelys coriacea*), loggerhead (*Caretta caretta*), olive ridley (*Lepidochelys olivacea*), and green (*Chelonia mydas*) have the potential to be found in the NWTT Action Area. The three hard-shell turtles of the Cheloniidae family (loggerhead, olive ridley, and green) are considered tropical, subtropical, and warm temperate species that rarely stray into cold waters (Eckert 1993b). Most hard-shell turtles seek optimal seawater temperatures near 65 degrees Fahrenheit (°F) (18.3 degrees Celsius [°C]) and are cold-stressed at seawater temperatures below 50°F (10°C) (Mrosovsky 1980; Schwartz 1978). Under certain oceanographic conditions (e.g., warmer currents), all four species could occur off the Washington and Oregon coasts (and occasionally in Alaska waters). However, the cold waters off Washington and Oregon are above the typical northern limits for the three hard-shell turtles; therefore, these species are considered rare in the NWTT Action Area. In contrast, leatherback sea turtles regularly occur in cold temperate waters characteristic of higher latitudes (Eckert et al. 1989a; Pritchard 1982b).

Green sea turtles occur along the coasts of British Columbia and the States of Washington, Oregon, and northernmost California (Bowlby et al. 1994), but those occurrences are usually associated with mild or strong El Nino currents that push warmer water masses northward. When those water masses dissipate, as has happened at least twice over the past two years, green sea turtles become hypothermic in the colder, ambient temperatures. Because the NWTT Action Area occurs at the thermal limits of green sea turtles (primarily because of low sea surface temperatures), the probability of green sea turtles occurring in the Action Area is sufficiently small for us to conclude that green sea turtles are not likely to be exposed to the activities considered in this consultation. As a result, this species will not be considered in greater detail in the remainder of this Opinion.

Critical habitat was designated in 1998. There is no designated green sea turtle critical habitat within the Action Area. Therefore, Navy NWTT activities in the Action Area will not affect the designated critical habitat for green sea turtles. As a result, we will not consider this critical habitat further in the remainder of this Opinion.

4.1.4 Loggerhead Sea Turtle

Loggerhead sea turtles occur along the coasts of British Columbia and the States of Washington, Oregon, and northernmost California, but those occurrences are usually associated with mild or strong El Nino currents that push warmer water masses northward. When those water masses dissipate, as has happened at least twice over the past two years, loggerhead sea turtles become hypothermic in the colder, ambient temperatures. Because the Action Area occurs at the thermal limits of loggerhead sea turtles (primarily because of low sea surface temperatures), the probability of loggerhead sea turtles occurring in the Action Area is sufficiently small for us to conclude that loggerhead sea turtles are not likely to be exposed to the activities considered in this consultation. As a result, this species will not be considered in greater detail in the remainder of this Opinion.

4.1.5 Olive Ridley Sea Turtle

Like green sea turtles, olive ridley sea turtles also occur along the coasts of British Columbia and the States of Washington, Oregon, and northernmost California, but those occurrences are usually associated with mild or strong El Nino currents that push warmer water masses northward. When those water masses dissipate, as has happened at least twice over the past two years, olive ridley sea turtles become hypothermic in the colder, ambient temperatures. Because the Action Area occurs at the thermal limits of olive ridley sea turtles (primarily because of low sea surface temperatures), the probability of olive ridley sea turtles occurring in the Action Area is sufficiently small for us to conclude that olive ridley sea turtles are not likely to be exposed to the activities considered in this consultation. As a result, this species will not be considered in greater detail in the remainder of this Opinion.

4.1.6 Ozette Lake Sockeye Salmon

The NMFS originally listed Ozette Lake sockeye salmon ESU as a threatened species in 1999 (64 FR 14528). This classification was retained on June 28, 2005 (70 FR 37160). For additional detail on the life history of steelhead, please refer to section 4.2.16.

Spatial Structure and Diversity: This species includes all naturally spawned populations of sockeye salmon in Ozette Lake and streams and tributaries flowing into Ozette Lake, Washington, and progeny of two artificial propagation programs. The Lake Ozette (LO) Technical Recovery Team concluded that five extant spawning aggregations in Ozette Lake are different subpopulations within a single population (Currens et al. 2009; NMFS 2009b). The subpopulations can be grouped according to whether they spawn in tributaries or near lake beaches (NMFS 2009b).

Abundance and Productivity: LO sockeye salmon population sizes remain very small compared to historical sizes. Additionally, population estimates remain highly variable and uncertain, making it impossible to detect changes in abundance trends or in productivity in recent years. The most recent brood years (1999 to 2003) have had the lowest average recruits per spawner. Spatial structure and diversity are also difficult to appraise; there is currently no successfully quantitative program to monitor beach spawning or spawning at other tributaries. Assessment methods must improve to evaluate the status of this species and its responses to recovery actions. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011a).

The most recent five-year geometric mean (2006 to 2010) of natural-origin, LO sockeye salmon returning to spawn in the Lake Ozette watershed is 1,683 (Table 16).

Table 16. Five-year geometric means for adult natural-origin and hatchery-origin spawners for the LO sockeye salmon ESU (unpublished data, Norma Jean Sands, NWFSC, Apr. 10, 2012; as cited in NMFS 2015).

Year Range	Natural-origin Escapement	Hatchery-origin Escapement
1977-1980	1,746	0
1981-1985	1,700	0
1986-1990	2,432	17
1991-1995	849	85
1996-2000	2,285	307
2001-2005	3,431	285
2006-2010	1,683	33

Spawning habitat capacity estimates for beach and tributary habitats (combined) range from 90,000 to 120,000 adult OL sockeye salmon (PSTRT 2007). These estimates are based upon a relatively low spawning density target of one female per three sq. meters of suitable habitat. However, historical spawning density may have been as high as one female/sq. meter, which would triple the capacity estimates. Nonetheless, the most recent five-year average for natural origin adult sockeye escapement is only 1.9 percent of the lower estimate (1,716/90,000).

Juvenile LO sockeye abundance can be estimated from escapement data. Fecundity estimates for the ESU average 3,050 eggs per female (Haggerty et al. 2009), and the proportion of female spawners is assumed to be 50 percent of escapement. By applying fecundity estimates to the expected escapement of females (both natural-origin and hatchery-origin spawners – 858 females), the ESU is estimated to produce approximately 2.62 million eggs annually. Analyzing data from1991 to 2007 for the Lake Washington sub-basin, (McPherson and Woodey 2009) found an average egg-to-fry survival rate of 13.5 percent (range 1.9 to 32.0 percent). Assuming a similar 13.5 percent egg-to-fry survival for Lake Ozette, the ESU should produce roughly 353,282 natural outmigrants annually.

Limiting factors include (NMFS 2009b; NOAA Fisheries 2011; USDC 2009a):

Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, lake beach spawning habitat, and stream substrate have been degraded as a result of cumulative impacts of forest practices, agriculture, and development.

Predation: Harbor seals and river otters, and predaceous non-native and native fish species, are reducing the abundance of adult fish that successfully spawn, and the abundance of sockeye smolts escaping seaward from the watershed each year.

4.1.6.1 ***Conclusion***

As stated in the *Effects of the Proposed Action* section of this Opinion, 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. (1976a) 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 (north of the NWTT Action Area). Pearcy and Fisher (1990) observed the highest CPUE of juvenile sockeye inshore of 37 km, though some were caught over 55 km offshore. They noted that, similar to juvenile chum salmon, juvenile sockeye salmon were less abundant than either coho or salmon off the Oregon and Washington coast.

As described in the *Effects of the Proposed Action* section of this Opinion, the only stressor we determined was likely to adversely affect ESA-listed salmonids was explosive detonation. Since the Navy will not conduct explosive activities in the Olympic Coast National Marine Sanctuary and considering the oceanic migration path described above, Lake Ozette sockeye would be expected to be north of the Action Area before they would be far enough offshore to co-occur with Navy explosive activities. Therefore, the likelihood of affecting Ozette Lake sockeye salmon is so low as to be discountable.

4.1.7 Southern California Steelhead

Southern California steelhead were listed as endangered in 1997 (62 FR 43937), when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). For additional detail on the life history of steelhead, please refer to section 4.2.17.

Spatial Structure and Diversity: The geographic range of this DPS extends from the Santa Maria River, near Santa Maria, to the California-Mexico border, which represents the known southern geographic extent of the anadromous form of *O. mykiss*. NMFS described historical and recent steelhead abundance and distribution for the southern California coast through a population characterization (Boughton et al. 2006). Surveys in Boughton et al. (2005) indicate between 58 percent and 65 percent of the historical steelhead basins currently harbor *O. mykiss* populations at sites with connectivity to the ocean. Most of the apparent losses of steelhead were noted in the south, including Orange and San Diego counties (Boughton et al. 2005).

Abundance and Productivity: While 46 drainages support this DPS (Boughton et al. 2005), only 10 population units possess a high and biologically plausible likelihood of being viable and independent[5] (Boughton et al. 2006). Very little data regarding abundances of Southern California Coast steelhead are available, but the picture emerging from available data suggest very small (<10 fish) but surprisingly consistent annual runs of anadromous fish across the diverse set of basins that are currently being monitored (Williams et al. 2011). The most significant population that has been recently monitored is in Topanga Creek, where mark-recapture studies were done in 2007 and 2008. According to the authors (Bell et al. 2011), that data indicated a population of resident fish whose abundance is on the order of 500 individuals, including all size and age classes in Topanga Creek. It is believed that population abundance trends can significantly vary based on yearly rainfall and storm events within the range of the Southern California Coast DPS (Williams et al. 2011). A relatively large number of adult steelhead were observed in 2008, two years after an extended wet spring that presumably gave smolts ample opportunity to migrate to the ocean. Some of the strength of the 2008 season may also be an artifact of conditions that year. Low rainfall appears to have caused many spawners to get trapped in freshwater, where they were observed during the summer; in addition, low rainfall probably improved conditions for viewing fish during snorkel surveys, and for trapping fish in weirs (Williams et al. 2011). There is little new evidence to suggest that the status of the Southern California DPS has changed appreciably in either direction since publication of the most recent collections of status reviews (Good et al. 2005a; NMFS 2011d; Williams et al. 2011).

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[5] Independent population: a collection of one or more local breeding units whose population dynamics or extinction risk over a 100-year time period is not substantially altered by exchanges of individuals with other populations (Boughton et al. 2006).

Limiting Factors: The majority of lost populations (68 percent) in this DPS have been associated with anthropogenic barriers to steelhead migration (e.g., dams, flood-control structures, culverts, etc.). Additionally, investigators have found that barrier exclusions are statistically associated with highly-developed watersheds. This DPS experiences a high magnitude of threat to a small number of extant populations vulnerable to extirpation due to loss of accessibility to freshwater spawning and rearing habitat, low abundance, degraded estuarine habitats and watershed processes essential to maintain freshwater habitats (NMFS 2011d). The recovery potential is low to moderate due to the lack of additional populations, lack of available/suitable freshwater habitat, steelhead passage barriers, and inadequate instream flow.

4.1.7.1 ***Conclusion***

Given the extremely low abundance of ESA-listed Southern California steelhead in general and within the Action Area and the limited likelihood of co-occurrence with training and testing stressors, the likelihood of affecting Southern California steelhead is so low as to be discountable.

4.1.8 Southern DPS of Green Sturgeon

Green sturgeon are long-lived, slow-growing fish and the most marine-oriented of the sturgeon species. Mature males range from 4.5 to 6.5 feet (1.4 to 2 m) in "fork length" and do not mature until they are at least 15 years old, while mature females range from 5 to 7 feet (1.6 to 2.2 m) fork length and do not mature until they are at least 17 years old. Maximum ages of adult green sturgeon are likely to range from 60 to 70 years (Moyle 2002b). This species is found along the west coast of Mexico, the United States, and Canada.

Although they are members of the class of bony fishes, the skeleton of sturgeons is composed mostly of cartilage. Sturgeon lack scales; however, they have five rows of characteristic bony plates on their body called "scutes". The backbone of the sturgeon curves upward into the caudal fin, forming their shark-like tail. On the ventral, or underside, of their flattened snouts are sensory barbels and a siphon-shaped, protrusible, toothless mouth. Recent genetic information suggests that green sturgeon in North America are taxonomically distinct from morphologically similar forms in Asia.

4.1.8.1 ***Populations***

Two DPSs have been defined for green sturgeon (*Acipenser medirostris*), a northern DPS (spawning populations in the Klamath and Rogue rivers) and a southern DPS (spawners in the Sacramento River). Southern green sturgeon includes all naturally-spawned populations of green sturgeon that occur south of the Eel River in Humboldt County, California. Only the Southern DPS of green sturgeon is listed under the ESA.

4.1.8.2 ***Distribution***

Green sturgeon occur along the west coast of North America from Mexico to the Bering Sea (Adams et al. 2002; Colway and Stevenson 2007; Moyle 2002a), as well in freshwater rivers and estuaries in between. Both northern DPS and southern DPS green sturgeon occupy coastal estuaries and nearshore marine waters from southern California to Alaska, including Humboldt Bay, the lower Columbia River estuary, Willapa Bay, Grays Harbor and southeast Alaska. In general, green sturgeon are more common north of Point Conception, California. This DPS occupies a limited number of river systems, primarily the main stems and tributaries of the Sacramento, Feather, and San Joaquin rivers, all of which drain to San Francisco Bay (Israel et al. 2009). The Sacramento River is the only known spawning habitat for the Southern green sturgeon DPS (CDFG 2002; Moyle et al. 1992b). Spawning in the San Joaquin River does not presently occur, likely as a result of damming and irrigation development. Adult and juvenile green sturgeon occur throughout the Sacramento River. Although Southern DPS individuals in freshwater occur primarily in the aforementioned waterways, some have been captured in rivers of the Northern DPS, including the Klamath, Rogue, and Columbia rivers of northern California and Oregon. Individuals have also rarely been captured in Southern California waters, but their abundance increases north of Point Conception.

4.1.8.3 ***Habitat and movement***

Green sturgeon are believed to spend the majority of their lives in nearshore oceanic waters, bays, and estuaries. Early life-history stages reside in fresh water, with adults returning to freshwater to spawn when they are more than 15 years of age and more than 4 feet (1.3 m) in size. Spawning is believed to occur every 2 to 5 years (Moyle 2002b). Adults typically migrate into fresh water beginning in late February; spawning occurs from March through July, with peak activity from April through June (Moyle et al. 1995b). Females produce 60,000 to 140,000 eggs (Moyle et al. 1992a). Juvenile green sturgeon spend 1 to 4 years in fresh and estuarine waters before dispersal to saltwater (Beamesderfer and Webb 2002a). Upon outmigration from fresh water, subadult green sturgeon disperse widely along through continental shelf waters of the west coast within the 110 meter contour (BRT 2005; Erickson and Hightower 2007; Moyle et al. 1992b). It appears that Southern DPS green sturgeon preferentially distribute north of their natal river during fall and move into bays and estuaries during summer and fall (Israel et al. 2009; Moser and Lindley 2007). Spawning individuals move south during spring to natal freshwater habitat (Lindley et al. 2008).

Green sturgeon appear to prefer marine areas with high seafloor complexity and boulder presence (Huff et al. 2011). Preferred areas also include depths of 20 to 60 m, with temperatures of 9.5 to 16° C (Huff et al. 2011). In the San Joaquin River, green sturgeon appear to prefer slopes or shoulders of navigational channels, not shallower shoals of less than 7 m (Peterson et al. 2011). Channel centers are also frequently utilized (Peterson et al. 2011).

4.1.8.4 ***Reproduction***

Adults infrequently occupy freshwater reaches primarily to spawn and spend the majority of their lives in estuaries and coastal marine waters (Wilson and McKinley 2004). Sturgeon, like salmon, possess strong site fidelity and will return to their natal streams to spawn (Bemis and Canard 1997). This occurs between late February and July, with a peak in mid-April to mid-June, hold in deep pools and return to salt water in the fall early, often with the first increases in fall flows, although Northern DPS individuals in the Rogue River seek out slow-flow and off-channel coves (Erickson et al. 2001; Erickson and Webb 2007a; Heublein et al. 2009; Moyle et al. 1992b; Moyle et al. 1995a; Rien et al. 2001). Fish then tend to aggregate in deep pools, where they will over-summer before outmigrating in the fall, although some fish have been observed outmigrating relatively soon after presumed spawning events (Heublein et al. 2009). However, multiple spawning events may occur annually, unlike annual spawning of salmonids (Heublein et al. 2009). Adults may spend over 6 months in freshwater until water temperatures drop below 50º F in fall or winter or when a significant flow even occurs, although temperature or flow cues are not always needed to initiate downstream migration (Erickson and Webb 2007a; Heublein et al. 2009). In the Sacramento River adult green sturgeon spawn in late spring and early summer above Hamilton City, above Red Bluff Diversion Dam, and possibly as far upstream as Keswick Dam (CDFG 2002; Heublein et al. 2009). It appears that specific habitat for spawning includes large cobblestones (where eggs can settle between), although spawning is known to occur over clean sand or bedrock.

Adults likely return on a 2 to 5 year basis for spawning starting at 15 to 30 years of age for males and 17 to 40 for females (Adams et al. 2002; Moyle 2002a; Van Eenennaam et al. 2006). Most male spawners are young (17 to 18 years) while females on the spawning grounds are often older (27 to 28 years). Females produce roughly 60,000 to 140,000 eggs per spawning event (Moyle et al. 1992b; Scott and Crossman 1973). Each egg is large by sturgeon standards so as to contain greater nourishment for embryos (Cech Jr. et al. 2000). Temperature may trigger spawning behavior, with ranges of 48º to 62º F being influential (Moyle et al. 1995a). Water temperature is also critical for egg survival with temperatures above 68º F being fatal to developing embryos (Cech Jr. et al. 2000).

4.1.8.5 ***Growth***

Green sturgeon spend their first 1 to 4 years in their natal streams and rivers (Beamesderfer and Webb 2002b; Nakamoto et al. 1995), although they are believed to be physiologically adapted to sea water survival at 6 months of age (Allen and Cech 2007; Allen et al. 2009; Allen and J.J. Cech 2007). Larvae are active at night, a behavior that likely reduces predation and avoids being moved downstream more than necessary (Cech Jr. et al. 2000). Green sturgeon larvae grow very rapidly, reaching about 300 mm by age one (Deng 2000). Growth of the larval stage is ideal at 59º F, reduced below 52º F and above 66º F, and highly reduced above 75º F (Cech Jr. et al.

2000). Juvenile green sturgeon of the Northern DPS (Rogue River) have been captured in the Rogue River estuary from April until the end of November (Rien et al. 2001). Green sturgeon are a long-lived fish, and likely live for 60 to 70 years (Moyle 2002a).

4.1.8.6 ***Feeding***

While in fresh water, juveniles feed on a variety of fishes and invertebrates (Moyle et al. 1992b). One juvenile from the Sacramento-San Joaquin estuary was found to have preyed most commonly upon opisthobranch mollusks (*Philline* sp.), with bay shrimp (*Crangon* sp.) and overbite clams (*Potamocorbula amurensis*) as secondary prey. Other juveniles in the Sacramento River delta feed on opossum shrimp (*Neomysis mercedis*) and *Corophium* amphipods (Radtke 1966).

The limited feeding data available for adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods, and small fish (Dumbauld et al. 2008; Houston 1988; Moyle et al. 1992b; Wilson and McKinley 2004). Starting as larvae, sturgeon use electroreception to identify prey. Olfaction and taste may also be important to foraging, while vision is thought play a minor role in prey capture (Miller 2004).

4.1.8.7 ***Hearing and vocalization***

We do not have specific information on hearing in green sturgeon. However, Meyer and Popper (Meyer and Popper 2002) recorded auditory evoked potentials to pure tone stimuli of varying frequency and intensity in lake sturgeon and reported that lake sturgeon detect pure tones from 100 to 2000 Hz, with best sensitivity from 100 to 400 Hz. They also compared these sturgeon data with comparable data for oscar (*Astronotus ocellatus*) and goldfish (*Carassius auratus*) and reported that the auditory brainstem responses for the lake sturgeon are more similar to the goldfish (which is considered a hearing specialist that can hear up to 5,000 Hz) than to the oscar (which is a non-specialist that can only detect sound up to 400 Hz); these authors, however, felt additional data were necessary before lake sturgeon could be considered specialized for hearing.

Lovell et al. (2005) also studied sound reception in and the hearing abilities of paddlefish (*Polyodon spathula*) and lake sturgeon (*Acipenser fulvescens*). They concluded that both species were responsive to sounds ranging in frequency from 100 to 500 Hz with lowest hearing thresholds from frequencies in bandwidths between 200 and 300 Hz and higher thresholds at 100 and 500 Hz. We assume that the hearing sensitivities reported for these other species of sturgeon are representative of the hearing sensitivities of southern green sturgeon.

4.1.8.8 ***Status and trends***

NMFS listed the southern population of the North American green sturgeon as threatened on April 7, 2006 (71 FR 17757). Trend data for green sturgeon is severely limited. Available information comes from two predominant sources, fisheries and tagging. Only three data sets were considered useful for the population time series analyses by NMFS' biological review team: the Klamath Yurok Tribal fishery catch, a San Pablo sport fishery tag returns, and Columbia River commercial landings (BRT 2005). Using San Pablo sport fishery tag recovery data, the California Department of Fish and Game produced a population time series estimate for the southern DPS. San Pablo data suggest that green sturgeon abundance may be increasing, but the data showed no significant trend. The data set is not particularly convincing, however, as it suffers from inconsistent effort and since it is unclear whether summer concentrations of green sturgeon provide a strong indicator of population performance (BRT 2005). Although there is not sufficient information available to estimate the current population size of southern green sturgeon, catch of juveniles during state and federal salvage operations in the Sacramento delta are low in comparison to catch levels before the mid-1980s.

4.1.8.9 ***Natural threats***

Green sturgeon eggs and larvae are likely preyed upon by a variety of larger fish and animals, while sub-adult and adult sturgeon may occasionally be preyed upon by shark, sea lions, or other large body predators. Physical barriers, changes in water flow and temperatures may also affect fresh water survival.

4.1.8.10 ***Anthropogenic threats***

The principle threat to southern green sturgeon comes from a drastic reduction in available spawning area from impassible barriers (e.g., Oroville, Shasta and Keswick dams). Other threats include potentially lethal temperature limits, harvest, entrainment by water projects and toxins and invasive species (Adams et al. 2007; Erickson and Webb 2007b; Lackey 2009). Since this DPS is composed of a single spawning population within the Sacramento River, stochastic variation in environmental conditions and significant fluctuations in demographic rates increases the risk of extinction for this DPS.

Climate change has the potential to affect sturgeon in similar, if not more significant ways it affects salmonids. Elevated air temperatures could lead to precipitation falling as rain instead of snow. Additionally, snow would likely melt sooner and more rapidly, potentially leading to greater flooding during melting and lower water levels at other times, as well as warmer river temperatures. Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Moyle et al. 1992a; Moyle et al. 1995b). Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007). If water flow is low during migration events, it is likely that new obstacles can impede or block sturgeon movement. As with other anadromous fishes, sturgeon are uniquely evolved to the environments that they live in. Because of this specificity, broad scale changes in environment can be difficult to adapt to, including changes in water temperature. Sturgeon are also sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initial spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007). If river and lake temperatures become anomalously warm, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges in freshwater habitats. Apart from direct changes to sturgeon survival, altered water temperatures may disrupt habitat, including the availability of prey (ISAB 2007). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Overall, it is likely that global warming will increase pressures on sturgeon survival and recovery.

Poaching also poses an unknown but potentially serious threat because of high demand for sturgeon caviar. Retention of green sturgeon in both recreational and commercial fisheries is now prohibited within the western states, but the effect of capture/release in these fisheries is unknown. There is evidence of fish being retained illegally, although the magnitude of this activity likely is small (NOAA Fisheries 2011).

Studies from other sturgeon species indicate that sturgeon readily bioaccumulate contaminants. White sturgeon from the Kootenai River have been found to contain aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, zinc, DDE, DDT, PCBs, and other organochlorines (Kruse and Scarnecchia 2001). Mercury has also been identified from white sturgeon of the lower Columbia River (Webb et al. 2006). Numerous organochlorines, including DDT, DDD, DDE, chlordane, and dieldrin have also been identified in these fish (Foster et al. 2001). Observed concentrations are likely sufficient to influence reproductive physiology.

4.1.8.11 *Conclusion*

The offshore Action Area overlaps with the marine distribution of green sturgeon off the coast of Oregon, Washington, and California. As stated in the *Status of Listed Resources* section of this Opinion, green sturgeon prefer marine areas with high seafloor complexity and boulder presence at depths of 20 to 60 m, with temperatures of 9.5 to 16° C (Huff et al. 2011). The limited feeding data available for adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods, and small fish (Dumbauld et al. 2008; Houston 1988; Moyle et al. 1992b; Wilson and McKinley 2004).

Information regarding their preference for areas of high seafloor complexity and prey selection in coastal waters (benthic prey) indicate green sturgeon reside and migrate along the seafloor while in coastal waters. Additionally, Huff et al. (2011) suggest that green sturgeon occur at low densities in the coastal marine environment. Given the infrequent nature of underwater explosions in the offshore environment, the low density of green sturgeon in the offshore environment, the species' preference for benthic habitat when in marine waters, the likelihood of a green sturgeon encountering an explosive activity taking place anywhere in the offshore is remote. Therefore, explosions in the offshore Action Area are not likely to adversely affect green sturgeon.

Green sturgeon are not known to spawn in any rivers in Puget Sound. Additionally, very few green sturgeon have been found in Puget Sound, with only two confirmed observations in 2006 (NMFS 2012b). Given the infrequent nature of underwater explosions in inland waters of the Action Area and the low documented abundance of green sturgeon in Puget Sound, we believe the likelihood of these species encountering elevated underwater noise from an explosion to be remote. Because of this, green sturgeon are not likely to be adversely affected by underwater explosions in inland waters.

The southern DPS of green sturgeon disperse from their Californian natal rivers and move northward along the continental shelf as adults. During this time they typically remain in the nearshore environment within the range of the 110 m depth contour and remain along the bottom of the seafloor. As such, they have a limited likelihood of exposure to non-impulsive acoustic stressors such as sonar at levels capable of eliciting a response.

Given the low abundance of ESA-listed green sturgeon within the Action Area and the limited likelihood of co-occurrence with other training and testing stressors, the likelihood of affecting green sturgeon is so low as to be discountable.

4.1.9 Critical Habitat for the Southern DPS of Green Sturgeon

On October 9, 2009, NMFS designated critical habitat for the Southern DPS of green sturgeon (74 FR 52300).

Critical habitat is designated in coastal U.S. marine waters within 60 fathoms depth from Monterey Bay, California (including Monterey Bay), north to Cape Flattery, Washington, including the Strait of Juan de Fuca, Washington, to its United States boundary; the Sacramento River, lower Feather River, and lower Yuba River in California; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; the lower Columbia River estuary; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor) and freshwater (USDC 2009b). For freshwater rivers north of and including the Eel River, the areas upstream of the head of the tide were not considered part of the geographical area occupied by the southern DPS.

Table 17 below delineates physical and biological features for southern green sturgeon.

Table 17. Physical or biological features of critical habitat designated for southern green sturgeon and corresponding species life history events.

Physical or Biological Features		Species Life History Event
Site Type	Site Attribute	
Freshwater riverine system	Food resources Migratory corridor Sediment quality Substrate type or size Water depth Water flow Water quality	Adult spawning Embryo incubation, growth and development Larval emergence, growth and development Juvenile metamorphosis, growth and development
Estuarine areas	Food resources Migratory corridor Sediment quality Water flow Water depth Water quality	Juvenile growth, development, seaward migration Subadult growth, development, seasonal holding, and movement between estuarine and marine areas Adult growth, development, seasonal holding, movements between estuarine and marine areas, upstream spawning movement, and seaward post-spawning movement
Coastal marine areas	Food resources Migratory corridor Water quality	Subadult growth and development, movement between estuarine and marine areas, and migration between marine areas Adult sexual maturation, growth and development, movements between estuarine and marine areas, migration between marine areas, and spawning migration

Several activities have been identified that threaten the PCEs in coastal bays and estuaries and necessitate the need for special management considerations or protection. The application of pesticides is likely to adversely affect prey resources and water quality within the bays and estuaries, as well as the growth and reproductive health of Southern DPS green sturgeon through bioaccumulation. Other activities of concern include those that disturb bottom substrates, adversely affect prey resources, or degrade water quality through re-suspension of contaminated sediments. Of particular concern are activities that affect prey resources. Prey resources are affected by: commercial shipping and activities generating point source pollution and non-point source pollution that discharge contaminants and result in bioaccumulation of contaminants in green sturgeon; disposal of dredged materials that bury prey resources; and bottom trawl fisheries that disturb the bottom (but result in beneficial or adverse effects on prey resources for green sturgeon). In addition, petroleum spills from commercial shipping and proposed hydrokinetic energy projects are likely to affect water quality or hinder the migration of green sturgeon along the coast (USDC 2009b).

4.1.9.1 *Conclusion*

Training and testing activities do not occur in freshwater habitats and will not affect critical habitat designated in these areas. Offshore areas of critical habitat and those at the mouth of Puget Sound co-occur with portions of the NWTT Action Area. The critical habitat's biological feature important to the life stages of green sturgeon that could be affected by NWTT activities are food resources. No changes in water quality, depth, flow, sediment quality, or impediment of migration corridors would be expected to occur. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Therefore, NWTT activities are not likely to adversely affect green sturgeon critical habitat.

4.1.10 Critical Habitat for ESA-listed Rockfish

On February 11, 2015, NMFS designated critical habitat for the threatened DPS of yelloweye rockfish (*Sebastes ruberrimus*), the threatened DPS of canary rockfish (*S. pinniger*), and the endangered DPS of bocaccio (*S. paucispinus*) (79 FR 68041). The specific areas in the final designation include 590.4 square miles (1,529 square km) of nearshore habitat in Puget Sound/Georgia Basin, Washington for canary rockfish and bocaccio, and 414.1 square miles (1,072.5 square km) of deepwater habitat in Puget Sound/Georgia Basin for yelloweye rockfish, canary rockfish and bocaccio.

Benthic habitats or sites deeper than 30 m (98ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for rockfishes to avoid predation, seek food and persist for decades. Several attributes of these sites determine the quality of the habitat. Physical or biological features essential to the conservation of adult canary rockfish and bocaccio, and adult and juvenile yelloweye rockfish include the following: 1) Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities, 2) water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities, and 3) the type and amount of structure and rugosity that supports feeding opportunities and predator avoidance (79 FR 68041).

Juvenile settlement habitats located in the nearshore with substrates such as sand, rock and/or cobble compositions that also support kelp (families Chordaceae, Alariaceae, Lessoniacea, Costariaceae, and Laminaricea) are essential for conservation because these features enable forage opportunities and refuge from predators and enable behavioral and physiological changes needed for juveniles to occupy deeper adult habitats. Several attributes of these sites determine the quality of the area. The physical or biological features essential to the conservation of juvenile canary rockfish and bocaccio include the following: 1) Quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities; and 2) water quality and sufficient levels of dissolved oxygen to support growth, survival, reproduction, and feeding opportunities (79 FR 68041).

Many forms of human activities have the potential to affect the essential features of listed rockfish species: (1) Nearshore development and in-water construction (e.g., beach armoring, pier construction, jetty or harbor construction, pile driving construction, residential and commercial construction); (2) dredging and disposal of dredged material; (3) pollution and runoff; (4) underwater construction and operation of alternative energy hydrokinetic projects (tidal or wave energy projects) and cable laying; (5) kelp harvest; (6) fisheries; (7) non-indigenous species introduction and management; (8) artificial habitats; (9) research activities; (10) aquaculture, and; (11) activities that lead to global climate change and ocean acidification. All of these activities may have an effect on one or more physical or biological features via their potential alteration of one or more of the following: adult habitats, food resources, juvenile settlement habitat, and water quality (79 FR 68041).

Section 4(a)(3) of the ESA precludes the Secretary from designating military lands as critical habitat if those lands are subject to an Integrated Natural Resource Management Plan (INRMP) under the Sikes Act that the Secretary certifies in writing benefits the listed species. NMFS consulted with the DOD and determined that there are several installations with INRMPs which overlap with marine habitats occupied by listed rockfish: (1) Joint base Lewis-McCord: (2) Manchester Fuel Department, (3) Naval Air Station Whidbey Island, (4) Naval Station Everett, and (5) Naval Station Kitsap. The final rule designating critical habitat for rockfish in Puget Sound/Georgia Basin excluded areas identified within INRMPs from critical habitat designation (79 FR 68041).

4.1.10.1 ***Conclusion***

Few training and testing activities occur in Puget Sound (primarily acoustic stressors) and would co-occur with portions of rockfish critical habitat. The critical habitat's biological feature important to the life stages of rockfish that could be affected by NWTT activities are food resources. No changes in water quality would be expected to occur. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Therefore, NWTT activities are not likely to adversely affect rockfish critical habitat.

4.1.11 Critical Habitat for Pacific Eulachon

On October 20, 2011, NMFS designated critical habitat for the southern DPS of Pacific eulachon. Critical habitat for eulachon includes portions of 16 rivers and streams in California, Oregon, and Washington (USDC 2011). All of these areas are designated as migration and spawning habitat for this species. In Oregon, 24.2 miles of the lower Umpqua River, 12.4 miles of the lower Sandy River, and 0.2 miles of Tenmile Creek have been designated. The mainstem Columbia River from the mouth to the base of Bonneville Dam, a distance of 143.2 miles, is also designated as critical habitat.

Table 18 delineates the designated physical or biological features for eulachon.

Table 18. Physical or biological features of critical habitats designated for eulachon and corresponding species life history events.

Physical or biological features		Species Life History Event
Site Type	Site Attribute	
Freshwater spawning and incubation	Flow Water quality Water temperature Substrate	Adult spawning Incubation
Freshwater migration	Flow Water quality Water temperature Food	Adult and larval mobility Larval feeding

The range of eulachon in the Pacific Northwest overlaps with the range of several ESA-listed stocks of salmon and steelhead as well as green sturgeon. Although the habitat requirements of these fishes differ somewhat from eulachon, efforts to protect habitat generally focus on the maintenance of watershed processes that would be expected to benefit eulachon. The Biological Review Team (BRT) identified dams and water diversions as moderate threats to eulachon in the Columbia and Klamath rivers where hydropower generation and flood control are major activities. Degraded water quality is common in some areas occupied by southern DPS eulachon. In the Columbia and Klamath systems, large-scale impoundment of water has increased winter water temperatures, potentially altering the water temperature during eulachon spawning periods (Gustafson et al. 2010). Numerous chemical contaminants are also present in spawning rivers, but the exact effect these compounds have on spawning and egg development is unknown (Gustafson et al. 2010). The BRT identified dredging as a low to moderate threat to eulachon in the Columbia River. Dredging during eulachon spawning would be particularly detrimental.

The lower Columbia River mainstem provides spawning and incubation sites, and a large migratory corridor to spawning areas in the tributaries. Prior to the construction of Bonneville Dam, eulachon ascended the Columbia River as far as Hood River, Oregon. Major tributaries that support spawning runs include the Grays, Skamokawa, Elochoman, Kalama, Lewis and Sandy rivers.

4.1.11.1 *Conclusion*

None of the NWTT activities occur in freshwater environments; therefore, no Eulachon critical habitat will be affected.

4.1.12 Critical Habitat for Puget Sound Chinook Salmon

The NMFS designated critical habitat for Puget Sound Chinook salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes portions of the Nooksack River, Skagit River, Sauk River, Stillaguamish River, Skykomish River, Snoqualmie River, Lake Washington, Green River, Puyallup River, White River, Nisqually River, Hamma Hamma River and other Hood Canal watersheds, the Dungeness/Elwha Watersheds, and nearshore marine areas of the Strait of Georgia, Puget Sound, Hood Canal, and the Strait of Juan de Fuca. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high water line is not defined the lateral extent is defined as the bank full elevation.

The designation for this ESU includes sites necessary to support one or more life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The PCEs of PS Chinook salmon critical habitat are:

- Freshwater spawning sites with water quantity and quality conditions and substrate that support spawning, incubation, and larval development;
- Freshwater rearing sites with (1) water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility, (2) water quality and forage that support juvenile development, and (3) natural cover such as shade, submerged and overhanging large wood, logjams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks;
- Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks that support juvenile and adult mobility and survival;

- Estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;

- Nearshore marine areas free of obstruction and excessive predation with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and

- Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

Of 49 subbasins reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur for this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load, and channel stability are common limiting factors in areas of critical habitat. Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, and water quality have also been degraded for adult spawning, embryo incubation, and rearing as a result of cumulative impacts of agriculture, forestry, and development.

Residential and commercial development has reduced the amount of functioning nearshore and estuarine habitat available for salmon rearing and migration (NMFS 2011b; SSPS 2007b). The nearshore marine habitat has been extensively altered and armored by industrial and residential development near the mouths of many of Puget Sound's tributaries. A railroad runs along large portions of the eastern shoreline of Puget Sound, eliminating natural cover along the shore and natural recruitment of beach sand (SSPS 2007b). The loss of mudflats, eelgrass meadows, and macroalgae further limits salmon foraging and rearing opportunities in nearshore and estuarine areas.

4.1.12.1 ***Conclusion***

Training and testing activities do not occur in freshwater habitats and will not affect critical habitat designated in these areas. Critical habitat in offshore areas and estuarine areas may co-occur with portions of the NWTT Action Area. The critical habitat's biological feature important to the life stages of chinook salmon that could be affected by NWTT activities are forage resources. No changes in water quality, quantity, or natural cover would be expected to occur. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Therefore, NWTT activities are not likely to adversely affect chinook salmon critical habitat.

4.1.13 Critical Habitat for Puget Sound Steelhead (Proposed)

Proposed critical habitat for PS steelhead (78 FR 2726, NMFS 2013) is almost identical to that for PS Chinook salmon, including habitat areas suitable for spawning, rearing, and unobstructed migration corridors to and from spawning and rearing habitats.

4.1.13.1 *Conclusion*

Training and testing activities do not occur in freshwater habitats and will not affect critical habitat designated in these areas. Critical habitat in offshore areas and estuarine areas may co-occur with portions of the NWTT Action Area. The critical habitat's biological feature important to the life stages of steelhead that could be affected by NWTT activities are forage resources. No changes in water quality, quantity, or natural cover would be expected to occur. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Therefore, NWTT activities are not likely to adversely affect steelhead critical habitat.

4.1.14 Critical Habitat for Hood Canal Summer Run Chum Salmon

NMFS designated critical habitat for Hood Canal summer-run chum salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes the Skokomish River, Hood Canal subbasin, which includes the Hamma Hamma and Dosewallips rivers and others, the Puget Sound subbasin, Dungeness/Elwha subbasin, and nearshore marine areas of Hood Canal and the Strait of Juan de Fuca from the line of extreme high tide to a depth of 30 meters. This includes a narrow nearshore zone from the extreme high-tide to mean lower low tide within several Navy security/restricted zones. Additionally, about 8 miles of habitat was unoccupied at the time it was designated, including Finch, Anderson and Chimacum Creeks (69 FR 74572; 70 FR 52630), but has recently been re-seeded. The designation for Hood Canal summer-run chum, like others made at this time, includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bank full elevation.

The designation for this ESU includes sites necessary to support one or more life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The PCEs of Hood Canal summer run chum salmon critical habitat are:

- Freshwater spawning sites with water quantity and quality conditions and substrate that support spawning, incubation, and larval development;
- Freshwater rearing sites with (1) water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility, (2) water quality and forage that support juvenile development, and (3) natural cover such as shade, submerged and overhanging large wood, logjams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks;
- Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks that support juvenile and adult mobility and survival;
- Estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;
- Nearshore marine areas free of obstruction and excessive predation with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and
- Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

Of 17 subbasins reviewed in NMFS's assessment of critical habitat for the Hood Canal chum salmon, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to the conservation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include degraded floodplain and mainstem river channel structure, degraded estuarine conditions and loss of estuarine habitat, riparian area degradation and loss of in-river wood in mainstem, excessive sediment in spawning gravels, and reduced stream flow in migration areas.

4.1.14.1 *Conclusion*

Training and testing activities do not occur in freshwater habitats and will not affect critical habitat designated in these areas. Critical habitat in offshore areas and estuarine areas may co-occur with portions of the NWTT Action Area. The critical habitat's biological feature important to the life stages of chum salmon that could be affected by NWTT activities are forage resources. No changes in water quality, quantity, or natural cover would be expected to occur. In the event acoustic stressors (or any other stressors) affect forage species, the impact is anticipated to be temporary and of short duration (only occurring during ensonification or activity duration) and of negligible magnitude (in terms of area size and proportion of available forage). Therefore, NWTT activities are not likely to adversely affect chum critical habitat.

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 NWTT 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.

More detailed background information on the status of these species and critical habitat can be found in a number of published documents including status reviews, recovery plans for the blue whale (NMFS 1998b), fin whales (NMFS 2010b), fin and sei whale (NMFS 1998a), humpback whale (NMFS 1991), sperm whale (NMFS 2010c), a status report on large whales prepared by Perry et al. (1999a) and the status review and recovery plan for the leatherback sea turtle (NMFS and USFWS 1998a; NMFS and USFWS 2007b). Richardson et al. (1995d) and Tyack (2000) provide detailed analyses of the functional aspects of cetacean communication and their responses to active sonar. Finally, Croll et al. (1999), NRC (2005; 2000; 2003b), Martin et al. (2015b), Supin et al. (2011), Tyack (2010), Filadelfo et al. (2009), and Richardson and Wursig (1995) provide information on the potential and probable effects of active sonar on the marine animals considered in this Opinion.

For Pacific salmon, steelhead, and other relevant species, NMFS commonly uses four parameters to assess the viability of the populations that, together, constitute the species: spatial structure, diversity, abundance, and productivity (McElhany et al. 2000b). These "viable salmonid population" (VSP) criteria therefore encompass the species' "reproduction, numbers, or distribution" as described in 50 CFR 402.02. When these parameters are collectively at appropriate levels, they maintain a population's capacity to adapt to various environmental conditions and allow it to sustain itself in the natural environment. These attributes are influenced by survival, behavior, and experiences throughout a species' entire life cycle, and these characteristics, in turn, are influenced by habitat and other environmental conditions.

"Spatial structure" refers both to the spatial distributions of individuals in the population and the processes that generate that distribution. A population's spatial structure depends fundamentally on habitat quality and spatial configuration and the dynamics and dispersal characteristics of individuals in the population. "Diversity" refers to the distribution of traits within and among populations. These range in scale from DNA sequence variation at single genes to complex life history traits (McElhany et al. 2000b). "Abundance" generally refers to the number of naturally-produced adults (i.e., the progeny of naturally-spawning parents) in the natural environment (e.g., on spawning grounds). "Productivity," as applied to viability factors, refers to the entire life cycle; i.e., the number of naturally-spawning adults produced per parent. When progeny replace or exceed the number of parents, a population is stable or increasing. When progeny fail to replace the number of parents, the population is declining. McElhany et al. (2000b) use the terms "population growth rate" and "productivity" interchangeably when referring to production over the entire life cycle. They also refer to "trend in abundance," which is the manifestation of long-term population growth rate.

Many of the 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 2015). ESA take prohibitions do not apply to hatchery fish with clipped adipose fins from threatened ESUs/DPSs.

For species with multiple populations, once the biological status of a species' populations has been determined, NMFS assesses the status of the entire species using criteria for groups of populations, as described in recovery plans and guidance documents from technical recovery teams. Considerations for species viability include having multiple populations that are viable, ensuring that populations with unique life histories and phenotypes are viable, and that some viable populations are both widespread to avoid concurrent extinctions from mass catastrophes and spatially close to allow functioning as metapopulations (McElhany et al. 2000b).

4.2.1 Blue Whale

The blue whale, *Balaenoptera musculus* (Linnæus 1758), 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 1998b).

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 1998). 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 2003b). 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).

Blue whales appear to migrate to waters offshore of Washington, Oregon, and northern California to forage. Thus far, blue whales are associated with deeper, pelagic waters in the Action Area; they have not been reported to occur proximate to the coast or in Puget Sound itself. Širović et al. (Širović et al. 2012; Širović et al. 2011) reported detection of blue whale vocalizations in the offshore waters of Washington from late fall through February. Oleson and Hildebrand (2011) also reported visual blue whale sightings in these areas. Although a resident population of blue whales might occur off the coast of Vancouver Island throughout the year (Burtenshaw et al. 2004a), most blue whales that occur in the Action Area for this consultation appear to migrate between summer, foraging areas and winter rearing areas along the Pacific Coast of the United States. That seasonal migration brings them to waters off the NWTRC (with some individuals continuing north to the Gulf of Alaska) during the warm, summer season with a southward migration to waters off California, south to Central America, during the winter season (Calambokidis et al. 2009b; Gregr et al. 2000; Mate et al. 1998).

4.2.1.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

The U.S. west coast is known to be a feeding area for blue whales during summer and fall (Bailey et al. 2010; Calambokidis et al. 2009a), although primary occurrence for this species is south of 44° N (Forney et al. 2012; Hamilton et al. 2009; Sirovic et al. 2015). Blue whales feed in the area as late as October, although fewer individuals are seen because the majority of the population migrates south. Acoustic data collected by Sound Surveillance System hydrophones reveal that males call during the fall in this area (Stafford et al. 2001a). More recently, Navy-funded acoustic monitoring studies have detected blue whales along the Washington coast between August and February, with peak calling from October to December, and no detections between April and July (Širović et al. 2012; Širović et al. 2011). An individual blue whale was also sighted off Washington in January 2009, in waters approximately 3,281 ft. (1,000 m) deep (Oleson et al. 2012).

A recent study (Mate et al. 2015) was completed to determine movement patterns, occurrence, and residence times of blue and fin whales within Navy training and testing areas along the U.S. West Coast. Tagging occurred off the coast of southern California in August and September 2014. The study concluded that there was a great deal of individual variation in the tracks of both blue and fin whales, with blue whale locations extending from the northern tip of Vancouver Island in British Columbia to the Costa Rica Dome area off Central America, and those of fin whales extending from southern Oregon to central Baja California. The area of highest use for blue whales was between Point Dume and Mugu Canyon off southern California, out to approximately 30 kilometers (km) from shore. For fin whales, areas of highest use occurred south of San Miguel Island, approximately 100 km offshore, and approximately 70 to 80 km offshore along the south-central California coast between Arroyo Grande and Big Sur.

Mate et al. (2015) indicated that both blue and fin whales were tracked in the NWTRC and the Southern California Training Range Complex (SOCAL), but neither species traveled into the Gulf of Alaska Temporary Maritime Activities Area. Only one blue whale had locations within area W237 within the NWTRC area (in August, September, and October). Blue whale presence was observed in both SOCAL and NWTRC in August, September, October, and November. Fin whale locations occurred in SOCAL in all 5 months in which they were tracked (August, September, October, November, and December), but in only 2 months in NWTRC (August and September). Eighteen blue whales spent 1 to 48 percent (37 to 414 hours [h]) of their tracking periods within the SOCAL, while four blue whales spent 3 to 45 percent (15 to 1,249 h) of their tracking periods within the NWTRC. Finally, Mate et al. (2015) concluded that four fin whales spent from less than 1 to 39 percent (1 to 956 h) of their total tracking periods in SOCAL, and one fin whale spent 51 percent (811 h) of its tracking period in NWTRC.

4.2.1.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Blue whales are not expected to occur within the Inland Waters region of the Action Area since it is well inland of the areas normally inhabited by blue whales.

4.2.1.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Blue whales are not expected to occur within the SEAFAC region of the Action Area since it is well inland of the areas normally inhabited by blue whales.

4.2.1.2 ***Population Structure***

For this and all subsequent species, 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 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.

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), but 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 (Gilpatrick et al. (1997), Barlow et al. (1995), 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. 1997; 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 2003b). 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 2003a; Stafford et al. 2007; 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 2003a; Stafford et al. 2007; 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 2003a). 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 2003b). We assume that this population is the one affected by the activities considered in this Opinion.

4.2.1.3 ***Abundance Estimate***

Widespread whaling over the last century is believed to have decreased the blue whale population to approximately 1 percent of its pre-whaling population size (Sirovic et al. 2004); (Branch et al. 2007). The current best available abundance estimate for the eastern North Pacific population of blue whales that occur off California, Oregon, and Washington is 2,138 (95 percent CI: 1,774 to 2,584 (Monnahan et al. 2014b). Based on ship surveys conducted in the summer and fall from 1991 to 2008, it is estimated that 58 blue whales (CV = 0.41) occur in waters off Washington and Oregon (Barlow 2010a). There was a documented increase in the blue whale population size between 1979 and 80 and 1991 (Barlow 1994) and between 1991 and 1996 (Barlow 1997), but there has not been evidence to suggest an increase in the population of the eastern North Pacific stock since then (Barlow and Taylor 2001a; Carretta et al. 2012). Based on line-transect surveys conducted off California between 1991 and 2005, the abundance estimates of blue whales declined in these waters over the survey period (Barlow and Forney 2007). However, this apparent decline was likely due to variability in the distribution patterns of blue whales off the coast of North America rather than a true population decline (Calambokidis et al. 2009a). 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. 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 and sharks are also known to attack, injure, and kill very young or sick fin and humpback whales and likely hunt blue whales as well (Perry et al. 1999a) (Ford and Reeves 2008).

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 194, 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). Within the St. Lawrence Estuary, blue whales are believed to be affected by large amounts of recreational and commercial vessel traffic. Blue whales in the St. Lawrence appeared more likely to react to these vessels when boats made fast, erratic approaches or sudden changes in direction or speed (Edds and Macfarlane 1987).

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. 1997; 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. 1997; 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. 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 off California, Oregon, and Washington is 2,138 (95 percent CI: 1,774 to 2,584 (Monnahan et al. 2014b). 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. The results reflect an upward trend in abundance of blue whales along the U.S. West Coast, although their numbers are highly variable off California, most likely due to the variability of its use as a feeding area (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. (2006b) 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 (1997a) 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. 2006a; 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).

In Southern California, blues whales produce two predominant call types: Type B and D. B-calls are stereotypic of the blue whale population found in the eastern North Pacific (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 (Oleson et al. 2007c). In the SOCAL Range Complex region, D calls are produced in highest numbers during the late spring and early summer, and in diminished numbers during the fall, when A-B song dominates blue whale calling (Hildebrand et al. 2011; Hildebrand et al. 2012; Oleson et al. 2007d).

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. 2004b). 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 1997b; 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.

4.2.1.9 ***Critical Habitat***

NMFS has not designated critical habitat for blue whales.

4.2.2 Fin Whale

The fin whale, *Balaenoptera physalus* (Linnæus 1758), is a well-defined, cosmopolitan species of baleen whale (Gambell 1985a). 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).

4.2.2.1 ***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.

4.2.2.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

This species has been documented from 60° N to 23° N, and they have frequently been recorded in waters offshore Oregon and Washington (Barlow and Forney 2007). Based on predictive habitat models derived from line-transect survey data collected between 1991 and 2008 off the U.S. west coast, relatively high densities of fin whales are predicted off Washington during the summer and fall (Barlow et al. 2009a); (Becker et al. 2012b); (Forney et al. 2012). During visual surveys conducted from August 2004 to September 2008, there was a single sighting of two fin whales off the Washington coast in December 2005, in waters approximately 3,281 ft. (1,000 m) deep (Oleson et al. 2009).

Navy-funded offshore passive acoustic monitoring off Washington from 2004 to 2013 has reported fin whales as the most commonly detected baleen whale call type detected with peak calling in winter and spring, and low calling in summer (Kerosky et al. 2013); (Sirovic et al. 2012); U.S. Department of the Navy 2013a). Fin whale calls were detected on more than 90 percent of the days during the months of October, December, January, and February, but were not detected in either May or June (Sirovic et al. 2012). Between May 2010 and May 2013, 11 fin whales were tagged with satellite tracking tags off of Washington. In general, fin whales were most commonly using waters associated with the outer shelf edge (median distance to shore: 72 kilometers [km]) (DoN 2013c; Schorr et al. 2013).

4.2.2.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Fin whales are extremely rare within the Inland Waters. Strandings reported within Puget Sound have all been individuals struck by non-Navy ships, and they presumably were carried on the bow into the sound (Norman et al. 2004a).

4.2.2.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Fin whales were observed seven times in the summer during surveys of the inland waters of southeast Alaska from 1991 to 2007 (Dahlheim et al. 2009). Given the limited number of sightings in inland waters and their more pelagic nature, fin whales are considered rare in the SEAFAC region of the Action Area.

4.2.2.2 ***Population Structure***

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. Globally, fin whales are sub-divided into three major groups: Atlantic, Pacific, and Antarctic. Within these major areas, different organizations use different population structure.

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. Based on genetic analyses, Berube et al. (1998) concluded that fin whales in the Sea of Cortez represent an isolated population that has very little genetic exchange with other populations in the North Pacific Ocean (although the geographic distribution of this population and other populations can overlap seasonally). They also concluded that fin whales in the Gulf of St. Lawrence and Gulf of Maine are distinct from fin whales found off Spain and in the Mediterranean Sea.

Regardless of how different authors structure the fin whale population, mark-recapture studies have demonstrated that individual fin whales migrate between management units (Mitchell 1974; Sigurjonsson et al. 1989), which suggests that these management units are not geographically isolated populations.

Mizroch et al. (1984) identified five fin whale “feeding aggregations” in the Pacific Ocean: (1) an eastern group that move along the Aleutians, (2) a western group that move along the Aleutians (Berzin and Rovnin 1966; Nasu 1974); (3) an East China Sea group; (4) a group that moves north and south along the west coast of North America between California and the Gulf of Alaska (Rice 1974); and (5) a group centered in the Sea of Cortez (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.

Sighting data show no evidence of migration between the Sea of Cortez and adjacent areas in the Pacific, but seasonal changes in abundance in the Sea of Cortez suggests that these fin whales might not be isolated (Tershy et al. 1993). Nevertheless, Bérubé et al. (2002) concluded that the Sea of Cortez fin whale population is genetically distinct from the oceanic population and have lower genetic diversity, which suggests that these fin whales might represent an isolated population.

Fin whales also appear to migrate to waters offshore of Washington, Oregon, and northern California to forage. Most fin whales that occur in the Action Area for this consultation appear to migrate between summer, foraging areas and winter rearing areas along the Pacific Coast of the United States. Širović et al. (Širović et al. 2012; Širović et al. 2011) reported fin whale vocalizations off Washington from July through April with calls not detected from May to July. Moore et al. (1998) recorded fin whale vocalizations in waters off Washington and Oregon throughout the year, with concentrations between September and February, which demonstrates that fin whales are likely to occur in the Action Area throughout the year.

4.2.2.3 ***Abundance Estimate***

Currently there are no reliable population estimates for the Alaska/Northeast Pacific stock of fin whales. A minimum estimate for the stock is 1,214, based on surveys west of the Kenai Peninsula which covered only a portion of the stock's range (Allen and Angliss 2014). Zerbini et al. (2006a) have provided evidence of an increasing abundance trend for fin whales in Alaskan waters.

The current best available abundance estimate of fin whales in California, Oregon, and Washington waters is 3,051 (CV = 0.18) (Carretta et al. 2014). Based on ship surveys conducted in the summer and fall from 1991 to 2008, it is estimated that 416 fin whales (CV = 0.28) occur in waters off Washington and Oregon (Barlow 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). 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.

4.2.2.4 ***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. 1999a).

4.2.2.5 ***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. 2008c; 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. 2008c). 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).

Management measures aimed at reducing the risk of ships hitting right whales should also reduce the risk of collisions with fin whales. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 27 percent in the Bay of Fundy region.

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. 1997). 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).

4.2.2.6 ***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. 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 most recent abundance estimates for fin whales that we are aware of are 16,625 individuals in the North Pacific Ocean and 119,000 individuals worldwide (Braham 1991). 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 2011i).

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. 1999a). 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 2003b; Barlow and Taylor 2001b; Forney 2007). Barlow (2003b) noted densities of up to 0.0012 individuals/km^2 off Oregon and Washington and up to 0.004 individuals/km^2 off California.

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.

The current best available abundance estimate for the Hawaiian stock of fin whales is 174 (coefficient of variation = 0.72) (Barlow 2003a). The current best available abundance estimate of fin whales in California, Oregon, and Washington waters is 3,051 (CV = 0.18) (Carretta et al. 2014). Survey estimate numbers for both 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).

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.

4.2.2.7 ***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. Barlow (2003b) reported mean group sizes of 1.1 to 4.0 during surveys off California, Oregon, and Washington.

4.2.2.8 ***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 (1997a) 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 1997b; 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 2012b). 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 1997b). Several fin whales were tagged during the Southern California-10 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). 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. 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).

4.2.2.9 *Critical Habitat*

NMFS has not designated critical habitat for fin whales.

4.2.3 Humpback Whale

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 April 21, 2015, NMFS proposed to divide the globally listed endangered humpback whale population into 14 DPSs (80 FR 22304).

4.2.3.1 ***Distribution***

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.

4.2.3.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

The California, Oregon, and Washington stock of humpback whales uses the waters off the west coast of the United States as a summer feeding ground. They are present off the northern California coast mainly between April and December and off the Oregon and Washington coasts mainly from May through November (Dohl et al. 1983); (Green et al. 1992a); (Forney and Barlow 1998); (Calambokidis et al. 2004a); (2010). Visual surveys and acoustic monitoring studies have detected humpbacks along the Washington coast year-round, with peak occurrence during the summer and fall (Oleson et al. 2009). Consistent with previous recordings from two Navy-funded offshore passive acoustic monitoring devices (Kerosky et al. 2013; Oleson and Hildebrand 2012), humpback whales were most commonly detected in acoustic recordings between September and December, which is also the peak time for humpback whale singing (Kerosky et al. 2013). Lower levels of humpback whale calling were also detected from February through May (Kerosky et al. 2013; Oleson and Hildebrand 2012; Oleson et al. 2009). Visual and acoustic detections of humpback whales in this area do not fully overlap, as most visual sightings occur during the summer and early fall (Oleson et al. 2009), which is likely the result of the strong seasonal variation in humpback whale singing and other vocal behavior (Kerosky et al. 2013; Oleson and Hildebrand 2012). Photo-identification studies suggest that whales feeding in this region are part of a small sub-population that primarily feeds from central Washington to southern Vancouver Island (Calambokidis et al. 2004a); (Calambokidis et al. 2008b). Whales appear to range broadly throughout the continental shelf waters, with significant seasonal trends in distribution; however, detailed knowledge of habitat use and individual residency patterns while in this feeding area cannot be determined easily through visual surveys alone (Schorr et al. 2013). In winter and spring (roughly January through March), most whales are south on their breeding grounds and are likely not as abundant in this region of the Action Area during these times.

Off the U.S. west coast, humpback whales are more abundant in shelf and slope waters (< 6,562 ft. [2,000 m] deep) and are often associated with areas of high productivity (Becker et al. 2012b; Becker et al. 2010; Forney et al. 2012). Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992a; Tynan et al. 2005a). Off Washington, higher concentrations have been reported between Juan de Fuca Canyon and the outer edge of the shelf break in a region called "the Prairie," near Barkley and Nitnat Canyons, and near Swiftsure Bank (Calambokidis et al. 2004a). Five humpback whales were satellite tagged off Washington between May 2010 and May 2013. Although tag durations were short with a median duration of 7 days, tag tracks showed all five whales using both shelf and slope waters as well as some underwater canyons such as the Juan de Fuca Canyon (one of five whales) (DoN 2013c; Schorr et al. 2013).

Humpback whales feeding off the coast of the west coast of the United States are likely to belong to either the proposed Mexico DPS or the Central America DPS.

4.2.3.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Although humpback whales were common in inland Washington waters prior to the whaling period, few sightings had been reported in this area until the last 10 years (Calambokidis and Steiger 1990; Pinnell and Sandilands 2004; Scheffer and Slipp 1948b). More recently, with the creation (in 2011) of the Orca Network online forum to compile whale sighting reports, and increased public interest in reporting whale sightings, the number of humpback whale sightings in inland waters has increased. Inland water opportunistic sightings primarily occur from April through July, but sightings are reported in every month of the year. Most sightings occur in the Strait of Juan de Fuca and in the San Juan Island area, with only occasional sightings in Puget Sound.

In Puget Sound (defined as south of Admiralty Inlet), Calambokidis et al. (2002) recorded only six individuals between 1996 and 2001. However, from January 2003 through July 2012 there were over 60 sightings of humpback whales reported to Orca Network, some of which could be the same individuals (Orca Network 2012). A review of the reported sightings in Puget Sound indicates that humpback whales usually occur as individuals or in pairs (Orca Network 2012).

Sightings of humpback whales in Puget Sound vary by location but are infrequent. From the Rich Passage to Agate Passage area in the vicinity of (NAVBASE Kitsap Bremerton and Keyport, only one unverified sighting of a humpback whale was reported to Orca Network (Orca Network 2012) from January 2003 through July 2012. In Hood Canal and Dabob Bay (where NAVBASE Kitsap Bangor and the Dabob Bay Range Complex [DBRC] are located, respectively), one humpback whale was observed for several weeks in January and February 2012. Prior to this sighting, there were no confirmed reports of humpback whales entering Hood Canal or Dabob Bay. In the Saratoga Passage area (between NAVSTA Everett and NASWI), one humpback whale was reported in Penn Cove south of Crescent Harbor in July 2008. This is the only humpback report from January 2003 through September 2012 that was considered a likely positive identification (Orca Network 2012). There have been no verified humpback sightings in the Carr Inlet area between January 2003 and July 2012. Two unverified sightings were reported to Orca Network to the north of Carr Inlet, near Point Defiance, Tacoma, over the same time period. The last verified sighting was in June and July of 1988 when two individually identified juvenile humpback whales were observed traveling throughout the waters of southern Puget Sound for several weeks (Calambokidis and Steiger 1990).

Given their general migration patterns, this species is rare in the inland waters, but is expected to be more likely to occur in the warmer months (May through November). However, it is not expected to be present in all areas, nor remain for long time periods.

4.2.3.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

In summer, relatively high densities of humpback whales occur throughout much of southeast Alaska (Allen and Angliss 2012). Because this species makes extensive use of inland coastal waters, it is the large whale species most likely to be found in the Southeast Alaska area. Humpback whales are commonly sighted in Ernest Sound (north of the SEAFAC) and near the mouth of Boca de Quadra (south of SEAFAC), but specific data are lacking (DoN 1991). Although specific data are lacking, it is likely that humpback whales occasionally use the Behm Canal heading to Gedney Pass (DoN 1991). Humpback whales were observed frequently during the 1991 to 2007 surveys (spring through fall) of the inland waters of southeast Alaska (Dahlheim et al. 2009). Although surveys were not conducted in the winter months in southeast Alaska, observations have been made of humpback whales that have not migrated south, but remained in Alaskan waters to feed (Moran et al. 2009).

Humpback whales feeding in the Western Behm Canal belong to the proposed Hawaii DPS[6].

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[6] Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. On April 21, 2015, NMFS proposed to divide the globally listed

4.2.3.2 ***Population Structure***

Descriptions of the population structure of humpback whales differ depending on whether an author focuses on where humpback whales winter or where they feed. During winter months in northern or southern hemispheres, adult humpback whales migrate to specific areas in warmer, tropical waters to reproduce and give birth to calves. During summer months, humpback whales migrate to specific areas in northern temperate or sub-arctic waters to forage. In summer months, humpback whales from different "reproductive areas" will congregate to feed; in the winter months, whales will migrate from different foraging areas to a single wintering area. In either case, humpback whales appear to form "open" populations; that is, populations that are connected through the movement of individual animals.

Based on genetic and photo-identification studies, the NMFS currently recognizes four stocks, likely corresponding to populations, of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (Hill and DeMaster 1998). However, gene flow between them may exist. Humpback whales summer in 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 (Johnson and Wolman 1984; Nemoto 1957; Tomilin 1967). These whales migrate to Hawai′i, southern Japan, the Mariana Islands, and Mexico during winter. However, more northerly penetrations in Arctic waters occur on occasion (Hashagen et al. 2009). The central North Pacific population winters in the waters around Hawai′i while the eastern North Pacific population (also called the California-Oregon-Washington-Mexico stock) winters along Central America and Mexico. However, Calambokidis et al. (1997) identified individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure. Herman (1979) presented extensive evidence that humpback whales associated with the main Hawaiian Islands immigrated there only in the past 200 years. Winn and Reichley (1985) identified genetic exchange between the humpback whales that winter off Hawaii and Mexico (with further mixing on feeding areas in Alaska) and suggested that humpback whales that winter in Hawai′i may have emigrated from Mexican wintering areas. A "population" of humpback whales winters in the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands, with occurrence in the Mariana Islands, at Guam, Rota, and Saipan from January to

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endangered humpback whale population into 14 DPSs (80 FR 22304). The humpback whales in the Action Area potentially belong to one of three proposed DPSs: the non-ESA-listed Hawaii DPS, the non-ESA-listed Mexico DPS, or the threatened Central America DPS. All three of these DPSs may feed seasonally in the Action Area.

March (Darling and Cerchio 1993; Eldredge 1991; Eldredge 2003; Rice 1998). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed (Angliss and Outlaw 2008; Calambokidis 1997; Calambokidis et al. 2001).

Separate feeding groups of humpback whales are thought to inhabit western U.S. and Canadian waters, with the boundary between them located roughly at the U.S./Canadian border. The southern feeding ground ranges between 32° to 48°N, with limited interchange with areas north of Washington State (Calambokidis et al. 2004b; Calambokidis et al. 1996). Humpback whales feed along the coasts of Oregon and Washington from May-November, with peak numbers reported May-September, when they are the most commonly reported large cetacean in the region (Calambokidis and Chandler. 2000; Calambokidis et al. 2004b; Dohl 1983; Green et al. 1992b). Off Washington State, humpback whales concentrate between Juan de Fuca Canyon and the outer edge of the shelf break in a region called "the Prairie," near Barkley and Nitnat canyons, in the Blanco upwelling zone, and near Swiftsure Bank (Calambokidis et al. 2004b). Humpback whales also tend to congregate near Heceta Bank off the coast of Oregon (Green et al. 1992b). Additional data suggest that further subdivisions in feeding groups may exist, with up to six feeding groups present between Kamchatka and southern California (Witteveen et al. 2009).

Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992b; Tynan et al. 2005b). Although humpback whales were common in inland Washington State waters in the early 1900s, severe hunting throughout the eastern North Pacific has diminished their numbers and few recent inshore sightings have been made (Calambokidis et al. 1990; Scheffer and Slipp 1948a).

Historically, humpback whales occurred in Puget Sound. Since the 1970s, however, humpback whales have become rare within Puget Sound, although at least five humpback whales have been observed in Puget Sound since 1976 (Calambokidis et al. 1990; Calambokidis et al. 2004b; Osborne et al. 1988a). Because of their contemporary rarity in Puget Sound, we assume that humpback whales would not be exposed to Navy training activities within the Sound itself, but would be exposed in waters offshore of Washington.

Although humpback whales no longer appear to occur regularly in Puget Sound, they have consistently been more common than any other large cetacean observed off the coast of Washington State for more than a decade (Calambokidis et al. 2009b; Calambokidis et al. 2004b; Forney 2007). Humpback whales occur in those waters seasonally from May through November, becoming fairly common beginning in July, and reaching peak densities from August to September with density declining substantially from September onward (Calambokidis 1997; Calambokidis and Chandler. 2000; Calambokidis et al. 2001; Calambokidis et al. 1997; Green et al. 1992b). During that time interval, humpback whales have been reported in coastal waters, on the continental shelf, and the continental slope, with concentrations occurring in steep slope water near Grays, Astoria, and Nitinat canyons (Forney 2007; Green et al. 1992b).

Several authors have reported that humpback whales do not occur off the coasts of Washington and Oregon in the winter (Green et al. 1992b). However, Shelden et al. (Shelden et al. 2000) reported observations of humpback whales north and south of Juan de Fuca canyon (off northern Washington) in late December. These authors also reported that humpback whales were common in Georgia Strait during the winter in the early 1900s and they suggested that, as their population increases, humpback whales might be re-occupying areas they had previously abandoned after their populations were decimated by whalers; these authors also allowed that humpback whales might remain in waters off Washington when their prey is abundant late in the year.

4.2.3.3 *Abundance Estimate*

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. 2008b). 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). Data indicate that the North Pacific population has been increasing at a rate of between 5.5 percent and 6.0 percent per year, approximately doubling every 10 years (Calambokidis et al. 2008b). Baker et al. (2013) commented that humpback whales display a complex population structure based on DNA samples taken from 10 humpback whale feeding and eight breeding regions within the Pacific.

The Central North Pacific stock has been estimated at 10,103 individuals based on data from their wintering grounds throughout the main Hawaiian Islands (Allen and Angliss 2014). In summer, the majority of humpback whales from the Central North Pacific stock are found in the Aleutian Islands, Bering Sea, Gulf of Alaska, and southeast Alaska/northern British Columbia, where relatively high densities of whales occur (Allen and Angliss 2012). There is a high rate of interchange between whales found in southeast Alaska and northern British Columbia, and based on data from both inshore and offshore waters in these regions, abundance estimates range from 2,883 to 6,414 animals (Calambokidis et al. 2008b).

The current best estimate for the California, Oregon, and Washington stock is 1,918 (CV = 0.03) (Carretta et al. 2014). Based on ship surveys conducted in the summer and fall from 1991 to 2008, it is estimated that 260 humpback whales (CV = 0.32) occur in waters off Washington and Oregon (Barlow 2010b). Abundance estimates derived from finer scale ship survey data collected off the northern Washington coast between 1995 and 2002 yielded line-transect estimates that ranged from 85 to 562 animals (CV = 0.21 to 0.33) and capture-recapture estimates that ranged from 89 to 343 animals (CV = 0.19 to 0.36) (Calambokidis et al. 2004a). Both of the high estimates were for 2002 when sighting rates and corresponding abundance estimates increased dramatically. Without the 2002 data, estimates ranged from 85 to 125 and 80 to 230 animals based on line-transect and capture-recapture methods, respectively. Carretta et al. (2014) estimate the Northern Washington/Southern British Columbia stock at 189 individuals.

The current best estimated abundance of the proposed DPSs are 12,000 for the Hawaii DPS; 6,000 to 7,000 for the Mexico DPS; and 600 for the Central America DPS (80 CFR 22303).

4.2.3.4 ***Natural 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. 1999a). 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.

4.2.3.5 ***Anthropogenic Threats***

Three human activities are known to threaten humpback whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for listing several species as endangered.

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). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al. 2005; Nelson et al. 2007). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. 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. 1997). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1 percent) showed evidence of collisions with ships (Laist et al. 2001). Between 1999 and 2005, there were 18 reports of humpback whales being struck by vessels along the Atlantic coast of the U.S. and the Maritime Provinces of Canada (Cole et al. 2005; Nelson et al. 2007). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by 9 percent.

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (Gauthier et al. 1997). 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 in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, 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). Contaminant levels are relatively high in humpback whales as compared to blue whales. Humpback whales feed higher on the food chain, where prey carry higher contaminant loads than the krill that blue whales feed on.

4.2.3.6 ***Status and Trends***

Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. On April 21, 2015, NMFS proposed to divide the globally listed endangered humpback whale population into 14 DPSs (80 FR 22304). The humpback whales in the Action Area potentially belong to one of three proposed DPSs: the non-ESA-listed Hawaii DPS, the non-ESA-listed Mexico DPS, or the threatened Central America DPS. All three of these DPSs may feed seasonally in the Action Area.

In the North Pacific the pre-exploitation population size may have been as many as 15,000 humpback whales, and current estimates are 6,000 to 8,000 whales (Calambokidis et al. 2009b; Rice 1978). It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whale to roughly 1,000 (Perry et al. 1999a). Population estimates have risen over time from 1,407 to 2,100 in the 1980s to 6,010 in 1997 (Baker 1985; Baker and Herman 1987; Calambokidis et al. 1997; Darling and Morowitz 1986). Based on surveys between 2004 and 2006, Calambokidis et al. (2008a) estimated that the number of humpback whales in the North Pacific consisted of about 18,300 whales, not counting calves. Because estimates vary by methodology, they are not directly comparable and it is not clear which of these estimates is more accurate or if the change from 1,407 to 18,300 is the result of a real increase or an artifact of model assumptions. Tentative estimates of the eastern North Pacific stock suggest an increase of 6 to 7 percent annually, but fluctuations have included negative growth in the recent past (Angliss and Outlaw 2005).

4.2.3.7 ***Diving***

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).

4.2.3.8 ***Social Behavior***

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.

4.2.3.9 ***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. 2006b; 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. 2006b; 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[7]) (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. (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. (2006a) 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

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[7] *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.*

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 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).

Humpback whales are the most abundant ESA-listed species observed during Navy visual surveys in the winter months. To date, humpback whales have been documented as the species which has received the highest sound pressure levels from US Navy mid-frequency active sonar (MFAS) training (i.e. at least 183 dB re 1microPa) based upon an analysis which utilized shipboard Marine Mammal Observer sightings on February 18, 2011 (Farak et al. 2011), combined with PMRF range hydrophone data (Martin and Manzano-Roth 2012). Analysis of PMRF range hydrophone data for purpose of estimating received levels on marine mammals has also been done in conjunction with satellite tagged animals (Baird et al. 2014) and aerial focal follows (Mobley Jr. and Pacini 2013). Passive acoustic monitoring (PAM) of PMRF hydrophones during Navy training for the month of Feb from 2011 to 2013 has shown that acoustically localized minke whales are reduced during periods involving MFAS training activity when compared to other periods of time (Martin et al. 2014) . PAM monitoring for beaked whale foraging dives at PMRF has also shown reduced foraging dive rates during periods of MFAS training with estimated receive levels on the group dive members (mean levels of 109 dB re 1 microPa) in February of 2012 (Manzano-Roth et al. 2013a). Acoustic analysis has also shown that marine mammals near the sea surface can be exposed to higher estimated receive levels due to ducted propagation, that typically exists at PMRF. Analysis of behaviors observed during one focal follow taken during aerial surveys, in conjunction with estimated received levels using PMRF passive acoustic data products, are reported as a case study of a single focal follow on a humpback whale in the vicinity of MFAS (Mobley Jr. and Pacini 2013).

4.2.3.10 *Critical Habitat*

NMFS has not designated critical habitat for humpback whales.

4.2.4 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 (12-18 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 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.

4.2.4.1 ***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. 1999a). 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. 2004). 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. 1999a). 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.

4.2.4.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

Sei whales are distributed offshore in waters off the U.S. west coast (Carretta et al. 2012). They are generally found feeding along the California Current, preferring deep water habitat along the continental shelf break (Perry et al. 1999a). During six systematic ship surveys conducted between 1991 and 2008 in waters off the U.S. west coast to approximately 300 nm offshore, there were a total of 10 sei whale sightings, four of which were in waters off Oregon and Washington (Barlow 2010b). There were no sei whale sightings during more coastal (out to about the 660 ft. [200 m] isobath) ship surveys off the northern Washington coast between 1995 and 2002 (Calambokidis et al. 2004a). Sei whales migrate into the Action area in summer and fall and are found farther south in winter (Matsuoka et al. 2013b; Perry et al. 1999b).

4.2.4.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Sei whales are considered rare in the Inland Waters including Puget Sound. A sei whale washed ashore west of Port Angeles in the Strait of Juan de Fuca during September 2003 (Preston 2003), but this is considered an unusual event.

4.2.4.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Sei whales are not expected to occur within the SEAFAC region of the Action Area since it is well inland of the areas normally inhabited by sei whales.

4.2.4.2 *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 1998). 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. 1999a).

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. Based on a sei whale that stranded near Port Angeles and the sei whales observed by Forney and her co-workers (Forney

2007), we know that these whales still occur in waters off Washington, Oregon, and northern California.

4.2.4.3 *Abundance Estimate*

In the North Pacific, the pre-exploitation sei whale population was estimated at 42,000 whales (Tillman 1977a). The most current population estimate for sei whales in the entire north Pacific is 9,110 (Calambokidis et al. 2008a) and 25,000 individuals worldwide (Braham 1991). 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) Carretta et al. (2014). Based on ship surveys conducted in the summer and fall from 1991 to 2008, it is estimated that 52 sei whales (CV = 0.62) occur in waters off Washington and Oregon (Barlow 2010b). 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).

4.2.4.4 *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).

4.2.4.5 *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. 1997). 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. 1997). 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). Two of these ship strikes were reported as having resulted in death. One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Waring et al. 2009). New rules for seasonal (June through December) slowing of vessel traffic in the Bay of Fundy to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to reduce sei whale ship strike mortality by 17 percent.

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.

4.2.4.6 ***Status and Trends***

The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973.

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. 1999a). 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. 2003).

4.2.4.7 ***Diving***

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).

4.2.4.8 ***Social Behavior***

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 1986). The dominant food for sei whales off California during June-August is northern anchovy, while in September-October whales feed primarily on krill (Rice 1977). 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.

4.2.4.9 ***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. Norris et al. (2012) reported that simultaneous acoustic detections of called were made from the towed array during three visual sightings. The encounters occurred primarily in the central and southern region of the study area, ranging from the island of Tinian to the southeast corner of the study area. A higher concentration was found in the southeast corner and along the Mariana Trench (Norris et al. 2012).

While no data on hearing ability for this species are available, Ketten (1997a) 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.

4.2.4.10 *Critical Habitat*

The NMFS has not designated critical habitat for sei whales.

4.2.5 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.

4.2.5.1 *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 (Perry et al. 1999a; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and

Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring 1993) where adult males join them to breed.

4.2.5.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

Sperm whales were seen in every season except winter (December through February) during systematic surveys off Washington and Oregon from 1989 to 1990 (Green et al. 1992a). More recently, sperm whales were detected acoustically year-round at offshore sites monitored from 2004 to 2008 off the Washington coast, with a peak occurrence from April to August, and at an inshore recording station over the continental shelf they were detected from April to November (Oleson et al. 2009). Acoustic detections of sperm whale were also reported at the inshore monitoring site every month from June through January in 2009; there was an absence of detections between February and May 2009 (Oleson and Hildebrand 2012).

Two noteworthy sperm whale stranding events occurred in this region of the Action Area. During November 1970, there was an incident that was well-publicized by the media of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. A mass stranding of 47 sperm whales occurred in Oregon during June 1979 (Norman et al. 2004a; Rice 1986).

4.2.5.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Given their documented preference for deep offshore waters, sperm whales are unlikely to occur within the Inland Waters region of the Action Area.

4.2.5.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Given their documented preference for deep offshore waters, sperm whales are unlikely to occur within the SEAFAC region of the Action Area since it is characterized by coastal waters removed from the continental shelf break.

4.2.5.2 *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 IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). 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. 2004). 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 2003a). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003a; 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 40^{o} 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). They are seen in every season except winter (December through February) in Washington and Oregon (Green et al. 1992b).

Sperm whales are seasonal migrants to waters off the coast of Washington and Oregon where their densities are highest during spring and summer; they do not appear to occur in these waters during the winter. Sperm whales also tend to occur in the deeper water at the western edge of the Action Area. In surveys of waters off Oregon and Washington conducted by Green et al. (1992b), no sperm whales were encountered in waters less than 200 meters deep, 12 percent of the sperm whales were encountered in waters 200 to 2,000 meters deep (the continental slope), and the remaining 88 percent of the sperm whales were encountered in waters greater than 2,000 meters deep. In surveys conducted by Forney and her co-workers (Forney 2007), sperm whales were reported from the Olympic Coast Slope transects (west of the Olympic Coast National Marine Sanctuary), but not from surveys conducted over the National Marine Sanctuary or the area immediately west of Cape Flattery.

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 2010b; 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).

4.2.5.3 ***Abundance Estimate***

It is estimated that there are between 200,000 and 1,500,000 sperm whales worldwide (NMFS 2009c). Currently there is no reliable abundance estimate for the Alaska/North Pacific stock of sperm whales (Allen and Angliss 2013). The number of sperm whales 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 current best available estimate of abundance for the California, Oregon, and Washington stock is

971 (CV = 0.31) (Carretta et al. 2014). Based on ship surveys conducted in the summer and fall from 1991 to 2008, it is estimated that 329 sperm whales (CV = 0.45) occur in waters off Washington and Oregon (Barlow 2010a). The Barlow (2010b) sperm whale density estimate for waters off Washington and Oregon (1.0 animals per 386 square miles [mi.$_2$] [1,000 square kilometers (Acevedo-Gutierrez et al.)]) is similar to the worldwide global average for this species (1.4 animals per 386 mi.$_2$ [1,000 km$_2$]; (Whitehead 2003b).

4.2.5.4 ***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).

4.2.5.5 ***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. 1997).

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 1998; 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.

4.2.5.6 ***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. 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).

There are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii, and western North Pacific (Whitehead 2002a). Minimum estimates in the eastern North Pacific are 1,719 individuals and 5,531 in the Hawaiian Islands (Carretta et al. 2007). The tropical Pacific is home to approximately 26,053 sperm whales and the western North Pacific has approximately 29,674 (Whitehead 2002a). There was a dramatic decline in the number of females around the Galapagos Islands during 1985 to 1999 versus 1978 to 1992 levels, likely due to migration to nearshore waters of South and Central America (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 (Reeves and Whitehead 1997). 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.

4.2.5.7 ***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 ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003a). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

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).

4.2.5.8 *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 have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997; 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.

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. 2004).

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. (2005b) 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).

4.2.5.9 *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 1997b). 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 1997b). 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 1975b). 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. 1999c).

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 1997a). 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 1997a). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean and those in the Pacific (Weilgart and Whitehead 1997a). 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 1975a). 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. 2011b).

4.2.5.10 *Critical Habitat*

NMFS has not designated critical habitat for sperm whales.

4.2.6 Southern Resident Killer Whale

Killer whales (*Orcinus orca*) are the most widely distributed cetacean (e.g., whales, dolphins, and porpoises) species in the world and likely represent the most widely distributed mammal species in the world. Killer whales have a distinctive color pattern, with black dorsal and white ventral portions. They also have a conspicuous white patch above and behind the eye and a highly variable gray or white saddle behind the dorsal fin.

The species shows considerable size "dimorphism". Adult males develop larger pectoral flippers, dorsal fins, tail flukes, and girths than females. Male adult killer whales can reach up to 32 feet (9.8 m) in length and can weigh nearly 22,000 pounds (10,000 kg); females can reach 28 feet (8.5 m) in length and can weigh up to 16,500 pounds (7,500 kg).

Most information on killer whale life history and biology is from long-term studies of several populations in the eastern North Pacific. Sexual maturity of female killer whales is achieved when the whales reach lengths of approximately 15 to 18 feet (4.6 to 5.4 m), depending on geographic region. The gestation period for killer whales varies from 15 to 18 months, and birth may take place in any month. Calves are nursed for at least 1 year, and may be weaned between 1 and 2 years of age. The birth rate for killer whales is not well understood, but, in some populations, is estimated as every 5 years for an average period of 25 years.

Life expectancy for wild female killer whales is approximately 50 years, with maximum longevity estimated at 80 to 90 years. Male killer whales typically live for about 30 years, with maximum longevity estimated at 50 to 60 years.

4.2.6.1 ***Distribution***

Three kinds of killer whales occur along the Pacific Coast of the United States: Eastern North Pacific (ENP) southern resident killer whales, ENP offshore killer whales, and ENP transient killer whales. Of these only the southern resident killer whales are listed as endangered or threatened under the ESA. Southern resident killer whales primarily occur in the inland waters of Washington State and southern Vancouver Island, although individuals from this population have been observed off the Queen Charlotte Islands (north of their traditional range) and off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (NMFS 2005b).

Southern Resident killer whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods regularly occur in the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca (Felleman et al. 1991; Heimlich-Boran 1988; Olson 1998; Osborne 1999). The K and L pods typically arrive in May or June and remain in this core area until October or November, although both pods make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000). The J pod will occur intermittently in the Georgia Basin and Puget Sound during late fall, winter and early spring. During the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Felleman et al. 1991; Ford et al. 2000; Heimlich-Boran 1988; Olson 1998).

The local movement of southern resident killer whales usually follows the distribution of salmon, which are their preferred prey (Heimlich-Boran 1988; Heimlich-Boran 1986; Nichol and Shackleton 1996). Areas that are major corridors for migrating salmon, and therefore, for southern resident killer whales, include Haro Strait and Boundary Passage, the southern tip of Vancouver Island, Swanson Channel off North Pender Island, and the mouth of the Fraser River delta, which is visited by all three pods in September and October (Felleman et al. 1991; Ford et al. 2000).

4.2.6.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

Three stocks of killer whale may occur and the U.S. Pacific coast: the West Coast Transient stock, the Offshore stock, and the Southern Resident stock (Carretta et al. 2012), although Northern Resident killer whales may be found infrequently in waters off Washington (Allen and Angliss 2012). Killer whales tend to show up along the Oregon coast during late April and May and may target gray whale females and calves migrating north. Based on food type, these probably are transients. As noted above, when observed offshore it is difficult to determine if a particular whale is a transient, offshore, or a resident ecotype.

Southern Resident killer whales regularly visit coastal sites off Washington State and Vancouver Island (Ford et al. 1994a) and in the winter are known to travel as far south as Monterey off central California (Black 2011).

4.2.6.1.2 Occurrence in the Inland Waters of the NWTT Action Area

The Southern Resident stock is most likely to occur in the Inland Waters region of the Action Area. The Southern Resident stock is a trans-boundary stock including killer whales in inland Washington and southern British Columbia waters.

In spring and summer months, the Southern Resident stock is most frequently seen in the San Juan Islands region with intermittent sightings in Puget Sound (Whale Museum 2012). In the fall and early winter months, the Southern Residents are seen more frequently in Puget Sound, where returning chum and Chinook salmon are concentrated (Osborne et al. 1988b). By winter, they spend progressively less time in the inland marine waters and more time off the coast of Washington, Oregon, and California (Black 2011).

While both Southern Resident killer whales and transient killer whales are frequently sighted in the main basin of Puget Sound, their presence near Navy installations varies from not present at all to infrequent sightings, depending on the season (Orca Network 2012; Whale Museum 2012).

Southern Resident killer whales have not been reported in Hood Canal or Dabob Bay since 1995 (NMFS 2008e). Southern resident killer whales (J pod) were historically documented in Hood Canal by sound recordings in 1958 (Ford 1991), a photograph from 1973, and also anecdotal accounts of historical use, but these latter sightings may be transient whales (NMFS 2008e). Near NAVBASE Kitsap Bremerton and Keyport, the Southern Resident killer whale is rare, with the last confirmed sighting in Dyes Inlet in 1997. There was a more recent confirmed Southern Resident occurrence along the Washington State Ferries route between Bremerton and Seattle in December 2007, but the exact location of the sighting is not known (Orca Network 2012). Both Southern resident killer whales and transients have been observed in Saratoga Passage and Possession Sound near NASWI and NAVSTA Everett respectively. Transients and Southern Resident killer whales have also been observed in southern Puget Sound in the Carr Inlet area.

4.2.6.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Individuals belonging to the Alaska Resident stock are the killer whales most likely to occur in the SEAFAC region of the Action Area, and are more likely from spring through fall (Dahlheim et al. 2009). Southern Resident killer whales (L pod, 30 individuals) were photographically identified in Chatham Strait, Southeast Alaska (northwest of Behm Canal), in June 2007. Southern Residents were previously thought to only range as far north as the Queen Charlotte Islands, B. C.; however, this sighting extends their known range about 200 mi. to the north (Barre 2012). However, Alaska Resident stock of killer whales are more likely to be encountered in and around the Behm Canal than Southern Resident killer whales. We would expect encounters with Southern Resident killer whales near SEAFAC to be very rare and therefore discountable.

4.2.6.2 ***Population Structure***

Southern resident killer whales are the only marine mammal that begin and end their lives almost entirely within the Action Area. Southern resident killer whales consist of three pods, or stable familial groups: the J pod, K pod, and L pod. The J pod is seen most frequently along the western shore of San Juan Island and is the only pod observed regularly in Puget Sound throughout winter (Heimlich-Boran 1988; Osborne 1999). The K pod is most frequently observed during May and June when they occur along the western shore of San Juan Island while searching for salmon. The L pod is the largest of the three pods (Ford et al. 1994b) and frequently breaks off into separate subgroups. During the months of July, August, and September, all three pods of southern resident killer whales remain in the inland waterways of Puget Sound, Strait of Juan de Fuca, and southern Georgia Strait. Since the late 1970s, K and L pods typically arrived in this area in May or June and remained there until October or November and appeared to have left these waters by December (Osborne 1999). Since the late 1990s, however, all three pods have tended to remain in this area through December and K and L pods have remained in inland waters until January or February for several years (NMFS 2008d). While they tend to spend most of their time in inland waters, both of these pods would periodically frequent the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000).

Less is known about the distribution and movements of southern resident killer whales from late fall, through winter, and into early spring. Over this time interval, the J pod has been observed periodically in the Georgia Basin and Puget Sound, but its movement at other times is uncertain (Osborne 1999); although this pod was sighted once off Cape Flattery, Washington, in March 2004 (NMFS 2008d). The K and L pods have been sighted as they passed through the Strait of Juan de Fuca in late fall, which led Krahn et al. (Krahn et al. 2002) to conclude that these pods might travel to the outer coasts of Vancouver Island and Washington, although they may continue to other areas from there. Based on sighting information and stranding data collected from 1975 through 2007, southern resident killer whales travel to Vancouver Island and the Queen Charlotte Islands, coastal Washington, coastal Oregon, and California (NMFS 2008d).

4.2.6.3 ***Abundance Estimate***

Photo-identification of individual whales through the years has resulted in a substantial understanding of this stock's structure, behaviors, and movements in inland waters. The current abundance estimate for this stock is 85 whales (Carretta et al. 2013a).

In the forty-year history of ORCA SURVEY, a long-term photo-identification study of this whale population, the greatest number of calves born in a year was 9 in 1977, and there were none born that survived in 2013 or 2014. However, since December 2014, five calves have been observed which could represent a turnaround in what has been a negative population trend in recent years.

4.2.6.4 ***Natural Threats***

Southern resident killer whales, like many wild animal populations, experience highest mortality in the first year age class (Krahn et al. 2002; Olesiuk et al. 1990), although the reasons for these mortalities are still uncertain. The causes could include poor mothering, infectious or non-infectious diseases, and infanticide (Gaydos et al. 2004).

Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. They did, however, identify pathogens in sympatric odontocete species that could threaten the long-term viability of the small southern resident population.

4.2.6.5 ***Anthropogenic Threats***

Several human activities appeared to contribute to the decline of southern resident killer whales. Southern resident killer whales were once shot deliberately in Washington and British Columbia (Baird 2001; Olesiuk et al. 1990). However, between 1967 and 1973, 43 to 47 killer whales were removed from the population for displays in oceanaria; because of those removals, the southern resident killer whale population declined by about 30 percent. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

Over the same time interval, southern resident killer whales have been exposed to changes in the distribution and abundance of their prey base (primarily Pacific salmon) which has reduced their potential forage base, potential competition with salmon fisheries, which reduces their realized forage base, disturbance from vessels, and persistent toxic chemicals in their environment.

Salmon, which are the primary prey species for southern resident killer whales, have declined because of land alteration throughout the Pacific Northwest associated with agriculture, timber harvest practices, the construction of dams, and urbanization, fishery harvest practices, and hatchery operations. Many of the salmon populations that were once abundant historically have declined to the point where they have been listed as endangered or threatened with extinction. Since the late 1800s, salmon populations throughout the Columbia River basin have declined (Krahn et al. 2002). Two recent studies have examined the relationships between salmon abundance and population dynamics of resident killer whales and support the belief that Chinook and chum salmon are most important to the Southern Residents. Both studies, however, are limited by incomplete data on salmon occurrence and year-round range use by the whales (NMFS 2008d).

Since the 1970s commercial shipping, whale watching, ferry operations, and recreational boat traffic have increased in Puget Sound and the coastal islands of southern British Columbia. This traffic exposes southern resident killer whales to several threats that have consequences for the species' likelihood of avoiding extinction and recovering if it manages to avoid extinction. First, these vessels increase the risks of southern resident killer whales being struck, injured, or killed by ships. In 2005, a southern resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. However, in 2006, an adult male southern resident killer whale, L98, was killed in a collision with a tug boat; given the gender imbalances in the southern resident killer whale population, we assume that the death of this adult male would have reduced the demographic health of this population.

Second, the number and proximity of vessels, particularly whale-watch vessels in the areas occupied by southern resident killer whales, represents a source of chronic disturbance for this population. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Cotton 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Evans et al. 1992; 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; Corkeron 1995; Erbe 2002b; Felix 2001; Magalhaes et al. 2002; 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. The whales' responses changed with these different variables and, in some circumstances, the whales did not respond to the vessels. In other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

In addition to the disturbance associated with the presence of vessel, the vessel traffic affects the acoustic ecology of southern resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of southern resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15 percent during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2009a) reported that southern resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales need to increase the duration of their vocalization to avoid masking by the boat noise.

Exposure to contaminants may also harm southern resident killer whales. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants has been documented in southern resident killer whales (Krahn et al. 2007; Ross et al. 2000). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007). Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

4.2.6.6 ***Status and Trends***

Southern resident killer whales were listed as endangered under the ESA in 2005 (70 FR 69903). In the mid- to late-1800s, southern resident killer whales were estimated to have numbered around 200 individuals. By the mid-1960s, they had declined to about 100 individuals. As discussed in the preceding section, between 1967 and 1973, 43 to 47 killer whales were removed from the population to provide animals for displays in oceanaria and the population declined by about 30 percent as a result of those removals. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

At population sizes between 75 and 90 individuals, we would expect southern resident killer whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox 2007) —including stochastic sex determination (Lande et al. 2003)—and the effects of these phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson et al. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008). For example, the small number of adult male southern resident killer whales might represent a stable condition for this species or it might reflect the effects of stochastic sex determination. Regardless, a high mortality rates among adult males in a population with a smaller percentage of males would increase the imbalance of male-to-female gender ratios in this population and increase the importance of the few adult males that remain.

At these population sizes, populations experience higher extinction probabilities because stochastic sexual determination leaves them with harmful imbalances between the number of male or female animals in the population (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson et al. 2006). In general, an individual's contribution to the growth (or decline) of the population it represents depends, in part, on the number of individuals in the population: the smaller the population, the more the performance of a single individual is likely to affect the population's growth or decline (Coulson et al. 2006). Given the small size of the southern resident killer whale population, the performance (= "fitness," measured as the longevity of individuals and their reproductive success over their lifespan) of individual whales would be expected to have appreciable consequences for the growth or decline of the southern resident killer whale population.

These phenomena would increase the extinction probability of southern resident killer whales and amplify the potential consequences of human-related activities on this species. Based on their population size and population ecology (that is, slow-growing mammals that give birth to single calves with several years between births), we assume that southern resident killer whales would have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities that result in the death or injury of individual whales (for example, ship strikes or entanglement) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) as well as endogenous threats resulting from the small size of their population. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer southern resident killer whales remain in these circumstances, the greater their extinction probability becomes.

4.2.6.7 ***Diving and Social Behavior***

Killer whales are highly social animals that occur primarily in groups or pods of up to 40-50 animals (Baird 2000; Dahlheim and Heyning 1999). Larger aggregations of up to several hundred individuals occasionally form, but are usually considered temporary groupings of smaller social units that probably congregate near seasonal concentrations of prey, for social interaction, or breeding (Baird 2000; Dahlheim and Heyning 1999; Ford et al. 2000). The basic social units are matrilines, which usually consist of an adult female, her sons and daughters, the offspring of her daughters, and might extend to include 3 to five generations of killer whales (Baird 2000; Ford 2002; Ford et al. 2000). The members of matrilines maintain such strong social connections that individuals rarely separate from these groups for more than a few hours. Groups of related matrilines are known as pods—for example, L Pod of southern resident killer whales consists of 12 matrilines—which are less cohesive than matrilines (matrilines within a pod might travel separately for weeks or months). Clans are the next level of social structure in resident killer whales and consist of pods with similar vocal dialects and common, but older, maternal heritage.

In terms of gender and age composition, southern and northern resident killer whales social groups consisted of 19 percent adult males, 31 percent adult females, and 50 percent immature whales of either sex in 1987 (Olesiuk et al. 1990). This composition is comparable with the composition of southern Alaska resident killer whales and killer whale populations in the Southern Ocean (Matkin et al. 2003; Miyazaki 1989).

4.2.6.8 *Vocalizations and Hearing*

Killer whales produce a wide variety of clicks, whistles, and pulsed calls (Ford 1989; Schevill and Watkins 1966; Thomsen et al. 2001). Their clicks are relatively broadband, short (0.1 to 25 milliseconds), and range in frequency from 8 to 80 kHz with an average center frequency of 50 kHz and an average bandwidth of 40 kHz (Au et al. 2004). Killer whales apparently use these signals to sense objects in their environment, such as prey; whales foraging on salmon produce these signals at peak-to-peak source levels ranging from 195 to 225 dB re 1 µPa at 1 m (Au et al. 2004).

Killer whale whistles are tonal signals that have longer duration (0.06 to 18 seconds) and frequencies ranging from 0.5 to 10.2 kHz (Thomsen et al. 2001). Killer whales are reported to whistle most often while they have been engaged in social interactions rather than during foraging and traveling (Thomsen et al. 2002). Northern resident killer whales whistles have source levels ranging from 133 to 147 dB re 1 µPa at 1 m (Miller 2006).

Killer whale pulsed calls are the most commonly observed type of signal associated with killer whales (Ford 1989). With both northern and southern resident killer whales, these signals are relatively long (600 to 2,000 ms) and range in frequency between 1 and 10 kHz; but may contain harmonics up to 30 kHz (Ford 1989). The variable calls of killer whales have source levels ranging from 133 to 165 dB while stereotyped calls have source levels ranging from 135 to 168 dB re 1 µPa at 1 m (Miller 2006). Killer whales use these calls when foraging and traveling (Ford 1989).

4.2.7 Guadalupe Fur Seal

Guadalupe fur seals are medium sized, sexually dimorphic otariids that are generally asocial with their conspecifics and other species (Belcher and T.E. Lee 2002; Reeves et al. 2002). Except for adult males, members of this species resemble California sea lions and northern fur seals. Distinguishing characteristics of the Guadalupe fur seal include the digits on their hind flippers (all of similar length), large, long foreflippers, unique vocalizations, and a characteristic behavior of floating vertically with their heads down in the water and their hind flippers exposed for cooling (Reeves et al. 2002).

Guadalupe fur seals' historic range included the Gulf of Farallones, California to the Revillagigedo Islands, Mexico (Belcher and T.E. Lee 2002; Rick et al. 2009b). Currently, they breed mainly on Guadalupe Island, Mexico, 155 miles off of the Pacific Coast of Baja California.

A smaller breeding colony, discovered in 1997, appears to have been established at Isla Benito del Este, Baja California, Mexico (Belcher and T.E. Lee 2002).

There are reports of individuals being sighted in the California Channel Islands, Farallone Islands, Monterey Bay, and other areas of coastal California and Mexico (Belcher and T.E. Lee 2002; Carretta et al. 2002; Reeves et al. 2002). A single female gave birth to a pup on the Channel Islands in 1997.

The Guadalupe fur seal population is slowly recovering from the brink of extinction. The current population abundance is approximately 10,000 animals. Of all the fur seal species, this one is the least studied due to their limited geographic locations. The Guadalupe fur seal population does appear to be increasing annually.

Before intensive hunting decreased their numbers, Guadalupe fur seals ranged from Monterey Bay, California, to the Revillagigedo Islands, Mexico (Aurioles-Gamboa and Camacho-Rios 2007), but have occasionally been identified from strandings (Northwest Region Stranding Database; (Wilkinson 2013)) or in archaeological contexts as far north as northern California, Oregon, and Washington (Etnier 2002; Rick et al. 2009a). Between 1989 and 2011, a total of 118 dead stranded animals were found along the Washington and Oregon coastline (Northwest Region Stranding Database; (Wilkinson 2013)). Between June 20 and November 1, 2007, 19 Guadalupe fur seals stranded on the Washington and Oregon outer coasts, prompting NOAA to declare an Unusual Mortality Event on October 19, 2007 (Lambourn et al. 2012). The Unusual Mortality Event was officially closed on December 11, 2009. In 2012, approximately 58 Guadalupe fur seals stranded on the outer coasts of Washington and Oregon (Lambourn 2013 pers. comm.). This is three times the number of strandings that prompted the Unusual Mortality Event in 2007. Of all the strandings reported off Washington and Oregon (1989 to 2012), most occurred from mid-May through August with occasional reports between October and December ((Lambourn et al. 2012); Northwest Region Stranding Database; (Wilkinson 2013)).Sightings of live animals off Washington and Oregon are more limited, although there is photo documentation of apparently healthy Guadalupe fur seals in offshore waters of Washington and British Columbia in recent years during summer and early autumn (Lambourn et al. 2012). Given the increased number of strandings in the Pacific Northwest, coupled with their increasing population, it is possible that Guadalupe fur seals are returning to their historic pelagic migration range suggested by the archaeological findings (Etnier 2002; Lambourn et al. 2012; Rick et al. 2009a).

4.2.7.1 *Occurrence in the Offshore Portion of the NWTT Action Area*

Based on their rookeries occurring in Baja California, Mexico, the species is predominantly distributed off Mexico, but with annual strandings in Oregon and Washington, Guadalupe fur seals are considered "seasonal" migrants within the offshore portion of the NWTT Action Area.

Given the lack of at-sea sightings by NMFS and the documented coastal strandings (Lambourn et al. 2012), Guadalupe fur seals are more likely to be coastally distributed given their extralimital at-sea occurrence and associated stranding records (Lambourn et al. 2012).

Most Guadalupe fur seal strandings in the Pacific Northwest likely represent young individuals at the extreme limits of their preferred geographic foraging range as indicated by the poor health of examined carcasses to date. There is no current evidence to support normal population expansion into the Pacific Northwest (e.g., lack of significant sightings of healthy individuals at-sea, lack of sightings of healthy individuals hauled-out on shore, lack of healthy individuals in the stranding record, etc.). There were only three at-sea sightings of Guadalupe fur seals made less than 30 miles from shore (Lambourn et al. 2012).

4.2.7.2 ***Occurrence in the Inland Waters of the NWTT Action Area***

Guadalupe fur seals are not expected to occur within the Inland Waters region of the Action Area. Strandings from the offshore portion of the Action Area have been documented as noted above, but they are considered extralimital in the inland waters.

4.2.7.3 ***Occurrence in the Western Behm Canal, Alaska***

Guadalupe fur seals are not expected to occur in the SEAFAC portion of the Action Area.

4.2.7.4 ***Critical Habitat***

NMFS has not designated critical habitat for Guadalupe fur seals.

4.2.8 Leatherback Sea Turtle

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.

Female leatherback sea turtles lay clutches of approximately 100 eggs on sandy, tropical beaches. Females nest several times during a nesting season, typically at 8 to 12 day intervals. After 60 to 65 days, leatherback hatchlings with white striping along the ridges of their backs and on the margins of the flippers emerge from the nest. Leatherback hatchlings are approximately 50 to 77 cm (2 to 3 inches) in length, with fore flippers as long as their bodies, and weigh approximately 40 to 50 grams (1.4 to 1.8 ounces).

Leatherback sea turtles lack the crushing chewing plates characteristic of sea turtles that feed on hard-bodied prey (Pritchard 1971). Instead, they have pointed tooth-like cusps and sharp edged jaws that are perfectly adapted for a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. A leatherback's mouth and throat also have backward-pointing spines that help retain such gelatinous prey.

4.2.8.1 ***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 1993a). 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 1993a; 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 1998a). 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 2007a). 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.

4.2.8.1.1 Occurrence in the Offshore Portion of the NWTT Action Area

Few quantitative data are available concerning the seasonality, abundance, or distribution of leatherbacks in the central North Pacific Ocean. 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, where longline fisheries operate (Dutton et al. 1998). Leatherbacks from nesting beaches in the Indo-Pacific region have been tracked migrating thousands of kilometers from nesting areas to summer foraging grounds off the coast of northern California (Benson et al. 2007), including a 6,385 mi. (10,276 km) migration from a nesting beach in Papua New Guinea to foraging grounds off the coast of Oregon (Benson et al. 2007). The waters off the Oregon and California coasts have been repeatedly recognized by scientists and agencies as comprising one of the most important leatherback foraging areas in the Pacific (NMFS and USFWS 1998b).

Leatherback turtles are regularly seen off the western coast of the United States. Off the California coast, the highest densities of leatherback sea turtles were found off central California (Benson et al. 2007). Telemetry studies have shown areas of concentration along the central California coast and in the waters of Oregon and Washington (Benson et al. 2011). Stinson (1984) concluded that the leatherback was the most common sea turtle in U.S. waters north of Mexico. Aerial surveys off Washington, Oregon, and California indicate that most leatherbacks occur in waters over the continental slope, with a few over the continental shelf (Eckert 1993b). Green et al. (1992a) conducted a study between 1989 and 1990 to assess the presence and abundance of federally listed species along the coasts of Washington and Oregon. During the study, 16 sea turtles were observed; all sightings were of leatherback sea turtles and all occurred between June and September, with most sightings (10) occurring in July (Green et al. 1992). The data suggest that leatherback sea turtles are most likely to occur in the NWTT Action Area in summer and early fall when water temperatures are warmer. Roe et al. (2014) produced some of the first Pacific wide leatherback probabilities maps based on use distribution from 135 satellite-tracked adult turtles.

4.2.8.1.2 Occurrence in the Inland Waters of the NWTT Action Area

Leatherback sea turtles are occasionally sighted within the Strait of Juan de Fuca, but are rare in Puget Sound (NMFS and USFWS 1998b; NMFS and USFWS 2007a). While leatherback sea turtles are capable of foraging in inland waters, their preferred foraging habitat is offshore. Therefore, the leatherback sea turtle is not analyzed further for military activities occurring in the inland waters portion of the NWTT Action Area.

4.2.8.1.3 Occurrence in the Western Behm Canal, Alaska - NWTT Action Area

Stinson's (1984) archival search of 363 sea turtles sighted along the Pacific coast from Baja California, Mexico to the Gulf of Alaska from 1917 to 1982 indicated that only 6 of the recorded leatherback sightings occurred in Alaska waters. From 1960 to 1968, 19 sightings of leatherback turtles in Alaska waters were recorded (11 were sightings, 3 were netted and released, 3 were netted and killed, and 2 were carcasses) (Hodge and Wing 2000). While leatherback sea turtles are physically capable of foraging in Alaska waters, they are rare in the area and prefer offshore waters to the south. Therefore, the leatherback sea turtle is not analyzed further for activities occurring in the Western Behm Canal portion of the Action Area.

4.2.8.2 *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.

There are no known nesting habitats for the leatherback sea turtle in the Action Area. The major nesting populations of the eastern Pacific Ocean stock occur in Mexico, Costa Rica, Panama, Colombia, Ecuador, and Nicaragua (Chaloupka et al. 2004; Dutton et al. 1999; Eckert and Sarti 1997; Márquez 1990; Sarti M. 1996; Spotila et al. 1996), with the largest ones in Mexico and Costa Rica. There are 28 known nesting sites for the western Pacific Ocean stock, with an estimated 5,000 to 9,100 leatherback nests annually across the western tropical Pacific Ocean, from Australia and Melanesia (Papua New Guinea, Solomon Islands, Fiji, and Vanuatu) to Indonesia, Thailand, and China (Chaloupka et al. 2004; Chua 1988; Dutton 2006; Hirth et al. 1993; Suarez et al. 2000).

Leatherbacks have been in decline in all major Pacific basin rookeries (nesting areas/groups) (NMFS and USFWS 2007a; TEWG 2007) for at least the last two decades (Gilman 2008; Sarti M. 1996; Spotila et al. 1996; Spotila et al. 2000). Causes for this decline include the nearly complete harvest of eggs and high levels of mortality during the 1980s, primarily in the high seas driftnet fishery, which is now banned (Chaloupka et al. 2004; Eckert and Sarti 1997; Sarti M. 1996). With only four major rookeries remaining in the western Pacific Ocean and two in the eastern Pacific Ocean, the Pacific leatherback is at an extremely high risk of extinction.

4.2.8.3 ***Natural Threats***

The various habitat types leatherback sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which leatherback sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Larger leatherback sea turtles, including adults, are also killed by sharks and other large, marine predators.

4.2.8.4 ***Anthropomorphic Threats***

Leatherback sea turtles are endangered by several human activities, including fisheries interactions, entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), direct harvest, egg collection, the destruction and degradation of nesting and coastal habitat, boat collisions, and ingestion of marine debris.

The foremost threat is the number of leatherback turtles killed or injured in fisheries. Spotila (2004) concluded that a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s is 1,500 animals. He estimates that this represented about a 23 percent mortality rate (or 33 percent if most mortality was focused on the East Pacific population). Spotila (2000) asserts that most of the mortality associated with the Playa Grande nesting site was fishery related.

Leatherback sea turtles are exposed to commercial fisheries in many areas of the Atlantic Ocean. For example, leatherback entanglements in fishing gear are common in Canadian waters where Goff and Lien (Goff and Lien 1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland and Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported taken by the many other nations that participate in Atlantic pelagic longline fisheries (see NMFS 2001, for a complete description of take records), including Taiwan, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland.

In the Pacific Ocean, between 1,000 and 1,300 leatherback sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004). Shallow-set longline fisheries based out of Hawai'i are estimated to have captured and killed several hundred leatherback sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about 1 or 2 leatherback sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawai'i are estimated to have captured about 19 leatherback sea turtles, killing about 5 of these sea turtles. A recent biological opinion on these fisheries expected this rate of interaction and deaths to continue into the foreseeable future. Leatherback sea turtles have also been and are expected to continue to be captured and killed in the deep-set based longline fisheries based out of Hawaii and American Samoa.

Shrimp trawls in the Gulf of Mexico capture the largest number of leatherback sea turtles: each year, they have been estimated to capture about 3,000 leatherback sea turtles with 80 of those sea turtles dying as a result. Along the Atlantic coast of the U.S., NMFS estimated that about 800 leatherback sea turtles are captured in pelagic longline fisheries, bottom longline and drift gillnet fisheries for sharks as well as lobster, deep-sea red crab, Jonah crab, dolphin fish and wahoo, and Pamlico Sound gillnet fisheries. Although most of these turtles are released alive, these fisheries combine to kill about 300 leatherback sea turtles each year; the health effects of being captured on the sea turtles that survive remain unknown.

Leatherback sea turtles are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Tomás et al. 2000). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alió-M 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50 to 95 percent (Eckert et al. 2007). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets. There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (Fretey 2001). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

On some beaches, nearly 100 percent of the eggs laid have been harvested. Spotila et al. (1996) and Eckert et al. (2007) note that adult mortality has also increased significantly, particularly as a result of driftnet and longline fisheries. Like green and hawksbill sea turtles, leatherback sea turtles are threatened by domestic or domesticated animals that prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

Oil spills are a risk for all sea turtles. Several aspects of sea turtles life histories put them at risk, including the lack of avoidance behavior of oiled waters and indiscriminate feeding in convergence zones. Sea turtles are air breathers and all must come to the surface frequently to take a breath of air. In a large oil spill, these animals may be exposed to volatile chemicals during inhalation (NMFS 2010d).

Additionally, sea turtles may experience oiling impacts on nesting beaches when they come ashore to lay their eggs, and their eggs may be exposed during incubation potentially resulting in increased egg mortality and/or possibly developmental defects in hatchlings. Hatchlings emerging from their nests may encounter oil on the beach and in the water as they begin their lives at sea (NMFS 2010d).

External Effects: Oil and other chemicals on skin and body may result in skin and eye irritation, burns to mucous membranes of eyes and mouth, and increased susceptibility to infection (NMFS 2010d).

Internal Effects: Inhalation of volatile organics from oil or dispersants may result in respiratory irritation, tissue injury, and pneumonia. Ingestion of oil or dispersants may result in gastrointestinal inflammation, ulcers, bleeding, diarrhea, and maldigestion. Absorption of inhaled and ingested chemicals may damage organs such as the liver or kidney, result in anemia and immune suppression, or lead to reproductive failure or death (NMFS 2010d).

4.2.8.5 ***Status and Trends***

The leatherback turtle was listed under the Endangered Species Act as endangered throughout its range in 1970. There is a recovery plan for this species (NMFS and USFWS 1998a).

Leatherback 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.

4.2.8.6 ***Diving***

The leatherback sea turtle is one of the deepest divers in the ocean, with dives as deep as 3,937 ft (1,200 m), although it spends most of its time feeding at a depth of less than 328 ft (100 m). Leatherback turtles primarily feed on gelatinous zooplankton such as cnidarians (jellyfish and siphonophores) and tunicates (salps and pyrosomas) (Bjorndal 1997; NMFS and USFWS 1998a). The leatherback dives continually and spends short periods of time on the surface between dives (Eckert et al. 1989b; Southwood et al. 1999). Typical dive durations averaged 6.9 to 14.5 minutes (min) per dive, with a maximum of 42 min (Eckert et al. 1996). Sea turtles typically remain submerged for several minutes to several hours depending upon their activity state (Standora et al. 1984). Long periods of submergence hamper detection and confound census efforts. During migrations or long distance movements, leatherbacks maximize swimming efficiency by traveling within 15 ft (5 m) of the surface (Eckert 2002).

4.2.8.7 ***Social Behavior***

Male leatherbacks do not return to land after they hatch from their nests whereas mature females return to land only to lay eggs (Spotila 2004). Aside from this brief terrestrial period, which lasts approximately three months during egg incubation and hatching, leatherback turtles are rarely encountered out of the water. Hatchling leatherbacks are pelagic, but nothing is known about their distribution during the first 4 years of life (Musick and Limpus 1997).

The Pacific coast of Mexico is generally regarded as the most important leatherback breeding ground in the world, although nesting on Pacific beaches under U.S. jurisdiction has always been rare (NMFS and USFWS 1998a). Based on a single aerial survey in 1980 of Michoacán, Guerrero, and Oaxaca, and on published and anecdotal data, Pritchard (Pritchard 1982a) estimated that 30,000 females nested annually in these three Mexican states. Lower-density nesting was (and still is) reported farther north in Jalisco (NMFS and USFWS 1998a) and in Baja California, where the northernmost eastern Pacific nesting sites are found (Fritts et al. 1982). Leatherbacks nest along the western coast of Mexico from November to February, although some females arrive as early as August (NMFS and USFWS 1998a), and in Central America from October to February (Lux et al. 2003). This species nests primarily on beaches with little reef or rock offshore. On these types of beaches erosion reduces the probability of nest survival. To compensate, leatherbacks scatter their nests over large geographic areas and lay on average two times as many clutches as other species (Eckert 1987). Females may lay up to nine clutches in a season (although six is more common), and the incubation period is 58–65 days. At Playa Grande, Costa Rica, and in French Guiana, the mean inter-nesting period was 9 days (Lux et al. 2003). Post-nesting adults appear to migrate along bathymetric contours from 656 to 11,483 ft (200 to 3,500 m) (Morreale et al. 1994), and most of the eastern Pacific nesting stocks migrate south (NMFS and USFWS 1998a). Other principal nesting sites in the Pacific Ocean indicate that gene flow between eastern and western Pacific nesting populations is restricted (Dutton et al. 2005; Dutton et al. 2006; Dutton et al. 1999; Dutton et al. 1996; Dutton et al. 2003).

4.2.8.8 ***Vocalization and Hearing***

Sea turtles do not have an external ear pinnae or eardrum. Instead, they have a cutaneous layer and underlying subcutaneous fatty layer that function as a tympanic membrane. The subcutaneous fatty layer receives and transmits sounds to the middle ear and into the cavity of the inner ear (Ridgway et al. 1969). Sound also arrives by bone conduction through the skull. Sound arriving at the inner ear via the columella (homologous to the mammalian stapes or stirrup) is transduced by the bones of the middle ear.

Sea turtle auditory sensitivity is not well studied, though a few preliminary investigations suggest that it is limited to low frequency bandwidths, such as the sounds of waves breaking on a beach. The role of underwater low-frequency hearing in sea turtles is unclear. It has been suggested that sea turtles may use acoustic signals from their environment as guideposts during migration and as a cue to identify their natal beaches (Lenhardt et al. 1983).

Lenhardt et al. (1983) applied audio frequency vibrations at 250 Hz and 500 Hz to the heads of loggerheads and Kemp's ridleys submerged in salt water to observe their behavior, measure the attenuation of the vibrations, and assess any neural-evoked response. These stimuli (250 Hz, 500 Hz) were chosen as representative of the lowest sensitivity area of marine turtle hearing (Wever and Vernon 1956). At the maximum upper limit of the vibratory delivery system, the sea turtles exhibited abrupt movements, slight retraction of the head, and extension of the limbs in the process of swimming. Lenhardt et al. (1983) concluded that bone-conducted hearing appears to be a reception mechanism for at least some of the sea turtle species, with the skull and shell acting as receiving surfaces. Finally, sensitivity even within the optimal hearing range was low as threshold detection levels in water are relatively high at 160 to 200 decibels referenced to one micro Pascal at a distance of one meter (dB re 1 μPa-m), which is the standard reference measure for underwater sound energy in this regard (Lenhardt et al. 1994a).

Ridgway et al. (1969) used aerial and mechanical stimulation to measure the cochlea in three specimens of green turtle, and concluded that they have a useful hearing span of perhaps 60 to 1,000 Hz, but hear best from about 200 Hz up to 700 Hz, with their sensitivity falling off considerably below 200 Hz. The maximum sensitivity for one animal was at 300 Hz, and for another was at 400 Hz. At the 400 Hz frequency, the green turtle's hearing threshold was about 64 dB in air (approximately 126 dB in water). At 70 Hz, it was about 70 dB in air (approximately 132 dB in water). We may be able to extrapolate this data to pertain to all hard-shell sea turtles (i.e., the olive ridley, green, loggerhead, and Kemp's ridley turtles). No audiometric data are available for the leatherback turtle, but based on other sea turtle hearing capabilities, they probably also hear best in the low frequencies.

For exposures to impulsive sound, a recent study on the effects of air guns on sea turtle behavior also suggests that sea turtles are most likely to respond to low-frequency sounds (McCauley et al. 2000). Loggerhead sea turtles will avoid air-gun arrays at 2 km and at 1 km, with received levels of 166 dB re 1 μPa-m and 175 dB re 1 μPa, respectively (McCauley et al. 2000). The sea turtles' response was consistent: above a level of about 166 dB re 1 μPa, the sea turtles noticeably increased their swimming activity. Above 175 dB re 1 μPa, their behavior became more erratic, possibly indicating that they were agitated (McCauley et al. 2000).

Currently it is believed that the range of maximum sensitivity for sea turtles is 200 to 800 Hz, with an upper limit of about 2,000 Hz (Lenhardt 1994; Moein et al. 1994). Green turtles are most sensitive to sounds between 200 and 700 Hz, with peak sensitivity at 300 to 400 Hz (Ridgway et al. 1969). They possess an overall hearing range of approximately 60 to 1,000 Hz (Ridgway et al. 1969). Juvenile loggerhead turtles hear sounds between 250 and 1,000 Hz and, therefore, often avoid low-frequency sounds (Bartol et al. 1999b). Finally, sensitivity even within the optimal hearing range is apparently low—threshold detection levels in water are relatively high at 160 to 200 dB re 1 μPa-m (Lenhardt 1994). Given the lack of audiometric information for leatherback turtles, the potential for TTS among leatherback turtles must be classified as unknown but would likely follow those of other sea turtles. In terms of sound emission, nesting leatherback turtles produce sounds in the 300 to 500 Hz range (Mrosovsky 1972).

4.2.9 Puget Sound/Georgia Basin Canary Rockfish

Canary rockfish are large rockfish that reach up to 2.5 feet (77 cm) in length and 10 pounds (4 kg) in weight. Adults have bright yellow to orange mottling over gray, 3 orange stripes across the head, and orange fins. Animals less than 14 inches long have dark markings on the posterior part of the spiny dorsal fin and gray along the lateral line.

4.2.9.1 ***Distribution***

Canary rockfish are found from the northern Baja peninsula north to the western Gulf of Alaska, and with the greatest abundance along British Columbia to central California (Cailliet et al. 2000; Hart 1973; Love et al. 2002; Miller and Lea 1972).

The Puget Sound/Georgia Basin DPS of canary rockfish occur throughout Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

4.2.9.2 ***Habitat***

Canary rockfish occupy a variety of habitats based upon their life stage. Larvae and younger juveniles tend to occupy shallow waters at the beginning of their lives, but generally remain in the upper 100 m of the water column (Love et al. 2002). Juveniles initially settle into tide pools and rocky reefs (Cailliet et al. 2000; Love et al. 1991; Love et al. 2002; Miller and Geibel 1973). Juveniles have also been observed in diurnal movements, occurring near sand-rock interfaces in groups by day and moving over sandy areas at night (Love et al. 2002). After as much as 3 years, juveniles move into deeper rocky reefs, forming loose schools, rarely on but generally near the bottom (Boehlert 1980; Cailliet et al. 2000; Johnson et al. 2003; Lamb and Edgell 1986; Methot and Stewart 2005; Phillips 1960; Rosenthal et al. 1998; Starr 1998; Tissot et al. 2007). Adults may be found in waters of up to 400 m, but tend to be most common in the 80 to 200 m range, or even shallower (Methot and Stewart 2005; Moser 1996b; Tissot et al. 2007). Mid shelf locations seem to have the highest concentrations of canary rockfish off Washington and Oregon (Weinberg 1994). Adults tend to occur in shallow areas in higher latitudes than their southern counterparts, although adults do appear to move into progressively deeper waters as they age (Methot and Stewart 2005; Vetter and Lynn 1997). It is believed that, within Puget Sound, canary rockfish were most common in the 1960s and 1970s in Tacoma Narrows, Hood Canal, San Juan Islands, Bellingham, and Appletree Cove (Delacy et al. 1972; Miller and Borton 1980). A latitudinal gradient may be present by age class, with older and larger individuals preferably occupying more northerly habitat (Dark et al. 1983).

4.2.9.3 ***Movement***

Individual canary rockfish can range widely (up to 700 km over several years), although patterns of residency have been observed (Casillas et al. 1998; DeMott 1983; Gascon and Miller 1981;

Lea et al. 1999; Love et al. 2002). In addition, seasonal movements have been found, with individuals moving from 160 to 210 m depths in late winter to 100 to 170 m in late summer (COSEWIC in press).

4.2.9.4 ***Reproduction***

Canary rockfish develop their young internally before giving birth to live young as larvae. During each annual spawning event, a female can produce 260,000 to 1.9 million eggs, depending upon her size and age (Guillemot et al. 1985; NMFS 2008c). Unlike some other rockfish, there does not appear to be a latitudinal or geographic gradient associated with number of eggs produced (Gunderson et al. 1980; Love et al. 2002). Birth takes place in Oregon and Washington waters between September through March, with a peak in December and January. The peak in British Columbia waters is slightly later (February) (Barss 1989; Hart 1973; Westrheim and Harling 1975; Wyllie Echeverria 1987).

4.2.9.5 ***Growth and Development***

When born, larvae are 3.6 to 4.0 mm in length and take from 1 to 4 months to develop into juveniles (Krigsman 2000; Love et al. 2002; Moser 1996a; Richardson and Laroche 1979; Stahl-Johnson 1985; Waldron 1968). As with other rockfish, females seem grow more quickly than do males, with females reaching sexual maturity at 7 to 9 years of age (35 to 45 cm in length) versus males at 7 to 12 years (~41 cm in length) off Oregon (Boehlert and Kappenman 1980; Lenarz and Echeverria 1991; STAT 1999; Westrheim and Harling 1975). Mean length at sexual maturity off Vancouver Island is 41 cm for females and 48 cm for males (Westrheim and Harling 1975). Canary rockfish are known to frequently reach 60 to 75 years of age and have been found to be as old as 84 years (Andrews et al. 2007; Cailliet et al. 2001; Cailliet et al. 2000). Maximum reported sizes are 76 cm and 4.5 kg (Boehlert 1980; IGFA 1991; Love et al. 2002; Methot and Stewart 2005; Williams et al. 1999).

4.2.9.6 ***Foraging***

Canary rockfish prey upon different species as they age. Larvae are planktivores, consuming invertebrate eggs, copepods, and nauplii (Love et al. 2002; Moser and Boehlert 1991). Juveniles feed upon zooplankton, including crustaceans, juvenile polychaetes barnacle cyprids, and euphasiid eggs and larvae (Gaines and Roughgarden 1987; Love et al. 1991). However, adults move into a carnivorous lifestyle as well as eating euphasiids and other crustaceans. Adults consume other fishes such as shortbelly rockfish, mytophids and stomiatiods (Cailliet et al. 2000; Love et al. 2002). However, oceanographic and climactic shifts can alter foraging such that canary rockfish feed on other available species (Lee and Sampson 2009).

4.2.9.7 *Vocalization and Hearing*

The hearing sensitivities of Georgia Basin canary rockfish have not been studied. However, they produce low frequency sounds (lower than 900 Hz) (Sirovic and Demer 2009) and are believed to be low-frequency hearing generalists (Croll et al. 1999b).

4.2.9.8 *Status and Trends*

Canary rockfish were proposed for listing on April 23, 2009 (74 FR 18516). Canary rockfish were once considered common in Puget Sound, but have declined at a faster rate than any other rockfish species in the region (Holmberg et al. 1967; NMFS 2008c). Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990s, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Canary rockfish have been noted for being much less frequently caught in the Puget Sound and Georgia Basin region since 1965 (NMFS 2008c). The rate of decline for rockfish in Puget Sound has been estimated at ~3 percent annually for the period 1965 to 2007.

Because of a lack of systematic sampling targeting rare rockfish, absolute estimates of population size cannot be generated with accuracy. A rough estimate of the order of magnitude of population size for all rockfish is determined to be less than 10,000 in the Georgia Basin and less than 1,000 in Puget Sound Proper (Main Basin, Whidbey Basin, SPS and Hood Canal) (74 FR 18516). Current trends and levels of abundance contribute significantly to short- and long-term risk of extinction. No core population or subpopulation is at normal levels of abundance. The DPS is at all-time low levels of abundance for the past several years, exhibiting a disturbing trend in abundance levels for the past several years. Recent 10-year abundance in the DPS is low in all or nearly all spawning populations relative to earlier periods. Several historically large populations have been lost, resulting in a patchy distribution of isolated populations. Surveys indicate that by the 2000s, there were far fewer size classes represented and few older fish remain in the population. No published studies have compared genetic characteristics of canary rockfish from Puget Sound and outer coastal areas. Low abundance is likely resulting in reduced productivity. The south Puget Sound populations are no longer viable. The loss of either the north or south Puget Sound populations could result in a contraction of the range within the DPS. Although adults are known to move into other areas, there does not appear to be a strong refugial population anywhere in the DPS. The species is more mobile than many other rockfish species, which may help preserve genetic diversity by increasing connectivity among breeding populations.

4.2.9.9 *Natural Threats*

Predators of adult canary rockfish include yelloweye rockfish, lingcod, salmon, sharks, dolphins, seals (Antonelis and Fiscus 1980; Merkel 1957; Morejohn et al. 1978; Rosenthal et al. 1982) and possibly river otters (Stevens and Miller 1983). In addition, studies of the effect of climate variability on rockfish are rare, but all the studies performed to date suggest that climate plays an extremely important role in population dynamics (Drake and Griffen 2010). Although the mechanism by which climate influences the population dynamics of rockfish remains unknown, several authors have reported negative correlations between the warm water conditions associated with El Nino and the population dynamics of rockfish (Moser et al. 2000b). Field and Ralston (2005) reported that recruitment in all species of rockfish appeared to be correlated at large scales and hypothesized that such synchrony was the result of large-scale climatic phenomena. Tolimieri and Levin (2005) reported that bocaccio recruitment off of California is correlated with specific sets of climate patterns. These phenomena are also believed to affect the population dynamics of Georgia Basin canary rockfish and are assumed to have led to recruitment failures in the early- to mid-1990s.

4.2.9.10 *Anthropogenic Threats*

Overharvesting the primary cause of canary rockfish declines, but habitat loss is also important. Canary rockfish are considered overfished by the Pacific Fisheries Management Council and are not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). Habitat loss is also a factor in canary rockfish decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Low oxygen levels as well as pollutant, chemical, and nutrient loading are also considered significant threats to canary rockfish recovery (NMFS 2008c).

4.2.10 Puget Sound/Georgia Basin Yelloweye Rockfish

Yelloweye rockfish are very large rockfish that reach up to 3.5 feet (~1 m) in length and about 40 pounds (18 kg) in weight. They are orange-red to orange-yellow in color and may have black on their fin tips. Their eyes are bright yellow. Adults usually have a light to white stripe on the lateral line; juveniles have 2 light stripes, one on the lateral line and a shorter one below the lateral line.

4.2.10.1 *Distribution*

Yelloweye rockfish occur from Baja California to the Aleutian Islands, but are most common from central California to Alaska (Love et al. 2002). Individuals from the Puget Sound/Georgia Basin DPS occur throughout Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island, and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

4.2.10.2 *Habitat*

Yelloweye rockfish habitat varies based upon life stage. Larvae maintain a pelagic existence but as juveniles, move into shallow high relief rocky or sponge garden habitats (Eschmeyer et al. 1983; Love et al. 1991; Richards et al. 1985). Juveniles may also associate with floating debris or pilings (Lamb and Edgell 1986). As adults, yelloweye rockfish move in to deeper habitats. Individuals have been found in waters as deep as 549 m, but are generally found in waters of less than 180 m (Eschmeyer et al. 1983; Love et al. 2002). However, adults continue to associate with rocky, high relief habitats, particularly with caves and crevices, pinnacles, and boulder fields (Carlson and Straty 1981; Love et al. 1991; O'Connell and Carlisle 1993; Richards 1986; Yoklavich et al. 2000). Yelloweyes generally occur as individuals, with loose, residential aggregations infrequently found (Coombs 1979; DeMott 1983; Love et al. 2002). In the Puget Sound region, sport catch records from the 1970's indicate that Sucia Island and other islands of the San Juans as well as Bellingham Bay had the highest concentrations of catches (Delacy et al. 1972; Miller and Borton 1980).

4.2.10.3 *Reproduction*

Yelloweye rockfish are live bearers with internal fertilization. Copulation occurs between September and April, with fertilization taking place later as latitude increases (DeLacy et al. 1964; Hitz 1962; Lea et al. 1999; O'Connell 1987; Westrheim 1975; Wyllie Echeverria 1987). Puget Sound yelloweyes mate between winter and summer, giving birth from spring to late summer (Washington et al. 1978). Gestation lasts roughly 30 days (Eldridge et al. 2002). Although yelloweye rockfish were once believed to reproduce annually, evidence exists that indicate the potential for multiple births per year (MacGregor 1970; Washington et al. 1978). Females produce more eggs as they grow older and larger, with each individual producing roughly 300 eggs per year per gram of body weight (1.2 to 2.7 million eggs per year)(Hart 1973; MacGregor 1970). In addition, older females of several rockfish species may be capable of provisioning their offspring better than their younger counterparts, meaning that they may be more a more influential component in a given year's recruitment success (Sogard et al. 2008).

4.2.10.4 *Growth and Development*

Larvae are born at 4 to 5 mm in length and maintain a pelagic existence for the first 2 months of life, before moving to nearshore habitats and settling into rocky reef habitat at about 25 mm in length (DeLacy et al. 1964; Love et al. 2002; Matarese et al. 1989; Moser 1996a). Yelloweye growth is thought to vary by latitudinal gradient, with individuals in more northerly regions growing faster and larger. Year class strength appears to be most strongly linked to survival of the larval stage (Laidig et al. 2007). In general, sexual maturity appears to be reached by 50 percent of individuals by 15 to 20 years of age and 40 to 50 cm in length (Yamanaka and Kronlund 1997). As with other rockfish, yelloweyes can be long-lived (reported oldest age is 118 years)(Munk 2001). Maximum size has been reported as 910 cm, but asymptotic size in Alaskan waters for both males and females was estimated to be 690 cm and 659 to 676 cm along British Columbia (Clemens and Wilby 1961; Love et al. 2005; Rosenthal et al. 1982; Westrheim and Harling 1975; Yamanaka et al. 2006).

4.2.10.5 *Movement*

Individuals shift to deeper habitats as they age. Juveniles tend to begin life in shallow rocky reefs and graduate to deeper rocky habitats as adults. Once adult habitat is established, individuals tend to remain at a particular site (Coombs 1979; DeMott 1983; Love 1978).

4.2.10.6 *Foraging*

As with other rockfish species, yelloweye rockfish prey upon different species and size classes throughout their development. Larval and juvenile rockfish prey upon phyto- and zooplankton (Lee and Sampson 2009). Adult yelloweyes eat other rockfish (including members of their own species), sand lance, gadids, flatfishes, shrimp, crabs, and gastropods (Love et al. 2002; Yamanaka et al. 2006).

4.2.10.7 *Hearing and Vocalization*

The hearing sensitivities of Georgia Basin yelloweye rockfish have not been studied. However, they produce low frequency sounds (lower than 900 Hz) (Sirovic and Demer 2009) and are believed to be low-frequency hearing generalists (Croll et al. 1999b).

4.2.10.8 *Status and Trends*

Yelloweye rockfish were proposed for listing on April 23, 2009 (73 FR 18516). Yelloweye rockfish abundance has been variable in the Puget Sound region over the past 60 years, ranging from less than 1 percent to greater than 3 percent of samples, although Wallace (2001) documented large historical population in the Strait of Georgia. The latest samples have been historic lows in abundance. Perhaps more importantly, age classes appear to have been truncated to younger, smaller fish, severely hampering the ability of the species to recover from its primary cause of decline: overfishing (Berkeley et al. 2004).

Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Over the period of 1965 to 2007, it is estimated that rockfish species has declined by 3 percent per year.

The most recent estimate of yelloweye rockfish abundance in the Puget Sound region was 3,000 individuals, with low abundance through spawning areas (Palsson et al. 2008). Long generation times reflect intrinsic low productivity for this species. Productivity of the DPS is reduced further by the decreased abundance in larger, older females which are the most productive. South Puget Sound populations are no longer viable. The loss of these populations may eventually result in a contraction of the DPS range. There is no evidence of spatially structured populations in the DPS. Although larval dispersal through currents may increase connectivity, adult movement is limited. The truncation of size and age structure may reduce the viability of offspring.

4.2.10.9 *Natural Threats*

Interspecies competition, predators, and climactic regimes are the primary natural factors that depress yelloweye rockfish numbers. Copper and quillback rockfish may compete with yelloweye rockfish in Puget Sound for available resources (NMFS 2008c). Lingcod, killer whales, and Steller sea lions are likely predators of yelloweye and other rockfish species (Beaudreau and Essington 2007; Lance and Jeffries 2007; Love et al. 2002). Yelloweye and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Black 2009; Harvey 2005; Moser et al. 2000a). Oceanographic conditions (such as sea level anomalies and nearshore temperature conditions) appear to strongly influence the strength of each year's recruitment (Laidig et al. 2007). Rates of natural mortality have been reported to range from 2 to 4.6 percent annually (Wallace 2007; Yamanaka and Kronlund 1997).

4.2.10.10 *Anthropogenic Threats*

Overfishing is considered the primary cause of yelloweye rockfish decline throughout their range, including in Washington State and British Columbian waters (NMFS 2008c; Wallace 2007). Although commercial harvesting of the species has ended, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). It has been estimated that yelloweye rockfish have fallen 30 percent in abundance within one-third of a generation in the past few decades, an astonishing rate of decline.

Habitat loss is also a factor in yelloweye decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Anoxic conditions and chemical contamination are also considered threats to yelloweye rockfish recovery (NMFS 2008c).

4.2.11 Puget Sound/Georgia Basin Bocaccio

Bocaccio were proposed for listing on April 23, 2009 (74 FR 18516). Among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline (MacCall and He 2002b). This is likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climactic/oceanographic conditions (MacCall and He 2002b).

4.2.11.1 *Distribution and Populations*

The distribution of bocaccio includes Stepovac Bay on the Alaska Peninsula to Punta Blanca in central Baja California (Love et al. 2002). The bocaccio that occur in Puget Sound/Georgia Basin are listed as endangered (75 FR 22276). The listing includes bocaccio throughout Puget Sound, which encompasses all waters south of a line connecting Point Wilson on the Olympic Peninsula and Partridge on Whidbey Island; West Point on Whidbey Island, Deception Island, and Rosario Head on Fidalgo Island; and the southern end of Swinomish Channel between Fidalgo Island and McGlinn Island, and the Strait of Georgia, which encompasses the waters inland of Vancouver Island, the Gulf Islands, and the mainland coast of British Columbia.

According to genetic analyses, there may be three general population regions of bocaccio along the west coast: a Queen Charlotte Island population, one from Vancouver Island to Point Conception (CA), and a third south of Point Conception. The Puget Sound/Georgia Basin DPS is distinct from the southern and northern coastal DPSs. Relative to other rockfishes, bocaccio have a greater potential to move long distances, suggesting the DPS for bocaccio could encompass a greater area than a DPS for more sedentary species. It is likely that there is some regional overlap between the DPSs, but the accepted western boundary is located at the Victoria sill because of this oceanographic feature's ability to restrict larval dispersal. The Puget Sound DPS can further be divided into five Basins based on the distribution of each species, geographic conditions and habitat features: (1) The San Juan/Strait of Juan de Fuca Basin, (2) Main Basin, (3) Whidbey Basin, (4) South Puget Sound, and (5) Hood Canal. Prior to severe population reductions, bocaccio appeared to frequent the Tacoma Narrows in Washington State (DeLacy et al. 1964; Haw and Buckley 1971; Miller and Borton 1980).

4.2.11.2 *Habitat and Movement*

Preferred bocaccio habitat is largely dependent upon the life stage of an individual. Larvae and young juveniles tend to be found in deeper offshore regions (1 to 148 km offshore), but associated with the surface and occasionally with floating kelp mats (Emery et al. 2006; Hartmann 1987; Love et al. 2002). As individuals mature into older juveniles and adults, they transition into shallow waters and settle to the bottom, preferring algae-covered rocky, eelgrass, or sand habitats and aggregating into schools (Eschmeyer et al. 1983; Love et al. 1991). After a few weeks, fish move into slightly deeper waters of 18 to 30 m and occupy rocky reefs (Carr

1983; Eschmeyer et al. 1983; Feder et al. 1974; Johnson 2006; Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12 to 478 m, but tend to remain in shallow waters on the continental shelf (20 to 250 m), still associating mostly with reefs or other hard substrate, but may move over mud flats (Feder et al. 1974; Kramer and O'Connell 1995; Love et al. 2005; Love et al. 2006; Love et al. 2002; Love and York 2005). Artificial habitats, such as platform structures, also appear to be suitable habitat for bocaccio (Love and York 2006). Adults may occupy territories of 200 to 400 hectares, but can venture outside of this territory (Hartmann 1987). Adults are not as benthic as juveniles and may occur as much as 30 m above the bottom and move 100 m vertically during the course of a day as they move between different areas (Love et al. 2002; Starr et al. 2002).

4.2.11.3 *Reproduction*

Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54 to 61 cm total length), they produce 20,000 to 2.3 million eggs annually, with the number increasing as females age and grow larger (Echeverria 1987; Hart 1973; Love et al. 2002). However, either sex has been known to attain sexual maturity as small as 35 cm or 3 years of age and, in recent years as populations have declined, average age at sexual maturity may have declined as well (Echeverria 1987; Hart 1973; Love et al. 2002; MacCall 2002). Mating occurs between August and November, with larvae born between January and April (Love et al. 2002; Lyubimova 1965; MacCall and He 2002b; Moser 1967; Westrheim 1975; Wyllie Echeverria 1987).

4.2.11.4 *Growth*

Upon birth, bocaccio larvae measure 4 to 5 mm in length. These larvae move into pelagic waters as juveniles when they are 1.5 to 3 cm and remain in oceanic waters from 3.5 to 5.5 months after birth (usually until early June), where they grow at ~0.5 to 1 mm per day (Love et al. 2002; MacCall 2003; MacCall and He 2002b; Matarese et al. 1989; Moser 1967; Woodbury and Ralston 1991). However, growth can vary from year-to-year (Woodbury and Ralston 1991). Once individuals are 3 to 4 cm in length, they return to nearshore waters, where they settle into bottom habitats. Females tend to grow faster than males, but fish may take 5 years to reach sexual maturity (MacCall 2003). Individuals continue to grow until they reach maximum sizes of 91 cm, or 9.6 kg, at an estimated maximum age of 50 years (Andrews et al. 2005; Eschmeyer et al. 1983; Halstead et al. 1990; Love et al. 2002; Piner et al. 2006; Ralston and Ianelli 1998). However, individuals tend to grow larger in more northerly regions (Dark et al. 1983).

4.2.11.5 *Foraging*

Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates (Love et al. 2002). Pelagic juveniles consume fish larvae, copepods, and krill, while older, nearshore juveniles and adults prey upon rockfishes, hake, sablefish, anchovies, lanternfish, and squid (Love et al. 2002; Reilly et al. 1992).

4.2.11.6 *Vocalizations and Hearing*

The hearing sensitivities of Georgia Basin bocaccio have not been studied. However, they produce low frequency sounds (lower than 900 Hz) (Sirovic and Demer 2009) and are believed to be low-frequency hearing generalists (Croll et al. 1999b).

All fish have two sensory systems that are used to detect sound in the water including the inner ear, which functions very much like the inner ear found in other vertebrates, and the lateral line, which consists of a series of receptors along the body of the fish (Popper 2008c). The inner ear generally detects higher frequency sounds while the lateral line detects water motion at low frequencies (below a few hundred Hz) (Hastings et al. 1996b). A sound source produces both a pressure wave and motion of the medium particles (water molecules in this case), both of which may be important to fish. Fish detect particle motion with the inner ear. Pressure signals are initially detected by the gas-filled swim bladder or other air pockets in the body, which then re-radiate the signal to the inner ear (Popper 2008c). Because particle motion attenuates relatively quickly, the pressure component of sound usually dominates as distance from the source increases.

The lateral line system of a fish allows for sensitivity to sound (Hastings and Popper 2005). This system is a series of receptors along the body of the fish that detects water motion relative to the fish that arise from sources within a few body lengths of the animal. The sensitivity of the lateral line system is generally from below 1 Hz to a few hundred Hz (Coombs and Montgomery 1999; Popper and Schilt 2009). The only study on the effect of exposure to sound on the lateral line system (conducted on one freshwater species) suggests no effect on these sensory cells by intense pure tone signals (Hastings et al. 1996b).

While studies on the effect of sound on the lateral line are limited, the work of Hasting et al. (1996b) showing limited sensitivity to within a few body lengths and to sounds below a few hundred Hz, make the effect of the mid-frequency sonar of the action unlikely to affect a fish's lateral line system. Therefore, further discussion of the lateral line in this analysis in unwarranted. Broadly, fish can be categorized as either hearing specialists or hearing generalists (Scholik and Yan 2002a). Fish in the hearing specialist category have a broad frequency range with a low auditory threshold due to a mechanical connection between an air filled cavity, such as a swim bladder, and the inner ear.

Specialists detect both the particle motion and pressure components of sound and can hear at levels above 1 kilohertz (kHz). Generalists are limited to detection of the particle motion component of low-frequency sounds at relatively high sound intensities (Amoser and Ladich 2005). It is possible that a species will exhibit characteristics of generalists and specialists and will sometimes be referred to as an "intermediate" hearing specialist. For example, most damselfish are typically categorized as generalists, but because some larger damselfish have demonstrated the ability to hear higher frequencies expected of specialists, they are sometimes categorized as intermediate. Of the ESA-listed fish species with distributions occurring in the NWTT Action Area for which hearing sensitivities are known, none are hearing specialists.

4.2.11.7 *Status and Trends*

Since the 1980s, bocaccio have declined precipitously. Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Similar trends are seen in recreational landings from Puget Sound (WDF 1975-1986).

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 1998a). Bocaccio are long-lived fishes, taking several years to reach sexual maturity and becoming more fecund with age (Dorn 2002). As harvesting targeted the largest individuals available, bocaccio have become less capable of recovering population numbers (Love et al. 1998b). At present, in the complete absence of directed or bycatch fishing pressure, it is estimated that bocaccio populations would have to have frequent good recruitment to restrain their present decline (Tolimieri and Levin 2005). In addition, bocaccio reproduction appears to be characterized by frequent recruitment failures, punctuated by occasional high success years (Love et al. 1998b; MacCall and He 2002b). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes (it should be noted that 1999 and 2002 also appear to have been strong, but survivorship into maturity is still pending).

Recruitment success appears to be linked to oceanographic/climactic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998b; MacCall 1996; Moser et al. 2000b; Sakuma and Ralston 1995). Overall, bocaccio have the highest variability of recruitment of any rockfish studied to date (MacCall and He 2002b; Tolimieri and Levin 2005).

The rate of decline for rockfish in Puget Sound has been estimated at ~3 percent annually for the period 1965 to 2007. Various rebuilding estimates for bocaccio populations have predicted recovery, but require long periods (98 to 170 years) and assume no mortality from fishing (intentional harvests are closed, but bycatch still occurs) (MacCall 2008; MacCall and He 2002a; NMFS 2008c). Although a population increase has been detected more recently, it is likely that this is due to a very strong recruitment event in 1999. Because of a lack of systematic sampling targeting rare rockfish, absolute estimates of population size cannot be generated with accuracy.

A rough estimate of the order of magnitude of population size for all rockfish is determined to be less than 10,000 in the Georgia Basin and less than 1,000 in Puget Sound Proper (Main Basin, Whidbey Basin, SPS and Hood Canal) (74 FR 18516). Given the species composition data and abundance estimates from other species (Palsson et al. 2009), it is estimated the Puget Sound DPS of bocaccio may be as few as 45 individuals. Sporadic recruitment results in an inconsistent growth trajectory. Even in absence of exploitation, bocaccio have a low intrinsic growth rate, 1.01 which suggests that populations of bocaccio are not capable of supporting continuous harvest.

4.2.11.8 *Natural Threats*

Interspecies competition, predators, and climactic regimes are the primary natural factors that depress bocaccio numbers. Copper and quillback rockfish may compete with bocaccio in Puget Sound for available resources (NMFS 2008c). King salmon, lingcod, terns and other seabirds, harbor seals, and Steller sea lions are known predators of bocaccio and other rockfish species (Beaudreau and Essington 2007; Lance and Jeffries 2007; Love et al. 2002). Bocaccio and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Harvey 2005; Moser et al. 2000a).

4.2.11.9 *Anthropogenic Threats*

Although overfishing is the primary reason for bocaccio being proposed as a listed species, bycatch and habitat loss are also human-related factors that have likely led to bocaccio decline. Although a frequent species captured in fisheries during the late 1970's, bocaccio were not recorded from any recreational surveys from 1996 to 2007 (Palsson et al. 2008; WDF 1975-1986). Apart from commercial fishing, recreational fishing (even catch-and-release) appears to incur significant mortality on bocaccio and other rockfishes (Schroeder and Love 2002). The species is considered overfished by the Pacific Fisheries Management Council and is not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008).

Habitat loss is also a factor in bocaccio decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Loss of kelp, which is valuable to juvenile fish recruitment, as well as anoxic conditions, exacerbate habitat loss (NMFS 2008c).

4.2.12 Southern DPS of Pacific Eulachon

Eulachon, *Thaleichthys pacificus,* (commonly called smelt, candlefish, or hooligan) are a small, anadromous fish from the eastern Pacific Ocean. The Southern DPS of eulachon was listed as threatened on March 18, 2010 (75 FR 13012). They are distinguished by the large canine teeth on the vomer, a bone in the roof of the mouth, and 18 to 23 rays in the anal fin. Like Pacific salmon they have an adipose fin; it is sickle-shaped. The paired fins are longer in males than in females. All fins have well-developed breeding tubercles (raised tissue "bumps") in ripe males, but these are poorly developed or absent in females. Adult coloration is brown to blue on the back and top of the head, lighter to silvery white on the sides, and white on the ventral surface; speckling is fine, sparse, and restricted to the back. They feed on plankton but only while at sea.

Eulachon typically spend 3 to 5 years in saltwater before returning to freshwater to spawn from late winter through mid-spring. During spawning, males have a distinctly raised ridge along the middle of their bodies. Eggs are fertilized in the water column. After fertilization, the eggs sink and adhere to the river bottom, typically in areas of gravel and coarse sand. Most eulachon adults die after spawning. Eulachon eggs hatch in 20 to 40 days. The larvae are then carried downstream and are dispersed by estuarine and ocean currents shortly after hatching. Juvenile eulachon move from shallow nearshore areas to mid-depth areas. Within the Columbia River Basin, the major and most consistent spawning runs occur in the mainstem of the Columbia River as far upstream as the Bonneville Dam, and in the Cowlitz River.

4.2.12.1 *Distribution*

The Southern DPS of Pacific eulachon consists of populations spawning in rivers south of the Nass River in British Columbia, Canada, to, and including, the Mad River in California (75 FR 13012). There are 10 subpopulations: Klamath River, Columbia River, Fraser River, Knight Inlet, Kingcome Inlet, Rivers inlet, Dean Channel, Gardner Canal, Douglas Channel and Skeena River.

Eulachon is an anadromous species that spawns in the lower portions of certain rivers draining into the northeastern Pacific Ocean ranging from Northern California to the southeastern Bering Sea in Bristol Bay, Alaska (NMFS 2010a; Schultz and DeLacy 1935). Eulachon have been described as “common” in Grays Harbor and Willapa Bay on the Washington coast, “abundant” in the Columbia River, “common” in Oregon’s Umpqua River, and “abundant” in the Klamath River in northern California. They have been described as “rare” in Puget Sound and Skagit Bay in Washington; Siuslaw River, Coos Bay, and Rogue River in Oregon; and Humboldt Bay in California (Emmett et al. 1991a). However, Hay and McCarter (2000) and Hay (2002) identified 33 eulachon spawning rivers in British Columbia and 14 of these were classified as supporting regular yearly spawning runs.

4.2.12.1.1 Distribution in the NWTT Action Area

The Southern DPS of Pacific eulachon may be present in the inland and offshore waters of the NWTT Action Area.

4.2.12.2 *Vocalizations and Hearing*

We do not have specific information on hearing in eulachon, but we assume that they are hearing generalists whose hearing sensitivities would be similar to salmon. Species in the family Salmonidae have similar auditory systems and hearing sensitivities (Popper 1977; Popper et al. 2007; Wysocki et al. 2007). Most of the data available on this group 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 eulachon 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).

4.2.12.3 *Status and trends*

Spawning runs from California to southeastern Alaska have declined by 97.7 percent in the past 20 years relative to the prior period. The Columbia River population, historically the largest, has experienced a severe decline in eulachon returns. A loss of the Columbia River eulachon population and any dependent coastal spawning populations could represent the loss of the species throughout its range in the United States, as well as the loss of substantial portion of its historical range. The most recent estimates from the Columbia River (2014) indicate an increase in eulachon spawning stock biomass from previous years (NMFS 2014). The populations in the Klamath River, Mad River, Redwood Creek and the Sacramento River are likely extirpated or nearly so.

4.2.12.4 *Natural Threats*

Eulachon have numerous avian predators including harlequin ducks, pigeon guillemots, common murres, mergansers, cormorants, gulls, and eagles. Marine mammals such as humpback whales, orcas, dolphins, Steller sea lions, California sea lions, northern fur seals, harbor seals, and beluga whales are known to feed on eulachon. During spawning runs, bears and wolves have been observed consuming eulachon. Fishes that prey on eulachon include white sturgeon, spiny dogfish, sablefish, salmon sharks, arrowtooth flounder, salmon, Dolly Varden charr, Pacific halibut, and Pacific cod. In particular, eulachon and their eggs seem to provide a significant food source for white sturgeon in the Columbia and Fraser Rivers (75 FR 13012).

4.2.12.5 *Anthropogenic Threats*

Southern eulachon are primarily threatened by increasing temperatures in the marine, coastal, estuarine, and freshwater environments of the Pacific Northwest that are at least causally related to climate change; dams and water diversions, water quality degradation, dredging operations in the Columbia and Fraser Rivers; commercial, recreational, and subsistence fisheries in Oregon and Washington that target eulachon; and bycatch in commercial fisheries. The high lipid content of eulachon suggests they are susceptible to absorption of lipophilic organic contaminants.

Contaminants considered of most concern include: 1) synthetic chlorinated organic chemicals, such as hexachlorobenzene, DDTs, and the polychlorinated biphenyls (PCBs); 2) polycyclic aromatic hydrocarbons (PAHs) from petroleum and creosoted pilings; 3) dioxins and a host of other organic compounds; 4) metals such as mercury, arsenic, and lead; and 5) endocrine-disrupting compounds and new toxics like PBDE.

Eulachon are particularly vulnerable to capture in shrimp fisheries in the United States and Canada as the marine areas occupied by shrimp and eulachon often overlap. In Oregon, the bycatch of various species of smelt (including eulachon) has been as high as 28 percent of the total catch of shrimp by weight (Hannah and Jones 2007). There are directed fisheries in Alaska state waters for eulachon in Upper Cook Inlet, the Copper River area, and in southeast Alaska. There has been little commercial activity in recent years, due to either lack of interest or closures resulting from concerns over diminished spawning runs, but there is potential for substantial amounts of harvest (Ormseth and Vollenweider 2007).

4.2.13 Chinook Salmon

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.

4.2.13.1 *Species Distribution*

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 2000a). Because of similarities in the life history and threats to the survival and recovery of the Chinook salmon "species" (as that term is defined in section 3 of the ESA) or evolutionary significant units (ESUs) that are included in this Opinion, we summarize the threats to Chinook salmon and their hearing sensitivity generally. Then we separately discuss specific information on their listing status, population status and trends, and impacts that are not shared for each of these species.

Chinook salmon distribute in the North Pacific Ocean north of about 40º North latitude where they may remain for 1 to 6 years, although 2 to 4 years are more common. Although salmon generally occur near the surface (within 8 to 10 meters of the surface), Chinook salmon have been caught at depths up to 110 meters.

4.2.13.1.1 Occurrence in the NWTT Action Area

All Chinook salmon ESUs may occur in the Offshore portion of the NWTT Action Area. Additionally, near shore critical habitat for the Puget Sound ESU is located in the Inland Waters portion of the Action Area. The Puget Sound Chinook salmon is the only ESU occurring in both the Inland Waters and Offshore portions of the Action Area. The remaining ESUs will only occur in the Offshore Area. Chinook salmon may also migrate and enter Behm Canal.

4.2.13.2 ***Reproduction***

The general Chinook salmon life cycle spans fresh and marine waters, with one reproductive event per adult (Chinook salmon are semelparous and die after spawning). Spawning migrations generally occur in the spring and fall, although the precise timing of spawning migrations and spawning varies across populations and can vary within populations. Temperature and stream flow can significantly influence the timing of upstream migrations and spawning, and the selection of spawning habitat (Geist et al. 2009; Hatten and Tiffan 2009). For Klamath River Chinook, temperatures above 21.9º C (mean average body temperature), 20.6º C (mean minimum daily body temperature), or 23.1º C (mean maximum daily body temperature) completely inhibited upstream spawning migration in rivers; these values are close to the upper lethal limits for this and other salmonid species (Strange 2010). A general latitudinal cline is apparent across the species range with spawning typically occurring earlier in the spring/summer at northern latitudes and later in southern latitudes (Healey 1991).

On the spawning grounds, mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Once fertilization occurs, female Chinook salmon bury the eggs in nests, termed "redds", and guard the nests until their death, which generally occurs a couple days later to a couple weeks after spawning. A female generally deposits eggs in more than one depression within a redd, excavating stream rock as she moves upstream, increasing the size of her redd until all eggs are deposited.

Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior (Roni and Quinn 1995). Generally, ocean-type salmon are at sea longer than their stream-type counterparts and tend to be larger in size at spawning. Body size can be important in determining reproductive success in terms of nest selection and mating competition (Foote 1990). Chinook salmon age at maturity ranges from 1 to 7 years with most returning to spawn between 3 and 4 years of age.

4.2.13.3 ***Habitat***

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 can prolong incubation. Incubation generally takes a couple of months or more.

Alevin (also called "yolk-sac" fry) remain buried until their yolk-sac is absorbed, at which time they become free swimming fry. Egg to fry survival can also vary widely across basins, years, and habitat conditions within a basin. In general, the survival of eggs and alevin, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawn timing, floods, redd and spawner density, and water temperatures.

Once emerged, fry behavior varies among populations and among individuals within races. Some juvenile Chinook salmon rear in freshwater for a few weeks to a few years, others move immediately downstream to coastal waters where they rear in estuaries for a few weeks to months, while others migrate directly to ocean waters. Stream-type Chinook salmon do not migrate to sea until the spring following emergence, and ocean-type Chinook salmon migrate to the ocean within their first year. Generally, most fry move at night probably to reduce detection by predators, although some fish will move downstream during daylight. Not all movement is volitional as stream flows often displace fry to downstream areas after emergence. Density-dependent factors such as space, prey, or stream flows may influence the outmigration behavior of individual juvenile Chinook salmon.

While in fresh water, juvenile Chinook salmon are often found in the lower reaches of a river near its estuary, where they inhabit river margins in areas of shallow water, near woody debris, or other areas of low water velocity. As juveniles grow in size, they tend to move away from the shoreline to deeper waters where the velocity is higher (Healey 1991). The transformation from the freshwater fry/parr juvenile stage to smolt involves multiple physiological changes including increases in: body silvering, hypoosmotic regulatory capability, salinity tolerance and preference, growth rate, oxygen consumption, ammonia production, endocrine activity (e.g., activation of thyroid, interregnal and pituitary growth hormone cells), and gill Na^+, K^+-ATPase activity. Several factors can affect the smoltification process, not only at the interface between freshwater and saltwater, but higher in the watershed as the process of transformation begins long before fish enter saltwater, including exposure to heavy metals and elevated water temperatures (Wedemeyer et al. 1980).

Life at sea varies according to population, race, and age-class. Chinook salmon tend to remain at sea between 1 and 6 years, with most fish returning to freshwater after 2 to 4 years at sea. Fishery catches indicate that ocean- and stream-type fish exhibit divergent migratory pathways while in the ocean (Healey 1983; Healey 1991). Ocean-type Chinook salmon tend to be found along the coastline, whereas stream-type Chinook salmon are found in the open ocean far from the coast (Healey 1983; Healey 1991). Juvenile Chinook along the Pacific northwest, Canada, and Alaska tend to remain within roughly 55 km (most within 28 km) of the coast (NPFMC 1990; PFMC 2000b). However, Chinook generally remain within 320 km of the coast (NPFMC 1990). Concentrations are known to occur around transient upwelling features (PFMC 2000b).

Individuals hatching in rivers south of Cape Blanco, Oregon tend to stay south of this point, while those outmigrating north of this point tend to move north into the Gulf of Alaska through coastal migratory corridors (PFMC 2000b).

Generally, Chinook salmon outmigrants (termed smolts) are about 5 to 13 cm long when they enter saline (often brackish) waters. The process of smoltification is physiologically demanding, involving osmoregulation—the maintenance of osmotic pressure as the fish enters waters of increased salinity, which is necessary to maintain body fluid concentration and composition to maintain homeostasis. Smaller fish tend to remain closer to shore, while larger fish will enter marine waters. The distribution of fish in this stage does not appear to be correlated with salinities, as most fish can survive immediate transfer from freshwater to saltwater (~30 ppm salinity; (Healey 1991). Although temperature ranges vary from 1° to 15° C, few individuals are found in waters below 5° C (MBC 1987; PFMC 2000b). Chinook salmon originating from the same freshwater region have similar age-dependent marine distributions (even if associated with different runs), which are distinct from the distributions of Chinook from other freshwater regions (Weitkamp 2009). This distribution does not appear to be influenced by oceanographic variability (Weitkamp 2009). Older individuals appear to disperse more broadly than their younger counterparts (Weitkamp 2009). Adults do not appear to exhibit a preference in substrates and may be found down to 250 m (Beauchamp et al. 1983).

4.2.13.4 ***Feeding***

Chinook salmon feed on a variety of prey organisms depending upon life stage. Adult oceanic Chinook salmon eat small fish, amphipods, and crab megalops (Brodeur et al. 2010; Healey 1991). Fish, in particular herring, make up the largest portion of an adult Chinook salmon's diet. In estuaries, Chinook salmon smolts tend to feed on the chironomid larvae and pupae *Daphnia*, *Eogammarus*, *Corphium* and *Neomysis*, as well as juvenile herring, sticklebacks and other small fish. In freshwater, Chinook salmon juveniles feed on adult and larval insects including terrestrial and aquatic insects such as dipterans, beetles, stoneflies, chironomids, and plecopterans (Healey 1991). During the first year of marine life, Puget Sound Chinook transition from nearshore foraging on insects and amphipods during June to offshore foraging on crab larvae and fish, such as herring during July through September (Duffy et al. 2010a).

4.2.13.5 ***Hearing***

Based on the information available, we assume that the Chinook 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).

4.2.13.6 *Natural Threats*

Chinook salmon are exposed to high rates of natural predation during freshwater rearing and migration stages, as well as during ocean migration. In general, Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest may have reduced the survival of some salmon populations; 10 percent of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011e). This estimate may be higher, with Sebring et al. (2013) finding that more than eight million released fall-run Chinook salmon (up to 35 percent of available salmon) are consumed by cormorants and terns. 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 2011e).

4.2.13.7 *Anthropogenic Threats*

Chinook salmon have declined under the combined effects of fishery over-harvest; 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 Chinook 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 (Buhle et al. 2009).

Population declines have resulted from several human-mediated causes, but the greatest negative influence has likely been the establishment of waterway obstructions such as dams, power plants, and sluiceways for hydropower, agriculture, flood control, and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs as a result. While some of these barriers remain, others have been reengineered, renovated, or removed to allow for surviving runs to access former habitat, but success has been limited. These types of barriers alter the natural hydrograph of basins, both upstream and downstream of the structure, and significantly reduce the availability and quality of spawning and rearing habitat (Hatten and Tiffan 2009). Many streams and rivers, particularly in urban or suburban areas, suffer from streamside development, which contributes sediment, chemical pollutants from pesticide applications and automobile or industrial activities, altered stream flows, loss of streamside vegetation and allochthonous materials to name a few. These factors can directly cause mortality, reduce reproductive success, or affect the health and fitness of all salmon life stages.

Artificial propagation of hatchery fish has had profound consequences on the viability of some natural salmon populations, but there are potential benefits to the artificial production of salmon as well. Potential adverse effects of artificial propagation include: a decline in the natural population from the taking of wild brood stock for artificial propagation, the genetic erosion of populations (introgression, hybridization), an increase incidence of disease to and increased rates of competition with and predation on naturally spawned salmon populations. Potential benefits to artificial propagation include the bolstering of the numbers of naturally spawning fish, the conservation of genetic resources, and guarding against the catastrophic loss of naturally spawned populations at critically low abundance levels.

Fishing for salmon has also negatively impacted salmon populations. Fishing reduces the number of individuals within a population and can lead to uneven exploitation of certain populations and size classes (Mundy 1997; Reisenbichler 1997). Targeted fishing of larger individuals results in excluding the most fecund individuals from spawning (Reisenbichler 1997). Genetic changes that promote smaller body sizes have occurred in heavily exploited populations in response to size-selective harvest pressures (Mundy 1997; Reisenbichler 1997; Swain et al. 2007). Age at maturity can also be accelerated by fishing pressure (Reisenbichler 1997). Pacific salmon species are exposed to a number of contaminants throughout their range and life history cycle.

Exposure to pollution is also of significant concern for all life stages, but is likely particularly significant for freshwater life stages. Organic pollutants, especially PCBs, DDT and its congeners, pesticides, and endocrine disruptors are particularly concerning. These chemicals can inhibit smell, disrupt reproductive behavior and physiology, impair immune function, and lead to mortality through impairment of water balance when traveling between fresh- and saltwater systems (Varanasi et al. 1993a; Varanasi et al. 1993b). Diffuse and extensive population centers contribute increase contaminant volumes and variety from such sources as wastewater treatment plants and sprawling development. Urban runoff from impervious surfaces and roadways often contains oil, copper, pesticides, PAHs, and other chemical pollutants and flow into surface waters. Point and nonpoint pollution sources entering rivers and their tributaries affect water quality in available spawning and rearing habitat for salmon. Juvenile salmonids that inhabit urban watersheds often carry high contaminant burdens, which is partly attributable to the biological transfer of contaminants through the food web (Brown et al. 1985; Stein et al. 1992; Varanasi et al. 1993a).

4.2.13.8 ***Chinook Salmon ESUs***

Each Chinook salmon ESU is treated as a separate species under the ESA (NMFS 2005c). Of these listed Chinook salmon ESUs, two are endangered (Sacramento River winter-run and Upper Columbia River spring-run) and seven are threatened (Snake River spring/summer-run, Snake River fall-run, Central Valley spring-run, California coastal, Puget Sound, Lower Columbia

River, and Upper Willamette River) (70 FR 37160). The distribution, as well as the status and trends, of the Chinook salmon ESUs considered in this Opinion are discussed below.

4.2.13.8.1 Puget Sound Chinook Salmon ESU

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308); that status was reaffirmed on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Puget Sound (PS) Chinook includes all naturally spawned populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Straits of Juan De Fuca from the Elwha River, eastward, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington, and progeny of 26 artificial propagation programs. The PS-TRT identified 22 historical populations, grouped into five major geographic regions, based on consideration of historical distribution, geographic isolation, dispersal rates, genetic data, life history information, population dynamics, and environmental and ecological diversity. The NMFS adopted the Shared Strategy for Puget Sound locally-developed listed species recovery plan for PS Chinook salmon in 2007 (Shared Strategy for Puget Sound 2007; SSPS 2007a).

Indices of spatial distribution and diversity have not been developed at the population level. Based on a Shannon Diversity Index at the ESU level, diversity is declining (due primarily to the increased abundance of returns to the Whidbey Basin region) for both distribution among populations and among regions (Ford 2011b). Overall, the new information on abundance, productivity, spatial structure and diversity since the 2005 status review does not indicate a change in the biological risk category (Ford 2011b).

Abundance and Productivity: No trend was notable for the total ESU escapements; while trends vary from decreasing to increasing among populations. Natural-origin pre-harvest recruit escapements remained fairly constant from 1985 to 2009. Returns (pre-harvest run size) from the natural spawners were highest in 1985, declined through 1994, remained low through 1999, increased in 2000 and again in 2001, and have declined through 2009, with 2009 having the lowest returns since 1997. Median recruits per spawner for the last 5-year period (brood years 2002 through 2006) is the lowest over any of the 5-year intervals. Many of the habitat and hatchery actions identified in the PS Chinook salmon recovery plan are likely to take years or decades to be implemented and to produce significant improvements in natural population attributes, and these trends are consistent with these expectations (Ford 2011b).

NMFS concluded in 1998 (Meyers et al. 1998), 2005 (Good et al. 2005a), and 2011 (Ford 2011b) that the PS Chinook ESU was likely to become endangered in the foreseeable future. In the first status review, the PS Chinook BRT estimated the total PS Chinook salmon run size in the early 1990s to be approximately 240,000 Chinook, with the vast majority as hatchery-origin. Based on current estimates, 67,000 of those fish were naturally produced Chinook salmon (Unpublished data, Norma Sands, NWFSC, March 5, 2010). ESU escapement increased to 45,214 (2000 to 2004); but has since declined to 37,409, during the most recent status review (2005 to 2009), and 30,955 from 2008 to 2012 (Table 19 and Table 20).

Table 19. Abundance–five-year geometric means for adult (age 3+) natural (natural and hatchery origin) and natural origin only spawners for the ESU with ranges and medians given for the populations (Ford 2011b).

Year Range	Natural Escapement			Natural Origin Escapement		
	ESU	Population Range	Population Median	ESU	Population Range	Population Median
1985-1989	36,750	48-8,276	770	28,601	30-7,965	725
1990-1994	26,094	101-5,511	395	19,511	20-5,304	381
1995-1999	28,981	104-6,729	479	19,011	18-5,982	380
2000-2004	45,214	202-12,109	999	32,794	71-11,678	430
2005-2009	37,409	81-10,345	909	25,848	44-9,724	482

Table 20. Average abundance estimates for PS Chinook salmon natural- and hatchery-origin spawners 2008-2012 (unpublished data, Mindy Rowse, NWFSC, Jan. 28, 2015; as cited in NMFS 2015).

Population Name	Natural-origin Spawners[a]	Hatchery-origin Spawners[b]	% Hatchery Origin	Minimum Viability Abundance[c]	Expected Number of Outmigrants[d]
N. Fork Nooksack	171	1,066	86.18%	16,000	98,960
S. Fork Nooksack	111	264	70.40%	9,100	30,000
Lower Skagit	1,343	63	4.48%	16,000	112,480
Upper Skagit	6,545	135	2.02%	17,000	534,400
Cascade	316	9	2.77%	1,200	26,000
Lower Sauk	372	0	0.00%	5,600	29,760
Upper Sauk	700	18	2.51%	3,000	57,440
Suiattle	335	2	0.59%	600	26,960
N. Fork Stillaguamish	533	393	42.44%	17,000	74,080
S. Fork Stillaguamish	57	8	12.31%	15,000	5,200
Skykomish	1,534	510	24.95%	17,000	163,520
Snoqualmie	804	231	22.32%	17,000	82,800
Sammamish	165	1,249	88.33%	10,500	113,120
Cedar	776	168	17.80%	11,500	75,520
Duwamish/Green	599	890	59.77%	17,000	119,120
White	957	674	41.32%	14,200	130,480
Puyallup	482	596	55.29%	17,000	86,240
Nisqually	552	1,404	71.78%	13,000	156,480
Skokomish	253	1,055	80.66%	12,800	104,640
Mid-Hood Canal	58	171	74.67%	11,000	18,320
Dungeness	115	139	54.72%	4,700	20,320
Elwha	171	898	84.00%	15,100	85,520
ESU Average[e]	18,127	11,089	37.96%		2,337,280

[a] Five-year geometric mean of post-fishery natural-origin spawners.

[b] Five-year geometric mean of post-fishery hatchery-origin spawners.

[c] Ford 2011

[d] Expected number of outmigrants=Total spawners*40% proportion of females*2,000 eggs per female*10% survival rate from egg to outmigrant

[e] ESU Average is calculated as the geometric mean of the annual totals and not the sum of the geometric means.

Juvenile PS Chinook abundance estimates come from escapement data, the percentage of females in the population, and fecundity. Fecundity estimates for the ESU range from 2,000 to 5,500 eggs per female, and the proportion of female spawners in most populations is approximately 40 percent of escapement. By applying a conservative fecundity estimate (2,000 eggs/female) to the expected female escapement (both natural-origin and hatchery-origin spawners – 14,608 females), the ESU is estimated to produce approximately 23.4 million eggs annually. Smolt trap studies have researched egg to migrant juvenile Chinook salmon survival rates in the following Puget Sound tributaries: Skagit River, North Fork Stillaguamish River, South Fork Stillaguamish River, Bear Creek, Cedar River, and Green (Beamer et al. 2000; Griffith et al. 2004; Seiler et al. 2002; Seiler et al. 2004; Seiler et al. 2005; Volkhardt et al. 2005). The average survival rate in these studies was 10 percent, which corresponds with those reported by Healey (1991). With an estimated survival rate of 10 percent, the ESU should produce roughly 2.34 million natural outmigrants annually.

<u>Limiting Factors</u> include (NOAA Fisheries 2011; Shared Strategy for Puget Sound 2007):

- Degraded nearshore and estuarine habitat: Residential and commercial development has reduced the amount of functioning nearshore and estuarine habitat available for salmon rearing and migration. The loss of mudflats, eelgrass meadows, and macroalgae further limits salmon foraging and rearing opportunities in nearshore and estuarine areas.
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, and water quality have been degraded for adult spawning, embryo incubation, and rearing as a result of cumulative impacts of agriculture, forestry, and development.
- Anadromous salmonid hatchery programs: Salmon and steelhead released from Puget Sound hatcheries operated for harvest augmentation purposes pose ecological, genetic, and demographic risks to natural-origin Chinook salmon populations.
- Salmon harvest management: Total fishery exploitation rates have decreased 14 to 63 percent from rates in the 1980s, but weak natural-origin Chinook salmon populations in Puget Sound still require enhanced protective measures to reduce the risk of overharvest in Chinook salmon-directed fisheries.

4.2.13.8.2 Lower Columbia River Chinook Salmon ESU

Lower Columbia River Chinook salmon were listed as threatened on March 24, 1999 with the threatened status reaffirmed on June 28, 2005.

<u>Spatial Structure and Diversit</u>y: Lower Columbia River (LCR) Chinook includes all naturally-spawned populations of Chinook salmon in the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon east of the Hood River and the White Salmon River; the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River; and progeny of seventeen

artificial propagation programs.[8] LCR Chinook populations exhibit three different life history types base on return timing and other features: fall-run (a.k.a. "tules"), late-fall-run (a.k.a. "brights"), and spring-run. The Willamete/Lower Columbia Technical Recovery Team (WLC-TRT) identified 32 historical populations of LCR Chinook salmon— seven in the coastal subregion, six in the Columbia Gorge, and 19 in the Cascade Range. Spatial structure has been substantially reduced in several populations. Low abundance, past broodstock transfers and other legacy hatchery effects, and ongoing hatchery straying may have reduced genetic diversity within and among LCR Chinook salmon populations. Hatchery-origin fish spawning naturally may also have reduced population productivity (Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a). Out of the 32 populations that make up this ESU, only the two late-fall runs, the North Fork Lewis and Sandy, are considered viable. Most populations (26 out of 32) have a very low probability of persistence over the next 100 years (and some are extirpated or nearly so) (Ford 2011b; Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a). Five of the six strata fall significantly short of the WLC-TRT criteria for viability; one stratum, Cascade late-fall, meets the WLC TRT criteria (NMFS 2013).

Abundance and Productivity: A&P ratings for LCR Chinook salmon populations are currently "low" to "very low" for most populations, except for spring Chinook salmon in the Sandy River, which are "moderate" and late-fall Chinook salmon in North Fork Lewis River and Sandy River, which are "very high" (NMFS 2013). Low abundance of natural-origin spawners (100 fish or fewer) has increased genetic and demographic risks. Other LCR Chinook salmon populations have higher total abundance, but several of these also have high proportions of hatchery-origin spawners. Particularly for tule fall Chinook salmon populations, poor data quality prevents precise quantification of population abundance and productivity; data quality has been poor because of inadequate spawning surveys and the presence of unmarked hatchery-origin spawners (Ford 2011b).

In 1998, NMFS assessed the abundance in smaller tributary streams in the range of the species to be in the hundreds of fish (Meyers et al. 1998). Larger tributaries (e.g., Cowlitz River basin) contained natural runs of Chinook salmon ranging in size from 100 to almost 1,000 fish. In 2005, NMFS calculated adult abundance using the geometric mean of natural-origin spawners in the five years previous to 2003 (Good et al. 2005a). In 2005, NMFS estimated the LCR Chinook salmon abundance at approximately 14,130 fish (Good et al. 2005a). Data that are more recent

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[8] In 2009, the Elochoman tule fall Chinook salmon program was discontinued and four new fall Chinook salmon programs have been initiated. In 2011, NMFS recommended removing the Elochoman program from the ESU and adding the new programs to the ESU NMFS. 2011a. 5-year review: summary and evaluation of Lower Columbia River Chinook, Columbia River chum, Lower Columbia River coho, and Lower Columbia River steelhead. National Marine Fisheries Service, editor, Portland, Oregon.

place the abundance of naturally produced LCR Chinook salmon at approximately 13,594 spawners (Table 21).

Table 21. Abundance estimates for LCR Chinook salmon populations (ODFW 2014a; WDFW 2014).

Stratum (Run)	Population	Years	Total	HOR(1)	NOR(2)
Coastal (Fall)	Youngs Bay	2012-13	6,686	6,516	170
	Grays/Chinook	2008-2012	319	106	213
	Big Creek	2012-13	1,096	1,041	55
	Elochoman/Skamokowa	2008-2012	1,091	628	463
	Clatskanie	2012-13	3,205	2,999	206
	Mill/Abernathy/Germany	2008-2012	817	302	515
	Scappoose		na	na	na
Cascade (Fall)	Lower Cowlitz	2008-2012	617	0	617
	Upper Cowlitz	2008-2012	2,670	2,204	466
	Toutle	2008-2012	na	na	na
	Coweeman	2008-2012	1,080	891	189
	Kalama	2008-2012	5,420	4,198	1,222
	Lewis	2004-2008	1,060	0	1,060
	Washougal	2012-13	321	261	60
	Clackamas	2008-2012	3,050	1,216	1,834
	Sandy	2012-13	714	146	568
Columbia Gorge (Fall)	Lower gorge	2003-2007	146	Unknown	146
	Upper gorge	2008-2012	827	Unknown	345
	Hood		na	na	na
	White Salmon	2008-2012	1,522	Unknown	1,524
Cascade (Late Fall)	Sandy		na	na	na
	North Fork Lewis	2008-2012	843	134	709
Cascade (Spring)	Upper Cowlitz	2005-2009	589	0	589
	Cispus		na	na	na
	Tilton		na	na	na
	Toutle		na	na	na
	Kalama	2006-2011	606	0	606
	North Fork Lewis	2007-2012	199	0	199
	Sandy	2008-2012	4,064	2,226	1,838
Gorge (Spring)	White Salmon		na	na	na
	Hood		na	na	na
Total			36,942	22,868	13,594

(1) Hatchery Origin (HOR) spawners.

(2) Natural Origin (NOR) spawners.

The Oregon and Washington recovery plans rate all but three Chinook populations as low to very low for abundance and productivity (Lower Columbia Fish Recovery Board 2010b; ODFW 2010b). The range of abundance recommended for recovery is from 300 (Kalama spring-run) to 7,300 (North Fork Lewis late fall-run). Current abundance estimates from WDFW and ODFW suggest that only five populations are at or have exceeded abundance goals, and for one of these (the White Salmon), we do not know what portion of the spawners are hatchery origin.

NWFSC publishes juvenile abundance estimates each year in the annual memorandum estimating percentages of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time; however, the average outmigration for the years 2010 through 2014 is shown in Table 22 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Table 22. Average estimated outmigration for ESA-listed LCR Chinook salmon (2010-2014).

Origin	Outmigration
Natural	13,271,270
Listed hatchery intact adipose	1,070,253
Listed hatchery adipose clip	35,337,495

Limiting Factors include (NMFS 2013; NOAA Fisheries 2011):

- Degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development.
- Reduced access to spawning and rearing habitat mainly as a result of tributary hydropower projects
- Hatchery-related effects
- Harvest-related effects on fall Chinook salmon
- An altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity
- Reduced access to off-channel rearing habitat in the lower Columbia River
- Reduced productivity resulting from sediment and nutrient-related changes in the estuary
- Juvenile fish strandings that result from ship wakes
- Contaminants affecting fish health and reproduction

4.2.13.8.3 California Coastal Chinook Salmon ESU

California Coastal Chinook salmon were listed as threatened on September 16, 1999 (64 FR 50393), and they retained their threatened status on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: The California Coastal (CC) Chinook salmon ESU was historically comprised approximately 38 Chinook salmon populations[9] (Bjorkstedt et al. 2005; Spence et al. 2008). Many of these populations (about 21) were independent or potentially independent, meaning they had a high likelihood of surviving for 100 years absent anthropogenic impacts. The remaining populations were likely more dependent upon immigration from nearby independent populations (Bjorkstedt et al. 2005; Spence et al. 2008).

Abundance and Productivity: Data on CC Chinook abundance, both historical and current, are sparse and of varying quality (Bjorkstedt et al. 2005). Estimates of absolute abundance are not available for populations in this ESU (Meyers et al. 1998). In 1965, the California Department of Fish and Game (CDFG)(CDFG 1965) estimated escapement for this ESU at over 76,000. Most were in the Eel River (55,500), with smaller populations in Redwood Creek (5,000), Mad River (5,000), Mattole River (5,000), Russian River (500) and several smaller streams in Humboldt County (Meyers et al. 1998). Currently available data indicate abundance is far lower, suggesting an inability to sustain production adequate to maintain the ESU populations. Recent growth rates are fluctuating (depending upon the year) coastwide in California; for example, in 2007 to 2009, dramatic declines in Chinook salmon returns occurred throughout California. More recently, Chinook salmon counts in the Russian River have continually increased since record lows in 2008. The highest count recorded since monitoring began in 2000 was surpassed in November, 2012 (Sonoma County Water Agency 2012).

Although there are limited population-level estimates of abundance for CC Chinook salmon populations, Table 23 summarizes the information that is available for the major watersheds in the ESU. Based on this limited information, the current average run size for CC Chinook ESU is 7,144 adults. While we currently lack data on naturally-produced juvenile CC Chinook salmon production, it is possible to make rough estimates of juvenile abundance from adult return data. Juvenile CC Chinook salmon population abundance estimates come from escapement data, the

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[9] Population is defined by Bjorkstedt, E., and coauthors. 2005. An analysis of historical population structure for evolutionarily signficant units of Chinook salmon, coho salmon, and steelhead in the north-Central California Coast Recovery Domain, NOAA-TM-NMFS-SWFSC-382. and McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000a. Viable salmonid populations and the recovery of evolutionarily significant units. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center. as, in brief summary, a group of fish of the same species that spawns in a particular locality at a particular season and does not interbreed substantially with fish from any other group. Such fish groups may include more than one stream. These authors use this definition as a starting point from which they define four types of populations (not all of which are mentioned here).

percentage of females in the population, and fecundity. Average fecundity for female CC Chinook is not available. However, Healey and Heard (1984) indicates that average fecundity for Chinook salmon in the nearby Klamath River is 3,634 eggs for female. By applying an average fecundity of 3,634 eggs per female to the estimated 3,572 females returning (half of the average total number of spawners), and applying an estimated survival rate from egg to smolt of 10 percent, the ESU could produce roughly 1,298,065 natural outmigrants annually.

Table 23. Abundance Geometric Means for Adult CC Chinook Salmon Natural-origin Spawners.

Population	Years	Spawners	Expected Number of Outmigrants[ab]
Redwood Creek[c]	2009-2013	1,745	317,067
Mad River[d]	2008 - 2013	76	13,809
Freshwater Creek[e]	2008 - 2013	3	545
Eel River mainstem[f]	2008 - 2013	1,379	250,564
Eel River (Tomki Creek)[d]	2008 - 2013	61	11,084
Eel River (Sproul Creek)[d]	2008 - 2013	187	33,978
Mattole River[g]	2006 - 2009	1,170	212,589
Russian River[h]	2008 - 2013	2,523	458,429
Total		7,144	1,298,065

[a] Expected number of outmigrants=Total spawners*50% proportion of females*3,634 eggs per female*10% survival rate from egg to outmigrant.

[b] Based upon number of natural-origin spawners.

[c] (Metheny and Duffy 2014)

[d] PFMC (2013)

[e] (Ricker et al. 2014)

[f] source: http://www.pottervalleywater.org/van_arsdale_fish_counts.html

[g] (Group 2011)

[h] source: http://www.scwa.ca.gov/chinook/

Limiting Factors: Because of their prized status in the sport and commercial fishing industries, CC Chinook salmon have been the subject of many artificial production efforts, including out-of-basin and out-of-ESU stock transfers (Bjorkstedt et al. 2005). It is, therefore, likely that CC Chinook salmon genetic diversity has been significantly adversely affected despite the relatively wide distribution of populations within the ESU. An apparent loss of the spring-run Chinook life history in the Eel River Basin and elsewhere in the ESU also indicates risks to the diversity of the ESU. Data from the 2009 adult CC Chinook salmon return counts and estimates indicated a further decline in returning adults across the range of CC Chinook salmon on the coast of California (Jeffrey Jahn, NMFS, personal communication, 2010). Ocean conditions are suspected as a primary short term cause because of the wide geographic range of declines (Lindley et al. 2009). However, the number of adult CC Chinook salmon returns in the Russian River Watershed increased substantially in 2010/2011 compared to 2008/09 and 2009/10 returns.[10] Increases in adult Chinook salmon returns during 2010/2011 have been observed in the Central Valley populations as well. These numbers must be taken in context of the overall Chinook salmon abundance in the ESU which has recently been reviewed by Williams et al. (2011), who found no evidence of a substantial change in the status of the CC Chinook ESU since the last status review by Good et al. (2005a). Based on this information, NMFS chose to maintain the threatened listing of CC Chinook salmon (76 FR 50447).

4.2.13.8.4 Upper Columbia River Spring-run Chinook ESU

The Upper Columbia River Spring-run Chinook ESU was listed as endangered on June 28, 2005 (70 FR37160).

Spatial Structure and Diversity: Upper Columbia River (UCR) spring Chinook includes all naturally-spawned populations of Chinook salmon in all river reaches accessible to Chinook salmon in Columbia River tributaries upstream of the Rock Island Dam and downstream of Chief Joseph Dam (excluding the Okanogan River), the Columbia River upstream to Chief Joseph Dam, and progeny of six artificial propagation programs. The IC-TRT identified four independent populations of UCR spring-run Chinook salmon in the upriver tributaries of Wenatchee, Entiat, Methow, and Okanogan (extirpated), but no major groups due to the relatively small geographic area affected (Ford 2011b; NMFS 2011g).

The composite SS/D risks for all three of the extant populations in this MPG are at "high" risk. The spatial processes component of the SS/D risk is "low" for the Wenatchee River and Methow River populations and "moderate" for the Entiat River (loss of production in lower section increases effective distance to other populations). All three of the extant populations in this MPG are at "high" risk for diversity, driven primarily by chronically high proportions of hatchery-origin spawners in natural spawning areas and lack of genetic diversity among the natural-origin spawners (Ford 2011b).

Increases in natural origin abundance relative to the extremely low spawning levels observed in the mid-1990s are encouraging; however, average productivity levels remain extremely low. Overall, the viability of UCR spring Chinook salmon ESU has likely improved somewhat since

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[10] http://www.scwa.ca.gov/chinook/

the last status review, but the ESU is still clearly at "moderate-to-high" risk of extinction (Ford 2011b).

Abundance and Productivity: UCR spring Chinook salmon is not currently meeting the viability criteria (adapted from the IC-TRT) in the Upper Columbia Recovery Plan. A&P remains at "high" risk for each of the three extant populations in this MPG/ESU. The 10-year geometric mean abundance of adult natural origin spawners increased for each population relative to the levels for the 1981 to 2003 series, but the estimates remain below the corresponding IC-TRT thresholds. Estimated productivity (spawner to spawner return rate at low to moderate escapements) was on average lower over the years 1987 to 2009 than for the previous period. The combinations of current abundance and productivity for each population result in a "high" risk rating.

From the year 2006 through 2010, the five-year average return to the ESU—as measured primarily by spawning surveys—was 3,900 (Salmonid Population Summary (SPS) query, April 2014[11]); of these, approximately 65 percent were of hatchery origin. Counts at Rock Island Dam in 2008, 2010, and 2011 showed an average estimated 1,668 natural fish retuning to the ESU which, given a 35 percent natural origin for the overall return, indicated that the total return was on the order of 4,766 fish. These figures demonstrate that there is some degree of variability in the various sources for returning adult numbers. As a result, it is sometimes difficult to take all the various factors into account (survey types, data gaps, various dam counts, hatchery vs. wild components, etc.) and clearly and accurately determine what the returns actually are. Nonetheless, the figures we believe to be the most likely to represent the actual returns come from the U.S. v. Oregon Technical Advisory Committee (TAC) numbers derived from dam counts and complied by the WDFW (WDFW 2013). These numbers are widely used throughout the region for management purposes (particularly in setting harvest quotas), and at this point represent the very best available scientific and technical knowledge to which we have access. The most recent year for which these numbers have been calculated and published is 2014 (via the FCRPS Adaptive Management Implementation Plan). That year, the UCR spring Chinook total return to Rock Island Dam was 3,986 natural adults. The most recent four-year average to that date was 3,170 fish. Given that these fish comprise approximately 35 percent of the total run, it signifies that the total return for 2014 was 11,388 fish and the most recent four year average was 9,057 adults.

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[11] The data contained in the SPS database are primarily summary data, compiled at the population level. The database also includes a limited number of series representing the aggregate returns to groups of populations (e.g., Lower Granite Dam counts) or counts of spawners within a subsection of a population where expansions to the population level were not feasible.

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of ESA-listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. The averages of the five most recent projections for the UCR spring Chinook juvenile outmigration are displayed below in Table 24.

Table 24. Recent five-year average projected outmigrations for UCR Chinook (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	Outmigration
Natural	570,965
Listed Hatchery: Adipose Clipped*	504,620
Listed Hatchery: Intact Adipose*	931,815

***When the above species was listed, NMFS included certain artificially propagated (hatchery-origin) populations in the listing. Some of those listed fish have had their adipose fins clipped at their respective hatcheries and some have not.**

Limiting Factors include (NOAA Fisheries 2011; Upper Columbia Salmon Recovery Board 2007):

- Mainstem Columbia River hydropower–related adverse effects: upstream and downstream fish passage, ecosystem structure and function, flows, and water quality
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Degraded estuarine and nearshore marine habitat
- Hatchery related effects: including past introductions and persistence of non-native (exotic) fish species continues to affect habitat conditions for listed species
- Harvest in Columbia River fisheries

4.2.13.8.5 Upper Willamette River Chinook ESU

The NMFS originally listed Upper Willamette River Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their status on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Upper Willamette River (UWR) Chinook includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River; in the Willamette River and its tributaries above Willamette Falls, Oregon; and progeny of seven artificial propagation programs. All seven historical populations of UWR Chinook salmon identified by the WLC-TRT occur within the Action Area and are contained within a single ecological subregion, the western Cascade Range. The McKenzie River population currently characterized as at a "low" risk of extinction and the Clackamas population has a "moderate" risk (Ford 2011b). Consideration of data collected since the last status review in 2005 has confirmed the high fraction of hatchery origin fish in all of the populations of this species (even the Clackamas and McKenzie rivers have hatchery fractions above WLC-TRT viability thresholds). All of the UWR Chinook salmon populations have "moderate" or "high" risk ratings for diversity. Clackamas River Chinook salmon have a "low" risk rating for spatial structure (Ford 2011b).

Abundance and Productivity: The Clackamas and McKenzie river populations currently have the best risk ratings for A&P, spatial structure, and diversity. Data collected since the Biological Review Team (BRT) status update in 2005 highlighted the substantial risks associated with pre-spawning mortality. Although recovery plans are targeting key limiting factors for future actions, there have been no significant on-the-ground-actions since the last status review to resolve the lack of access to historical habitat above dams nor have there been substantial actions removing hatchery fish from the spawning grounds. Overall, the new information does not indicate a change in the biological risk category since the last status review (Ford 2011b).

Recent data on returning adults are summarized in Table 25 (ODFW 2014a; ODFW and WDFW 2013; ODFW and WDFW 2014b) (ODFW and WDFW 2011; ODFW and WDFW 2012). Abundance of adult UWR spring Chinook has declined since the highs witnessed around the turn of this century. The 5-year average return for UWR spring Chinook salmon is 11,061 naturally produced adults and 38,135 hatchery adults (2010 to 2014). Average escapement for the years 2010 to 2014 was a combined total of 49,196 hatchery- and naturally-produced adult Chinook.

Table 25. Adult UWR spring Chinook escapement to the Clackamas River and Willamette Falls fish ladder (ODFW 2014a; ODFW and WDFW 2013; ODFW and WDFW 2014b) (ODFW and WDFW 2011; ODFW and WDFW 2012).

Year	Total Escapement	Hatchery Escapement	Natural Escapement
2010	78,032	66,543	11,489
2011	51,922	36,506	15,416
2012	43,012	32,334	10,678
2013	35,714	24,332	11,382
2014	37,300	30,959	6,341
Average	49,196	38,135	11,061

The NWFSC publishes juvenile abundance estimates each year in the annual memorandum estimating percentages of ESA-listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time; however the average outmigration for the years 2010 to 2014 is shown in Table 26 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Table 26. Average estimated outmigration for ESA-listed UWR Chinook salmon (2010-2014).

Origin	Outmigration
Natural	1,813,726
Listed hatchery intact adipose	42,420
Listed hatchery adipose clipped	6,006,713

Limiting Factors include (NOAA Fisheries 2011; ODFW and NMFS 2011):

- Significantly reduced access to spawning and rearing habitat because of tributary dams
- Degraded freshwater habitat, especially floodplain connectivity and function, channel structure and complexity, and riparian areas and large wood recruitment as a result of cumulative impacts of agriculture, forestry, and development
- Degraded water quality and altered temperature as a result of both tributary dams and the cumulative impacts of agriculture, forestry, and urban development
- Hatchery-related effects
- Anthropogenic introductions of non-native species and out-of-ESU races of salmon or steelhead have increased predation on, and competition with, native UWR Chinook salmon
- Ocean harvest rates of approximately 30 percent

4.2.13.8.6 Central Valley Spring-run Chinook ESU

On September 16, 1999, the Central Valley Spring-run Chinook salmon ESU was listed as threatened, and it was reaffirmed as threatened on June, 25 2005 (70 FR 37160).

Spatial Structure and Diversity: Historically, the predominant salmon run in the Central Valley was the spring-run Chinook salmon. Extensive construction of dams throughout the Sacramento-San Joaquin basin has reduced the Central Valley spring-run (CVS) Chinook salmon run to only a small portion of its historical distribution. The Central Valley drainage as a whole is estimated to have supported CVS Chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s (CDFG 1998). The ESU has been reduced to only three naturally-spawning populations that are free of hatchery influence from an estimated 17 historic populations.[12] These three populations (spawning in three tributaries to the Sacramento River - Deer, Mill, and Butte creeks), are in close geographic proximity, increasing the ESU's vulnerability to disease or catastrophic events. CVS Chinook salmon from the Feather River Hatchery (FRH) were included in the ESU because they are believed by NMFS to be the only population in the ESU that displays early run timing. This early run timing is considered by NMFS to represent an important

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[12] There has also been a small run in Big Chico Creek in recent years (Good et al. 2005).

evolutionary legacy of the spring-run populations that once spawned above Oroville Dam (70 FR 37160).

Abundance and Productivity: The FRH's goal is to release five million spring-run Chinook salmon per year. Over the past five years, the Feather River hatchery released an average of 2,178,601 juvenile adipose clipped CVS Chinook salmon (Table 27).

Table 27. Average CVS Chinook salmon smolt release 2009-2013 (Regional Mark Processing Center 2014).

Artificial propagation program	Run Timing	Clipped Adipose Fin	Intact Adipose Fin
Feather River Hatchery	Spring	2,178,601	-
Total		2,178,601	

Sacramento River tributary populations in Mill, Deer, and Butte creeks are likely the best trend indicators for the CVS Chinook salmon ESU as a whole because these streams contain the majority of the abundance, and are currently the only independent populations in the ESU. Generally, these streams have shown a positive escapement trend since 1991, displaying broad fluctuations in adult abundance, ranging from 1,013 in 1993 to 23,788 in 1998 (Table 28). Escapement numbers are dominated by Butte Creek returns, which averaged over 7,000 fish from 1995 to 2005 (peaking in 1998 at over 20,000 fish and 2005 at over 10,000 fish), but then declined in years 2006 through 2011 with an average of just over 3,000 (with the exception of 2008 which was almost 15,000 fish). During this same period, adult returns on Mill and Deer creeks have averaged over 2,000 fish total and just over 1,000 fish total, respectively. From 2001 to 2005, the CVS Chinook salmon ESU experienced a trend of increasing abundance in some natural populations, most dramatically in the Butte Creek population (Good et al. 2005a). Although trends were generally positive during this time, annual abundance estimates display a high level of fluctuation, and the overall number of CVS Chinook salmon remained well below estimates of historic abundance.

Table 28. CVS Chinook salmon population estimates from (CDFW 2013) with corresponding cohort replacement rates for years since 1986.

Year	Sacramento River Basin Escapement Run Size[a]	FRFH Population	Tributary Populations	5-Year Moving Average Tributary Population Estimate	Trib CRR[b]	5-Year Moving Average of Trib CRR	5-Year Moving Average of Basin Population Estimate	Basin CRR	5-Year Moving Average of Basin CRR
1986	3,638	1,433	2,205						
1987	1,517	1,213	304						
1988	9,066	6,833	2,233						
1989	7,032	5,078	1,954		0.89			1.93	

Year	Sacramento River Basin Escapement Run Size[a]	FRFH Population	Tributary Populations	5-Year Moving Average Tributary Population Estimate	Trib CRR[b]	5-Year Moving Average of Trib CRR	5-Year Moving Average of Basin Population Estimate	Basin CRR	5-Year Moving Average of Basin CRR
1990	3,485	1,893	1,592	1,658	5.24		4,948	2.30	
1991	5,101	4,303	798	1,376	0.36		5,240	0.56	
1992	2,673	1,497	1,176	1,551	0.60		5,471	0.38	
1993	5,685	4,672	1,013	1,307	0.64	1.54	4,795	1.63	1.36
1994	5,325	3,641	1,684	1,253	2.11	1.79	4,454	1.04	1.18
1995	14,812	5,414	9,398	2,814	7.99	2.34	6,719	5.54	1.83
1996	8,705	6,381	2,324	3,119	2.29	2.73	7,440	1.53	2.03
1997	5,065	3,653	1,412	3,166	0.84	2.77	7,918	0.95	2.14
1998	30,534	6,746	23,788	7,721	2.53	3.15	12,888	2.06	2.23
1999	9,838	3,731	6,107	8,606	2.63	3.26	13,791	1.13	2.24
2000	9,201	3,657	5,544	7,835	3.93	2.44	12,669	1.82	1.50
2001	16,869	4,135	12,734	9,917	0.54	2.09	14,301	0.55	1.30
2002	17,224	4,189	13,035	12,242	2.13	2.35	16,733	1.75	1.46
2003	17,691	8,662	9,029	9,290	1.63	2.17	14,165	1.92	1.43
2004	13,612	4,212	9,400	9,948	0.74	1.79	14,919	0.81	1.37
2005	16,096	1,774	14,322	11,704	1.10	1.23	16,298	0.93	1.19
2006	10,948	2,181	8,767	10,911	0.97	1.31	15,114	0.62	1.21
2007	9,726	2,674	7,052	9,714	0.75	1.04	13,615	0.71	1.00
2008	6,368	1,624	4,744	8,857	0.33	0.78	11,350	0.40	0.69
2009	3,801	989	2,812	7,539	0.32	0.69	9,388	0.35	0.60
2010	3,792	1,661	2,131	5,101	0.30	0.54	6,927	0.39	0.49
2011	4,967	1,969	3,067	3,961	0.65	0.47	5,731	0.78	0.53
2012	18,275	3,738	10,810	4,713	3.84	1.09	7,441	0.79	0.54
2013	38,556	4,294	18,499	7,464	8.68	2.76	13,878	2.00	0.86
Median	10,962	3,734	6,508	6,324	2.08	1.83	10,258	1.00	1.29

[a] NMFS is only including the escapement numbers from the Feather River Fish Hatchery (FRFH) and the Sacramento River tributaries in this table. Sacramento River Basin run size is the sum of the escapement numbers from the FRFH and the tributaries.

[b] Abbreviations: CRR = Cohort Replacement Rate, Trib = tributary

From 2005 through 2011, abundance numbers in most of the tributaries declined. Adult returns from 2006 to 2009, indicate that population abundance for the entire Sacramento River basin is declining from the peaks seen in the five years prior to 2006. Declines in abundance from 2005 to 2011, placed the Mill Creek and Deer Creek populations in the high extirpation risk category due to the rates of decline, and in the case of Deer Creek, also the level of escapement (NMFS 2011f). Butte Creek has sufficient abundance to retain its low extirpation risk classification, but the rate of population decline in years 2006 through 2011 is nearly sufficient to classify it as a high extirpation risk based on this criteria. Nonetheless, the watersheds identified as having the highest likelihood of success for achieving viability/low risk of extirpation include, Butte, Deer and Mill creeks (NMFS 2011f). Some other tributaries to the Sacramento River, such as Clear Creek and Battle Creek have seen population gains in the years from 2001 to 2009, but the overall abundance numbers have remained low. Year 2012 appeared to be a good return year for most of the tributaries with some, such as Battle Creek, having the highest return on record (799). Additionally, 2013 escapement numbers combined for Butte, Mill and Deer creeks increased (over 17,000), which resulted in the second highest number of spring-run Chinook salmon returning to the tributaries since 1998. However, 2014 appears to be lower, just over 5,000 fish, which indicates a highly fluctuating and unstable ESU.

While we currently lack data on naturally-produced juvenile CVS Chinook salmon production, it is possible to make rough estimates of juvenile abundance from adult return data. The CDFG (1998) published estimates in which average fecundity of spring-run Chinook salmon is 4,161 eggs per female. By applying the average fecundity of 4,161 eggs per female to the estimated 3,732 females returning (half of the average total number of spawners), and applying an estimated survival rate from egg to smolt of 10 percent, the ESU could produce roughly 1,552,885 natural outmigrants annually. In addition, hatchery managers could produce approximately 2,000,000 listed hatchery juvenile CVS Chinook salmon each year (Table 9 above).

Limiting Factors: Several actions have been taken to improve habitat conditions for CVS Chinook salmon, including: habitat restoration efforts in the Central Valley; and changes in freshwater harvest management measures. Although protective measures likely have contributed to recent increases in CVS Chinook salmon abundance, the ESU is still well below levels observed from the 1960s. Threats from climatic variation, high temperatures, predation, and water diversions still persist. Hatchery production can also pose a threat to salmonids. Potential adverse effects from hatchery production include competition for food between naturally-spawned and hatchery fish, run hybridization and genomic homogenization. Despite these potential impacts from hatchery production, NMFS ultimately concluded the FRH stock should be included in the CVS Chinook ESU because it still exhibited a spring-run migration timing and was the best opportunity for restoring a more natural spring-run population in the Feather River. In the most recent status review of this ESU, NMFS concluded that the FRH stock should be considered part of the CVS Chinook ESU (Williams et al. 2011). Because wild CVS Chinook salmon ESU populations are confined to relatively few remaining watersheds and continue to display broad fluctuations in abundance, the BRT concluded that the ESU is likely to become endangered within the foreseeable future. The most recent status review concludes the status of CVS Chinook salmon ESU has probably deteriorated since the 2005 status review (Williams et al. 2011).

4.2.13.8.7 Snake River Spring/Summer Run Chinook ESU

The NMFS originally listed Snake River spring/summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: This species includes all naturally-spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins; and progeny of fifteen artificial propagation programs. The IC-TRT currently believes there are 27 extant and 4 extirpated populations of Snake River (SR) spring/summer-run Chinook salmon, and aggregated these into major population groups (Ford 2011b; IC-TRT 2007). Each of these populations faces a "high" risk of extinction (Ford 2011b).

Abundance and Productivity: Population level status ratings remain at "high" risk across all MPGs within the ESU, although recent natural spawning abundance estimates have increased, all populations remain below minimum natural origin abundance thresholds. Spawning escapements in the most recent years in each series are generally well below the peak returns but above the extreme low levels in the mid-1990s. Relatively low natural production rates and spawning levels below minimum abundance thresholds remain a major concern across the ESU.

The ability of SR spring/summer-run Chinook salmon populations to be self-sustaining through normal periods of relatively low ocean survival remains uncertain. Factors cited by Good et al. (2005d) remain as concerns or key uncertainties for several populations. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

The 1997 to 2001 geometric mean total return for spring/summer Chinook was slightly more than 6,000 fish. This was a marked improvement over the previous ten years when the geometric mean return was 3,076. That increase continued relatively steadily through 2004, when 97,946 adults returned (including jacks), but dropped off precipitously in 2005 when only 39,126 fish (including jacks) returned above Ice Harbor Dam (Fish Passage Center 2005). The increases from 2001 through 2004 are generally thought to have been a result of good ocean conditions for rearing and good Columbia River flows for outmigration. But even with generally better trends in recent years, no population of SR spring/summer Chinook is known to be meeting its interim recovery goals (Ford 2011b). In fact, the most recent return numbers to individual populations show most of the runs to be at less than half the desired levels (Ford 2011b). Overall, from the year 2009 through the year 2013, the five-year average escapement of naturally produced SR spring/summer Chinook was 20,422 naturally produced and 60,058 hatchery propagated fish (Table 29).

Table 29. Estimated numbers of adult SR spring/summer Chinook salmon (ODFW and WDFW 2014a).

Year	Return to Columbia River		Hatchery Escapement[a]	Natural Escapement[a]
	Hatchery	Natural		
2009	68,937	20,240	48,750	14,313
2010	130,976	34,764	94,984	25,211
2011	92,639	30,567	72,264	23,844
2012	75,656	33,856	55,482	24,828
2013	45,400	21,929	28,811	13,916
Average	82,722	28,271	60,058	20,422

[a] Lower Granite Dam passage plus Tucannon River escapement.

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. The averages of the five most recent projections for the SR spring/summer Chinook salmon juvenile outmigration are displayed below.

Table 30. Recent five-year average projected outmigrations for SR spring/summer Chinook salmon (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	**Outmigration**
Natural	1,454,727
Listed Hatchery: Adipose Clipped*	4,381,302
Listed Hatchery: Intact Adipose*	1,158,078

***When the above species was listed, NMFS included certain artificially propagated (hatchery-origin) populations in the listing. Some of those listed fish have had their adipose fins clipped at their respective hatcheries and some have not.**

Limiting Factors include (NOAA Fisheries 2011):

- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, elevated water temperature, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Mainstem Columbia River and Snake River hydropower impacts
- Harvest-related effects
- Predation

4.2.13.8.8 Snake River Fall Run Chinook ESU

The NMFS originally listed Snake River fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Snake River (SR) fall Chinook includes all naturally-spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River, and progeny of four artificial propagation programs. The Interior Columbia Technical Recovery Team (IC-TRT) identified three populations of this species, although only the lower mainstem population exists at present, and it spawns in the lower main stem of the Clearwater, Imnaha, Grande Ronde, Salmon and Tucannon rivers. The extant population of SR fall Chinook salmon is the only remaining population from an historical ESU that also included large mainstem populations upstream of the current location of the Hells Canyon Dam complex (Ford 2011b; NMFS 2011g). The population is at moderate risk for diversity and spatial structure. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

Abundance and Productivity: The recent increases in natural origin abundance are encouraging. However, hatchery origin spawner proportions have increased dramatically in recent years – on average, 78 percent of the estimated adult spawners have been hatchery origin over the most recent brood cycle. The apparent leveling off of natural returns in spite of the increases in total brood year spawners may indicate that density dependent habitat effects are influencing production or that high hatchery proportions may be influencing natural production rates. The A&P risk rating for the population is "moderate." Given the combination of current A&P and SS/D ratings summarized above, the overall viability rating for Lower SR fall Chinook salmon would be rated as "maintained."[13]

The 1999 NMFS Status Review Update noted increases in the Lower Granite Dam counts in the mid-1990s, and the upward trend in returns—the 2001 count over Lower Granite Dam exceeded 8,700 adult fall Chinook—has largely continued. The largest increase in fall Chinook returns to the Snake River spawning area was from the Lyons Ferry Snake River stock component. Returns there increased from under 200 per year before to 1998 to over 1,200 and 5,300 adults in 2000 and 2001, respectively. The increase includes returns from the on-station release program as well as returns from large supplementation releases above Lower Granite Dam. Moreover, from the year 2003 through the year 2008, the five-year average return to the ESU was 11,321 adult fish (Ford 2011b); of these, approximately 78 percent were of hatchery origin. Overall, from the year 2010 through the year 2014, the five-year average escapement of SR fall Chinook was 14,438 naturally produced adult fish and 30,475 hatchery propagated adult fish Table 31.

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[13] "Maintained" population status is for populations that do not meet the criteria for a viable population but do support ecological functions and preserve options for ESU/DPS recovery.

Table 31. Estimated numbers of adult SR fall Chinook salmon (NWFSC 2015).

Year	Hatchery Escapement	Natural Escapement
2010	32,408	7,356
2011	15,516	8,064
2012	19,038	11,325
2013	30,794	20,444
2014	54,621	25,001
Average	30,475	14,438

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time; however the average outmigration for the years 2010 to 2014 is shown in Table 32.

Table 32. Recent five-year average projected outmigrations for SR fall Chinook salmon (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	**Outmigration**
Natural	570,821
Listed Hatchery: Adipose Clipped	3,076,642
Listed Hatchery: Intact Adipose	3,915,529

The number of natural fish should be viewed with caution, however, as it only addresses one of several juvenile life stages. Moreover, deriving any juvenile abundance estimate is complicated by a host of variables, including the facts that: (1) spawner counts and associated sex ratios and fecundity estimates can vary considerably between years; (2) multiple juvenile age classes (fry, parr, smolt) are present yet comparable data sets may not exist for all of them; and (3) survival rates between life stages are poorly understood and subject to a multitude of natural and human-induced variables (e.g., predation, floods, fishing, etc.). Listed hatchery fish outmigration numbers are also affected by some of these factors, however releases from hatcheries are generally easier to quantify than is natural production. These caveats are generally applicable to all salmonid ESUs.

<u>Limiting Factors</u> include (NOAA Fisheries 2011):

- Degraded freshwater habitat: Floodplain connectivity and function, and channel structure and complexity have been degraded as a result of cumulative impacts of agriculture, forestry, and development.
- Harvest-related effects

- Loss of access to historic habitat above Hells Canyon and other Snake River dams
- Mainstem Columbia River and Snake River hydropower impacts
- Hatchery-related effects
- Degraded estuarine and nearshore habitat

4.2.13.8.9 Sacramento River Winter-run Chinook ESU

NMFS listed Sacramento River winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160).

<u>Spatial Structure and Diversity</u>: The Sacramento River winter-run (SWR) Chinook salmon ESU has been completely displaced from its historical spawning habitat by the construction of Shasta and Keswick dams. Approximately, 300 miles of tributary spawning habitat in the upper Sacramento River is now inaccessible to the ESU. Most components of the SWR Chinook salmon life history (e.g., spawning, incubation, freshwater rearing) have been compromised by the habitat blockage in the upper Sacramento River. The only remaining spawning habitat in the upper Sacramento River is between Keswick Dam and Red Bluff Diversion Dam (RBDD). This habitat is artificially maintained by cool water releases from Shasta and Keswick Dams, and the spatial distribution of spawners in the upper Sacramento River is largely governed by the water year type and the ability of the Central Valley Project to manage water temperatures in this area.

<u>Abundance and Productivity</u>: A captive broodstock artificial propagation program for SWR Chinook salmon has operated since the early 1990s as part of recovery actions for this ESU. As many as 150,000 juvenile salmon have been released in a single year by this program, but in most cases the number of fish released was in the tens of thousands (Good et al. 2005a). NMFS reviewed this hatchery program in 2004 and concluded that as much as 10 percent of the natural spawners may be attributable to the program's support of the population (69 FR 33102). However, the naturally spawning component of this ESU has exhibited marked improvements in abundance and productivity in the 2000s (CDFG 2008). These increases in abundance are encouraging, relative to the years of critically low abundance of the 1980s and early 1990s; however, returns of several West Coast Chinook salmon and coho salmon stocks were lower in the 2000's, and SWR Chinook returns dropped below 1,000 in 2011. Returns have been increasing since that time and numbered approximately 6,000 in 2014 (PFMC 2014). This population remains below established recovery goals and the naturally-spawned component of the ESU is dependent on one extant population in the Sacramento River. There is particular concern about risks to the ESU's genetic diversity (genetic diversity is probably limited because there is only one remaining population) life-history variability, local adaptation, and spatial structure (Good et al. (2005a); 70 FR 37160).

Since 2000, the proportion of hatchery-origin, SRW Chinook spawning in the river has ranged up to 10 percent (Table 8), which is below the low-risk threshold for hatchery influence

(Williams et al. 2011). The current average run size for the SRW Chinook salmon ESU is 2,106 fish (2,023 natural-origin, 83 hatchery produced) (Table 33).

Table 33. Average abundance estimates for SRW Chinook salmon natural- and hatchery-origin spawners 2001-2011 (Killam 2012; O'Farrell et al. 2012).

Year	Natural-origin Spawners[a]	Hatchery-origin Spawners[b]	% Hatchery Origin	Expected Number of Outmigrants[c]
2001	8,120	104	1.3%	649,600
2002	7,360	104	1.4%	588,800
2003	8,133	85	1.0%	650,640
2004	7,784	85	1.1%	622,720
2005	15,730	109	0.7%	1,258,400
2006	17,197	99	0.6%	1,375,760
2007	2,487	55	2.2%	198,960
2008	2,725	105	3.7%	218,000
2009	4,416	121	2.7%	353,280
2010	1,533	63	3.9%	122,640
2011	738	89	10.8%	59,040
ESU Average[d]	2,023	83	3.9%	161,840

[a] Five-year geometric mean of post fishery natural-origin spawners (2007-2011).

[b] Five-year geometric mean of post fishery hatchery-origin spawners (2007-2011). Data from http://www.calfish.org/LinkClick.aspx?fileticket=Kttf%2boZ2ras%3d&tabid=104&mid=524.

[c] Expected number of outmigrants=Total spawners*40% proportion of females*2,000 eggs per female*10% survival rate from egg to outmigrant

[d] Averages are calculated as the geometric mean of the annual totals (2007-2011).

Juvenile SRW Chinook abundance estimates come from escapement data, the percentage of females in the population, and fecundity. Fecundity estimates for the ESU range from 2,000 to 5,500 eggs per female, and the proportion of female spawners in most populations is approximately 40 percent of escapement. By applying a conservative fecundity estimate (2,000 eggs/female) to the expected female escapement (both natural-origin and hatchery-origin spawners – 809 females), the ESU is estimated to produce approximately 1.6 million eggs annually. The average survival rate in these studies was 10 percent, which corresponds with those reported by Healey (1991). With an estimated survival rate of 10 percent, the ESU should produce roughly 161,840 natural outmigrants annually.

Limiting Factors: SRW Chinook salmon were first listed as threatened in 1989 under an emergency rule. In 1994, NMFS reclassified the ESU as an endangered species due to several factors, including: (1) the continued decline and increased variability of run sizes since its listing as a threatened species in 1989; (2) the expectation of weak returns in coming years as the result of two small year classes (1991 and 1993); and (3) continuing threats to the species. NMFS issued a final listing determination on June 28, 2005. Between the time Shasta Dam was built and the SRW Chinook salmon were listed in 1989, major impacts to the population occurred from warm water releases from Shasta Dam, juvenile and adult passage constraints at the RBDD, water exports in the southern Delta, and entrainment at a large number of unscreened or poorly-screened water diversions.

Designated critical habitat for SRW Chinook salmon has been degraded from conditions known to support viable salmonid populations. It does not provide the full extent of conservation values necessary for the recovery of the species. In particular, adequate river flows and water temperatures have been impacted by human actions, substantially altering the historical river characteristics in which the SRW Chinook salmon evolved. Depletion and storage of stream flows behind large dams on the Sacramento River and other tributary streams have drastically altered the natural hydrologic cycles of the Sacramento River and Delta. Alteration of flows results in migration delays, loss of suitable habitat due to dewatering and blockage; stranding of fish from rapid flow fluctuations; entrainment of juveniles into poorly screened or unscreened diversions, and increased water temperatures harmful to salmonids. Other impacts of concern include alteration of stream bank and channel morphology, loss of riparian vegetation, loss of spawning and rearing habitat, fragmentation of habitat, loss of downstream recruitment of spawning gravels, degradation of water quality, and loss of nutrient input.

4.2.14 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.

Unlike most species that rear extensively in fresh water, chum salmon form schools, presumably to reduce predation.

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.

4.2.14.1 *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.

4.2.14.1.1 Occurrence in the NWTT Action Area

Chum salmon may occur in the Offshore Area portion of the NWTT Action Area, while the Hood Canal summer-run ESU will occur in both Offshore Area and Inland Waters portions of the Action Area. Additionally, nearshore critical habitat is located in the Inland Waters of the Action Area. The Columbia River ESU will occur in the Offshore Area.

4.2.14.2 *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).

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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).

4.2.14.3 *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. 1997). 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. 1997).

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).

Chum salmon are found in freshwater to euryhaline water at depths ranging from the surface to 250 m, although juveniles are primarily epipelagic and are found from the surface down to 95 m and within 36 km of shore (Emmett et al. 1991b; Salo 1991b). The species is found at a wide range of temperatures from 3° to 22° C but prefer temperatures from 8.3° to 15.6° C (Pauley et al. 1988).

4.2.14.4 *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.

4.2.14.5 *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. 2007). 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).

4.2.14.6 *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 2011e).

4.2.14.7 *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 2011e).

4.2.14.8 ***Chum Salmon ESUs***

Each Chum salmon ESU is treated as a separate species under the ESA (NMFS 2005c). 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 2005c). The distribution, as well as the status and trends, of the Chum salmon ESUs considered in this Opinion are discussed below.

4.2.14.8.1 Columbia River Chum Salmon

The NMFS listed Columbia River chum salmon as threatened on March 25, 1999, and reaffirmed their status on June 28, 2005 (71 FR 37160).

Spatial Structure and Diversity: Columbia River (CR) chum includes all naturally-spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon, and progeny of three artificial propagation programs. The WLC-TRT identified 17 historical populations of CR chum salmon and aggregated these into four strata (Myers et al. 2006). CR chum salmon spawning aggregations identified in the mainstem Columbia River were included in the population associated with the nearest river basin.

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 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 the low abundance in the remaining populations (fewer than 100 spawners per year for most populations) (Lower Columbia Fish Recovery Board 2010a; NMFS 2013). The Lower Gorge population meets abundance and productivity criteria for very high levels of viability, but the distribution of spawning habitat (i.e., spatial structure) for the population has been significantly reduced (Lower Columbia Fish Recovery Board 2010a); spatial structure may need to be improved, at least in part, through better performance from the Oregon portion of the population (NMFS 2013).

Abundance and Productivity: Of the 17 populations that historically made up this ESU, 15 of them (six in Oregon and nine in Washington) are so depleted that either their baseline probability of persistence is very low or they are extirpated or nearly so (Ford 2011b; Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a). All three strata in the ESU fall significantly short of the WLC-TRT criteria for viability. Currently almost all natural production occurs in just two populations: the Grays/Chinook and the Lower Gorge. The Grays/Chinook population has a moderate persistence probability, and the Lower Gorge population has a high probability of persistence (Lower Columbia Fish Recovery Board 2010a; NMFS 2013).

WDFW regularly monitors several natural "index" populations in the basin, in Grays River, two in small streams near Bonneville Dam, and the mainstem area next to those two streams. Average annual natural escapement to the index spawning areas was approximately 1,300 fish from 1990 through 1998. The WDFW surveyed other (nonindex) areas in 1998 and found only small numbers of chum salmon (typically less than 10 fish per stream) in Elochoman, Abernathy, Germany, St. Cloud, and Tanner Creeks and in the North Fork Lewis and the Washougal Rivers. Consistent with the BRT status review (Ford 2011a), the ODFW recovery plan concluded that chum are extirpated or nearly so in all Oregon Columbia River populations (ODFW 2010a). A few chum are occasionally encountered during surveys or return to hatchery collection facilities, but these are likely either strays from one of the Washington populations or part of a few extremely small and erratic remnant populations. Recent estimates for the lower Columbia Gorge and Grays River chum salmon populations range from 10,000 to 20,000 adults. WDFW spawning surveys in the Grays/Chinook, Washougal, Lower Gorge, and Upper Gorge populations estimated an average of 8,508 adult chum for the years 2007 to 2011 (WDFW 2014). We do not have recent adult abundance data for any of the other populations.

The NWFSC publishes juvenile abundance estimates each year in the annual memorandum estimating percentages of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time; however the average outmigration for the years 2010 to 2014 is shown in Table 34 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Table 34. Average estimated outmigration for ESA-listed CR chum salmon (2010-2014).

Origin	Outmigration
Natural	2,978,550
Listed hatchery intact adipose	391,973
Listed hatchery adipose clipped	0

Limiting Factors include (NMFS 2013; NOAA Fisheries 2011):

- Degraded estuarine and nearshore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system
- Degraded freshwater habitat, in particular of floodplain connectivity and function, channel structure and complexity, stream substrate, and riparian areas and large wood recruitment as a result of cumulative impacts of agriculture, forestry, and development
- Degraded stream flow as a result of hydropower and water supply operations
- Loss of access and loss of some habitat types as a result of passage barriers such as roads and railroads
- Reduced water quality
- Current or potential predation from hatchery-origin salmonids, including coho salmon
- An altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity
- Reduced access to off-channel rearing habitat in the lower Columbia River
- Reduced productivity resulting from sediment and nutrient-related changes in the estuary
- Juvenile fish strandings that result from ship wakes
- Contaminants affecting fish health and reproduction

4.2.14.8.2 Hood Canal Summer-run Chum Salmon

The NMFS listed Hood Canal summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed this status on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Hood Canal summer (HCS) chum includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries; populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington; and progeny of eight artificial propagation programs. The Strait of Juan de Fuca population spawns in rivers and streams entering the eastern Strait and Admiralty Inlet. The HCS chum population includes all spawning aggregations within the Hood Canal area (Hood Canal Coordinating Council 2005; NMFS 2007b). The Puget Sound Technical Recovery Team (PS-TRT) identified two independent populations of HCS chum salmon (NMFS 2007a), which include 16 historical stocks or spawning aggregations (including eight that are extant), based on consideration of historical distribution, geographic isolation, dispersal rates, genetic data, life history information, population dynamics, and environmental and ecological diversity. The historical populations included at least those 16 spawning aggregation units and likely some additional undocumented and less-persistent aggregations (NMFS 2007a). Programs are underway to reintroduce summer-run chum salmon to several of the watersheds where stocks were lost.

Diversity is increasing from the low values seen in the 1990s, due both to the reintroduction of spawning aggregates and the more uniform relative abundance between populations; this is a good sign for viability in terms of spatial structure and diversity. Spawning survey data shows that the spawning distribution within most streams has been extended farther upstream as abundance has increased (WDFW and Point No Point Treaty Tribes 2007). Estimates of population viability from three time periods (brood years 1971 to 2006, 1985 to 2006, and 1990 to 2006) all indicate that Hood Canal and Strait of Juan de Fuca populations of summer-run chum salmon are not currently viable (Ford 2011b).

Abundance and Productivity: Overall, the new information considered does not indicate a change in the biological risk category since the last status review in 2005 (Ford 2011b). The spawning abundance of this species has clearly increased since the time of listing, although the recent abundance is down from the previous 5 years. However, productivity in the last 5-year period (2002 through 2006) has been very low, especially compared to the relatively high productivity in the 5 to 10 previous years (WDFW and Point No Point Treaty Tribes 2007). This is a concern for viability. Since abundance is increasing and productivity is decreasing, improvements in habitat and ecosystem function likely are needed.

The current average run size of 21,008 HCS chum adult spawners (17,556 natural-origin and 3,452 hatchery origin spawners; Table 23) is largely the result of aggressive reintroduction and supplementation programs throughout the ESU. In the Strait of Juan de Fuca population, the annual natural-origin spawners returns for Jimmycomelately Creek dipped to a single fish in 1999 and again in 2002 (unpublished data, Mindy Rowse, NWFSC, Nov. 25, 2014; as cited in NMFS 2015). From 2009 to 2013, Jimmycomelately Creek averaged 1,058 natural-origin spawners. Salmon and Snow Creeks have improved substantially. Natural-origin spawner abundance was 130 fish in 1999, whereas the average for Salmon and Snow creeks were 2,171 and 405, respectively, for the 2009 to 2013 period.

Table 35. Abundance of natural-origin and hatchery-origin HCS chum salmon spawners in escapements 2009-2013 (unpublished data, Mindy Rowse, NWFSC, Nov. 25, 2014; as cited in NMFS 2015).

Population	Spawning Aggregation	Natural-origin Spawners[a]	Hatchery-origin Spawners[b]	Expected Number of Outmigrants[c]
Strait of Juan de Fuca	Jimmycomelately Creek	1,058	1,867	427,781
	Salmon Creek	2,171	3	317,948
	Snow Creek	405	3	59,670
	Chimacum Creek	1,286	0	188,078
	Population Average[d]	5,219	1,879	1,038,083
Hood Canal	Big Quilcene River	3,064	0	448,110
	Little Quilcene River	623	0	91,114
	Big Beef Creek	120	0	17,550
	Dosewallips River	1,734	8	254,768
	Duckabush River	3,183	29	469,755
	Hamma Hamma River	1,220	39	184,129
	Anderson Creek	1	0	146
	Dewatto River	52	4	8,190
	Lilliwaup Creek	186	160	50,603
	Tahuya River	69	577	94,478
	Union River	883	19	131,918
	Population Average[d]	11,946	1,189	1,920,994
ESU Average[d]		17,556	3,452	3,072,420

[a] Five-year geometric mean of post fishery natural-origin spawners (2008-2012).

[b] Five-year geometric mean of post fishery hatchery-origin spawners (2008-2012).

[c] Expected number of outmigrants=Total spawners*45% proportion of females*2,500 eggs per female*13% survival rate from egg to outmigrant

[d] Averages are calculated as the geometric mean of the annual totals (2008-2012)

<u>Limiting factors</u> include (Hood Canal Coordinating Council 2005; NMFS 2007b; NOAA Fisheries 2011):

- Nearshore and estuarine habitat throughout the range of the species has been altered by human activities. Nutrient loading has lowered dissolved oxygen concentrations, which can kill or stress marine organisms, including salmon. Residential and commercial development has reduced the amount of functioning habitat available for salmon rearing and migration. The loss of mudflats, eelgrass meadows, and macroalgae further limits salmon foraging and rearing opportunities in nearshore and estuarine areas.

- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, and stream flow have been degraded as a result of cumulative impacts of agriculture, forestry, and development.

4.2.15 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).

4.2.15.1 *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).

4.2.15.1.1 Occurrence in the NWTT Action Area

All Coho salmon ESUs may occur in Offshore waters of the NWTT Action Area. Juvenile coho salmon are found in high concentrations within 60 km of the Washington, Oregon, and California coast, with a majority of those fish found within 37 km of the coast (Pearcy 1992; Pearcy and Fisher 1990). Adults of the Lower Columbia, Oregon Coast, and Southern Oregon and Northern California Coasts ESUs will occur in the Offshore Area.

4.2.15.2 *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).

4.2.15.3 *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 2000b). 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 2000b). 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 2000b). 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 2000b). Juveniles are also found closer to shore; generally within 74 km (NMFS-NWR 2005; PFMC 2000b). However, adults have been tracked well beyond the EEZ off Oregon (PFMC 2000b). 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).

4.2.15.4 ***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).

4.2.15.5 ***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. 2007). 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).

4.2.15.6 ***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 2011e).

4.2.15.7 *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 2011e).

4.2.15.8 *Coho Salmon ESUs*

Each Coho salmon ESU is treated as a separate species under the ESA (NMFS 2005c). There are currently seven ESUs of coho salmon in Washington, Oregon, and California (NMFS 2005c). 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 2005c) (70 FR 37160).

4.2.15.8.1 Lower Columbia River Coho Salmon

The NMFS listed Lower Columbia River Coho salmon as endangered on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Lower Columbia River (LCR) coho includes all naturally-spawned populations of coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia up to and including the Big White Salmon and Hood rivers; in the Willamette River to Willamette Falls, Oregon; and progeny of 25 artificial propagation programs.[14] Spatial diversity is rated "moderate" to "very high" for all the populations, except the North Fork Lewis River, which has a "low" rating for spatial structure.

Three status evaluations of LCR coho salmon status, all based on WLC-TRT criteria, have been conducted since the last NMFS status review in 2005 (McElhany et al. 2007; NMFS 2013). Out of the 24 populations that make up this ESU, 21 are considered to have a very low probability of persisting for the next 100 years, and none is considered viable (Ford 2011b; Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a).

Abundance and Productivity: In Oregon, the Clatskanie Creek and Clackamas River populations have "low" and "moderate" persistence probability ratings for A&P, while the rest are rated "very low." All of the Washington populations have "very low" A&P ratings. The persistence probability for diversity is "high" in the Clackamas population, "moderate" in the Clatskanie, Scappoose, Lower Cowlitz, South Fork Toutle, Coweeman, East Fork Lewis, and Sandy populations, and "low" to "very low" in the rest (NMFS 2013). Uncertainty is high because of a lack of adult spawner surveys. Smolt traps indicate some natural production in Washington populations, though given the high fraction of hatchery origin spawners suspected to occur in these populations it is not clear that any are self-sustaining. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b; NMFS 2011a; NMFS 2013).

Table 36 displays the most recent returns of naturally produced and hatchery LCR coho salmon. Based on the best available data and using a three year geometric mean, the estimated run size of LCR coho for 2015 is 20,765 naturally produced fish and 394,540 hatchery fish.

Table 36. Estimated abundance of adult LCR coho (ODFW and WDFW 2010; Yakima/Klickitat Fisheries Project 2014)).

Year	Total(1)	Natural(2)	Hatchery(2)
2011	275,989	13,799	262,190
2012	97,576	4,879	92,697
2013	390,828	19,541	371,287
Average(3)	219,149	10,957	208,192

(1) Estimated abundance is calculated by subtracting the number of fish that passed Willamette Falls, Lyle Falls on the Klickitat River, and The Dalles Dam from the total return for the Columbia River. Coho salmon that pass these features are not considered to be part of the LCR coho ESU.

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[14] The Elochoman Hatchery Type-S and Type-N coho salmon programs were eliminated in 2008. The last adults from these two programs returned to the Elochoman in 2010. NMFS has recommended that these two programs be removed from the ESU NMFS. 2011a. 5-year review: summary and evaluation of Lower Columbia River Chinook, Columbia River chum, Lower Columbia River coho, and Lower Columbia River steelhead. National Marine Fisheries Service, editor, Portland, Oregon.

(2) For LCR coho, the approximate percentages of origin are: 5% natural, 95% artificially propagated.
(3) Average is the geometric mean of the last three years of record.

NWFSC publishes juvenile abundance estimates each year in the annual memorandum estimating percentages of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time, however the average outmigration for the years 2010 through 2014 is shown in Table 37 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Table 37. Average estimated outmigration for ESA-listed LCR coho salmon (2010-2014).

Origin	Outmigration
Natural	839,118
Listed hatchery intact adipose	299,928
Listed hatchery adipose clipped	8,637,196

<u>Limiting Factors</u> include (NMFS 2013; NOAA Fisheries 2011):

- Degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system
- Fish passage barriers that limit access to spawning and rearing habitats
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Hatchery-related effects
- Harvest-related effects
- An altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity
- Reduced access to off-channel rearing habitat in the lower Columbia River
- Reduced productivity resulting from sediment and nutrient-related changes in the estuary
- Juvenile fish strandings that result from ship wakes
- Contaminants affecting fish health and reproduction

4.2.15.8.2 Oregon Coast Coho Salmon

The Oregon coast Coho salmon ESU was listed as a threatened species under the ESA on February 11, 2008 (73 FR 7816).

Spatial Structure and Diversity: Oregon coast (OC) coho includes populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco. The Cow Creek stock (South Umpqua population) is included as part of the ESU because the original brood stock was founded from the local, natural origin population and natural origin coho salmon have been incorporated into the brood stock on a regular basis. The OC-TRT identified 56 populations; 21 independent and 35 dependent. The dependent populations were dependent on strays from other populations to maintain them over long time periods. The TRT also identified 5 biogeographic strata (Lawson et al. 2007).

A 2010 BRT noted significant improvements in hatchery and harvest practices have been made (Stout et al. 2012). However, harvest and hatchery reductions have changed the population dynamics of the ESU. Current concerns for spatial structure focus on the Umpqua River. Of the four populations in the Umpqua stratum, the North Umpqua and South Umpqua were of particular concern. The North Umpqua is controlled by Winchester Dam and has historically been dominated by hatchery fish. Hatchery influence has recently been reduced, but the natural productivity of this population remains to be demonstrated. The South Umpqua is a large, warm system with degraded habitat. Spawner distribution appears to be seriously restricted in this population, and it is probably the most vulnerable of any population in this ESU to increased temperatures.

Current status of diversity shows improvement through the waning effects of hatchery fish on populations of OC coho salmon. In addition, recent efforts in several coastal estuaries to restore lost wetlands should be beneficial. However, diversity is lower than it was historically because of the loss of both freshwater and tidal habitat loss coupled with the restriction of diversity from very low returns over the past 20 years.

Abundance and Productivity: It has not been demonstrated that productivity during periods of poor marine survival is now adequate to sustain the ESU. Recent increases in adult escapement do not provide strong evidence that the century-long downward trend has changed. The ability of the OC coho salmon ESU to survive another prolonged period of poor marine survival remains in question. Wainwright (2008) determined that the weakest strata of OC coho salmon were in the North Coast and Mid-Coast of Oregon, which had only "low" certainty of being persistent. The strongest strata were the Lakes and Mid-South Coast, which had "high" certainty of being persistent. To increase certainty that the ESU as a whole is persistent, they recommended that restoration work should focus on those populations with low persistence, particularly those in the North Coast, Mid-Coast, and Umpqua strata.

While we currently lack data on how many natural juvenile coho salmon this ESU produces, it is possible to make rough estimates of juvenile abundance from adult return data. The three-year average of natural origin spawners for the years 2010 to 2012 is estimated at 229,872 total spawners (ODFW 2014b). Sandercock (1991) published fecundity estimates for several coho salmon stocks; average fecundity ranged from 1,983 to 5,000 eggs per female. By applying a very conservative value of 2,000 eggs per female to an estimated 115,000 females returning (roughly half of 229,872) to this ESU, one may expect approximately 230 million eggs to be produced annually. Nickelson (1998) found survival of coho from egg to parr in Oregon coastal streams to be around 7 percent. Thus, we can estimate that roughly 16 million juvenile coho salmon are produced annually by the OC coho ESU.

Limiting Factors include (NOAA Fisheries 2011; Stout et al. 2012):

- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, instream mining, dams, road crossings, dikes, levees, etc.
- Fish passage barriers that limit access to spawning and rearing habitats
- Adverse climate, altered past ocean/marine productivity, and current ocean ecosystem conditions have favored competitors and predators and reduced salmon survival rates in freshwater rivers and lakes, estuaries, and marine environments

4.2.15.8.3 Southern Oregon-Northern California Coast Coho Salmon

Southern Oregon/Northern California coast coho salmon were listed as threatened on May 7, 1997 (62 FR 24588); they retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Southern Oregon/Northern California coastal (SONCC) coho This species includes all naturally spawned populations of coho salmon in coastal streams from the Elk River near Cape Blanco, Oregon, through and including the Mattole River near Punta Gorda, California, and progeny of three artificial propagation programs (NMFS 2012c).

Table 38. SONCC coho salmon ESA-listed hatchery stock annual juvenile production goals (California Hatchery Scientific Review Group 2012; ODFW 2011)).

Artificial propagation program	Location (State)	Listed Hatchery Intact Adipose	Listed Hatchery Adipose Clipped
Cole Rivers Hatchery (ODFW stock #52)	Rogue River (Oregon)	0	200,000
Trinity River Hatchery	Trinity River (California)	500,000	N/A
Iron Gate Hatchery	Klamath River (California)	75,000	N/A

Williams et al. (2006) designated 45 populations of coho salmon in the SONCC coho salmon ESU. These populations were further grouped into seven diversity strata based on the geographical arrangement of the populations and basin-scale genetic, environmental, and ecological characteristics. Across the coastal basins of the SONCC Coho Salmon ESU, there existed sufficient geographical and environmental variability resulting in the TRT dividing the coastal basins into three sub-strata. The northern sub-stratum includes basins from the Elk River to the Winchuck River, including the lower portion of the Rogue River. The central substratum includes coastal basins from the Smith River to the Mad River, including the lower portion of the Klamath River. The southern stratum includes the Humboldt Bay tributaries south to the Mattole River, including the lower Eel River and Van Duzen River.

<u>Abundance and Productivity:</u> Although long-term data on abundance of SONCC coho salmon are scarce, available evidence from shorter-term research and monitoring efforts indicate that conditions have worsened for populations since the last formal status review was published (Williams et al. 2011). Because the extinction risk of an ESU depends upon the extinction risk of its constituent independent populations and the population abundance of most independent populations are below their depensation threshold, the SONCC coho salmon ESU is at high risk of extinction and is not viable (Williams et al. 2011).

Recent returns of naturally-produced adults to the Rogue, Trinity, Shasta, and Scott rivers have been highly variable. Wild coho salmon estimates derived from the beach seine surveys at Huntley Park on the Rogue River ranged from 414 to 24,481 naturally produced adults between 2003 and 2012 (Table 39). Similar fluctuation are noted in the Trinity, Shasta, and Scott river populations. Overall, the average annual abundance, for populations where we have abundance data, of naturally produced fish is only 5,586. However, abundance data is lacking for the Eel, Smith, and Chetco rivers, the other major populations in the ESU, as well as the numerous smaller coastal populations. Actual abundance is therefore likely to be higher than this estimate.

Table 39. Estimates of the natural and hatchery adult SONCC coho salmon returns (Chesney and Knechtle 2011; Knechtle and Chesney 2011; ODFW 2014c; Sinnen et al. 2011).

YEAR	Rogue River		Smith River Mill Creek	Trinity River		Klamath River		Redwood Creek	Freshwater Creek
						Shasta [a]	Scott		
	Hatchery	Natural	Natural	Hatchery	Natural	Natural	Natural	Natural	Natural
2003	7,296	6,805		24,211	3,941	187			
2004	9,092	24,509				373			731
2005	5,339	9,957		28,905	2,514	69			974
2006	3,496	3,911		18,673	1,405	47			789
2007	2,275	5,136		4,600	1,150	255	1,622		396
2008	158	414		8,684	1,298	31	62		262
2009	518	2,566		5,820	576	9	81		399
2010	752	3,671				44	927	373	89
2011	1,157	4,545				62	355	322	455
2012	1,423	5,474	482			115	> 201	803	624
2013	1,999	11,210	227					747	318
2014			260					705	
Average [b]	1,526	7,076	323	6,368	1,008	74	494	752	466

[a] Hatchery proportion unknown, but assumed to be low.

[b] 3-year average of most recent years of data.

While we currently lack data on naturally-produced juvenile SONCC coho salmon production, it is possible to make rough estimates of juvenile abundance from adult return data. Quinn (2005) published estimates for salmonids in which average fecundity for coho salmon is 2,878 eggs per female. By applying the average fecundity of 2,878 eggs per female to the estimated 5,096 females returning (half of the average total number of natural spawners), approximately 14 million eggs may be expected to be produced annually. Nickelson (1998) found survival of coho from egg to parr in Oregon coastal streams to be around seven percent. Thus, we can state that the ESU could produce roughly 1,026,707 juvenile natural SONCC coho salmon each year. In addition, hatchery managers could produce approximately 775,000 listed hatchery juvenile coho each year.

<u>Limiting Factors:</u> Threats from natural or man-made factors have worsened in the past 5 years, primarily due to four factors: small population dynamics, climate change, multi-year drought,

and poor ocean survival conditions (NMFS 2012c; NOAA Fisheries 2011). Limiting factors include:

- Lack of floodplain and channel structure
- Impaired water quality
- Altered hydrologic function (timing of volume of water flow)
- Impaired estuary/mainstem function
- Degraded riparian forest conditions
- Altered sediment supply
- Increased disease/predation/competition
- Barriers to migration
- Adverse fishery-related effects
- Adverse hatchery-related effects

4.2.15.8.4 Central California Coast Coho Salmon

NMFS originally listed the central California coast coho salmon ESU as threatened on October 31, 1996 (61 FR 56138) and later reclassified their status to endangered June 28, 2005 (70 FR 37160).

Spatial Structure and Diversity: Historically, the Central California Coast (CCC) coho salmon ESU comprised approximately 76 coho salmon populations. Most of these were dependent populations that needed immigration from other nearby populations to ensure their long term survival, as described above. Historically, there were 11 functionally independent populations and one potentially independent population of CCC coho salmon (Spence et al. 2008; Spence et al. 2012). Adams et al. (1999) found that in the mid 1990's coho salmon were present in only 51 percent (98 of 191) of the streams where they were historically present, although coho salmon were documented in 23 additional streams within the CCC coho salmon ESU for which there were no historical records. Recent genetic research in progress by both the SWFSC and the Bodega Marine Laboratory has documented a reduction in genetic diversity within subpopulations of the CCC coho salmon ESU (Bjorkstedt et al. 2005).

Abundance and Productivity: Brown et al. (1994) estimated that annual spawning numbers of coho salmon in California ranged between 200,000 and 500,000 fish in the 1940's, which declined to about 100,000 fish by the 1960's, followed by a further decline to about 31,000 fish by 1991. More recent abundance estimates vary from approximately 600 to 5,500 adults (Good et al. 2005a). Recent status reviews (Good et al. 2005a; Williams et al. 2011) indicate that the CCC coho salmon are likely continuing to decline in number and many independent populations that supported the species overall numbers and geographic distributions have been extirpated. The current average run size for the CCC coho salmon ESU is 1,621 fish (1,294 natural-origin; 327 hatchery produced) (Table 40).

Table 40. Geometric mean abundances of CCC coho salmon spawners in 2006-2012 escapements by population.

Stratum	Population	Years	Spawners		Expected Number of Outmigrants[cd]
			Natural-origin[a]	Hatchery-origin[b]	
Lost Coast – Navarro Point	**Ten Mile River**	2010-2011	237	-	16,940
	Usal Creek	2009-2011	5	-	
	Noyo River	2006-2010	398	-	38,150
	Pudding Creek	2008-2012	116	-	
	Caspar Creek	2006-2010	31	-	
	Big River	2009-2011	116	-	8,120
	Albion River	2009-2011	9	-	630
Navarro Point – Gualala Point	**Navarro River**	2009-2011	197	-	13,790
	Garcia River	2009-2011	34	-	2,380
	Gualala River	-	-	-	-
Coastal	**Russian River**	2008-2012	20	323	2,030
	Salmon Creek	2008	9	-	
	Walker Creek	2006-2008	4	-	280
	Lagunitas Creek	2007-2011	105	-	7,700
	Redwood Creek	2007-2011	5	-	
Santa Cruz Mountains	**Pescadero Creek**	-	-	-	-
	San Lorenzo River	-	-	-	560
	Waddell Creek	2007	2	-	
	Scott Creek	2008-2012	3	4	
	San Vicente Creek	2007-2008	1	-	
	Soquel Creek	2007	2	-	
ESU Average			**1,294**	**327**	**90,580**

[a] Source: http://swr.nmfs.noaa.gov/recovery/ccc_coho/

[b] J. Jahn, pers. comm., July 2, 2013 (cited in NMFS 2015)

[c] Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant

[d] Based upon natural-origin spawner numbers

While we currently lack data on how many natural juvenile coho salmon this ESU produces, it is possible to make rough estimates of juvenile abundance from adult return data. Sandercock (1991) published fecundity estimates for several coho salmon stocks; average fecundity ranged from 1,983 to 5,000 eggs per female. By applying a very conservative value of 2,000 eggs per female to an estimated 647 females returning (50 percent of the run) to this ESU, one may expect approximately 1.3 million eggs to be produced annually. Nickelson (1998) found survival of coho from egg to parr in Oregon coastal streams to be around 7 percent. Thus, we can estimate that roughly 90,000 juvenile coho salmon are produced annually by the CCC coho ESU.

Limiting Factors: Most of the populations in the CCC coho salmon ESU are currently doing poorly; low abundance, range constriction, fragmentation, and loss of genetic diversity is documented. The near-term (10 to 20 years) viability of many of the extant independent CCC coho salmon populations is of serious concern. These populations may not have enough fish to survive additional natural and human caused environmental change. NMFS has determined that currently depressed population conditions are, in part, the result of the following human-induced factors affecting critical habitat[15]: logging, agriculture, mining, urbanization, stream channelization, dams, wetland loss, and water withdrawals (including unscreened diversions for irrigation). Impacts of concern include altered stream bank and channel morphology, elevated water temperature, lost spawning and rearing habitat, habitat fragmentation, impaired gravel and wood recruitment from upstream sources, degraded water quality, lost riparian vegetation, and increased erosion into streams from upland areas ((Busby et al. 1996c; Weitkamp et al. 1995); 64 FR 24049; 70 FR 37160; 70 FR 52488). Diversion and storage of river and stream flow has dramatically altered the natural hydrologic cycle in many of the streams within the ESU.

4.2.16 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.

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[15] Other factors, such as over fishing and artificial propagation have also contributed to the current population status of these species. All these human induced factors have exacerbated the adverse effects of natural environmental variability from such factors as drought and poor ocean conditions.

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 also have a wholly freshwater life history form, called kokanee (Burgner 1991a). In some cases a single population will give rise to both the anadromous and freshwater life history form. While in fresh water juveniles of both life history types prey primarily upon insects. The presence of both life history types may be related to the energetic costs of outmigrating to sea, and the productivity of the lacustrine system they inhabit. In coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, kokanee populations are rare.

4.2.16.1 *Species Distribution*

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.

4.2.16.2 *Occurrence in the NWTT Action Area*

Adults of both the Snake River and the Ozette Lake ESUs will occur in the Offshore Area.

4.2.16.3 *Reproduction*

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Age at maturity varies by region from 2 to 5 years, but is generally 2 to 4 years in Washington State (Burgner 1991b). Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000 to 2,400 eggs per female up to 5,000 eggs, depending upon the population and average age of the female. Fecundity in kokanee is much lower and may range from about 300 to 2,000 eggs. Larger females are more fecund and realize greater reproductive success (Cunningham et al. 2013).

Incubation is a function of water temperatures, but generally lasts between 100 and 200 days (Burgner 1991b). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991b).

4.2.16.4 ***Habitat***

Once smolts enter the Pacific Ocean, they distribute widely across the North Pacific, generally above 40º N, where a current boundary is located. Season, temperature, salinity, life stage, age, size, availability of prey and population-of-origin are all factors that influence offshore movements (Burgner 1991b). Sockeye tend to occupy the upper water column, within 30 m of the surface and preferably within 15 m and tend to be closer to the surface at night versus daytime (Burgner 1991b; French et al. 1976b; Manzer 1964; Pauley et al. 1989). Sockeye migrate several thousand miles in search of prey and are considered to travel continuously (Royce et al. 1968). Juveniles leaving freshwater move north along the Alaskan coastal belt beginning in mid-May until early winter when they disperse into the Gulf of Alaska (Emmett et al. 1991a; NMFS-AKR 2005).

Thermoclines may also influence vertical distribution, with fish only mingling between surface and deeper waters when the boundary temperature difference is weak. Sockeye appear to prefer cooler waters relative to other salmon species, but younger salmon may prefer warmer sea surface temperatures (optimally 15° C) than larger, older fish (12° and 14° C), possibly an artifact of older fish being distributed further north (Pauley et al. 1989). Adult upstream migration may be blocked by temperatures above 21º C (McCullough 1999). However, temperatures below 21º C can stress fish by increasing their susceptibility to disease and elevating their metabolism (Brett 1979). Maturation and timing of return to spawn by sockeye appears to be linked to water temperature, with gonad development increasing in late May through early July (Nishiymama 1984).

4.2.16.5 ***Feeding***

While in freshwater, juveniles prey primarily upon insects. While at sea, sockeye prey upon a variety of organisms, including small fish (capelin, lantern fish, cod, sand lance, herring, and pollock), squid, crustacean larvae, krill, and other invertebrates (Foerster 1968; French et al. 1976b; Wing 1977).

4.2.16.6 ***Natural Threats***

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 2011e). In the ocean, marine mammals and other fish prey on sockeye, but the extent of such predation is not well known.

4.2.16.7 *Anthropogenic Threats*

Similar to the other salmonid species considered in this Opinion, sockeye salmon are also primarily threatened by direct and incidental harvest, hatcheries, lack of suitable habitat for some life stages, and hydroelectric dams inhibiting access to spawning habitat. 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 2011e).

4.2.16.8 *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. 2007). 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 sockeye 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).

4.2.16.9 *Sockeye Salmon ESUs*

Each sockeye salmon ESU is treated as a separate species under the ESA (NMFS 2005c). 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). The distribution, as well as the status and trends, of the sockeye salmon ESUs considered in this Opinion are discussed below. Ozette Lake sockeye salmon were considered in Section 4.1.6.

4.2.16.9.1 Snake River Sockeye Salmon

Snake River sockeye salmon were originally listed as endangered in 1991 and the NMFS retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160).

<u>Spatial Structure and Diversity:</u> Snake River (SR) sockeye includes all anadromous and residual sockeye salmon from the Snake River basin, Idaho, and artificially-propagated sockeye salmon from the Redfish Lake captive propagation program. The IC-TRT identified historical sockeye salmon production in at least five Stanley Basin and Sawtooth Valley lakes and in lake systems associated with Snake River tributaries currently cut off to anadromous access (e.g., Wallowa and Payette Lakes), although current returns of SR sockeye salmon are extremely low and limited to Redfish Lake (IC-TRT 2007).

<u>Abundance and Productivity</u>: This species is still at extremely high risk across all four basic risk measures (abundance, productivity, spatial structure and diversity. Although the captive brood program has been successful in providing substantial numbers of hatchery produced *O. nerka* for

use in supplementation efforts, substantial increases in survival rates across life history stages must occur to re-establish sustainable natural production (Hebdon et al. 2004; Keefer et al. 2008). Overall, although the risk status of the SR sockeye appears to be on an improving trend, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

Between 1997 and 2005, approximately 400 hatchery sockeye returned to the Stanley basin, total. Only 16 naturally produced adults returned to Redfish Lake between the time SR sockeye was listed as an endangered species in 1991 and 2005. Since that time, there has been a considerable improvement in the sockeye returns. From 2009 through 2012, an average of 1,348 adult sockeye (all from the broodstock program) passed Lower Granite Dam on their way to Redfish Lake. The year 2012 saw the lowest numbers of that period—with only 470 fish being counted at Lower Granite Dam. These numbers have been updated somewhat with the 2014 returns—which numbered 2,786 fish. The new four-year average return to Lower Granite Dam (through 2014) is 1,373 (IDFG data).

Each spring, the NWFSC produces a memorandum estimating the number of ESA-listed Pacific salmon and steelhead smolts expected to arrive at various locations in the Columbia River basin. The averages of the five most recent projections for the SR sockeye salmon juvenile emigrants are displayed below in Table 41.

Table 41. Recent five-year average projected outmigrations for SR sockeye (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	Outmigration
Natural	15,560
Listed Hatchery: Adipose Clipped	124,767

The BRT, reviewing the status of the species in 2010 (Ford 2011a), found that the recent increase in returns of hatchery-reared SR sockeye has reduced the risk of immediate loss, but that levels of naturally produced returns remain extremely low. Although the biological risk status of the ESU appeared to be on an improving trend, the new information did not indicate a change in category (extremely high risk) since the 2005 BRT status review.

Limiting Factors: The key factor limiting recovery of SR sockeye is survival outside of the Stanley Basin. Portions of the migration corridor in the Salmon River are impeded by water quality and temperature (Idaho Department of Environmental Quality 2011). Increased temperatures likely reduce the survival of adult sockeye returning to the Stanley Basin. The natural hydrological regime in the upper mainstem Salmon River Basin has been altered by water withdrawals. In most years, sockeye adult returns to Lower Granite suffer catastrophic losses (Reed et al. 2003) (e.g., > 50 percent mortality in one year) before reaching the Stanley Basin, although the factors causing these losses have not been identified. In the Columbia and lower Snake River migration corridor, predation rates on juvenile sockeye salmon are unknown, but terns and cormorants consume 12 percent of all salmon smolts reaching the estuary, and piscivorous fish consume an estimated 8 percent of migrating juvenile salmon (NOAA Fisheries 2011).

4.2.17 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).

4.2.17.1 *Species Distribution*

Steelhead, the common name of the anadromous form of *O. mykiss*, are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Good et al. 2005c; Good et al. 2005b; Moyle 1976; NMFS 1997b; Stolz and Schnell 1991).

4.2.17.1.1 Occurrence in the NWTT Action Area

All steelhead DPSs may occur in the Offshore Area portion of the Action Area. The Puget Sound DPS will occur in both the Inland Waters and Offshore Area portions of the Action Area.

4.2.17.2 *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. 2010a).

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).

4.2.17.3 *Habitat*

Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity. Intermittent streams may also be used for spawning (Barnhart 1986a; 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 1986b; 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 1986b). 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 1986b). 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).

In marine waters, steelhead reside from the surface down to 200 m in waters with temperatures up to 24° C, although 10° C is optimum (Pauley et al. 1986b).

4.2.17.4 *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)).

4.2.17.5 *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. 2007). 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).

4.2.17.6 *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 2011e). In the ocean, marine mammals and other fish prey on steelhead, but the extent of such predation is not well known.

4.2.17.7 *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 2011e).

4.2.17.8 *Steelhead DPSs*

Each steelhead DPS is treated as a separate species under the ESA (NMFS 2005c). 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.

4.2.17.8.1 Lower Columbia River Steelhead

The NMFS listed Lower Columbia River steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834).

Spatial Structure and Diversity: Four strata and 23 historical populations of Lower Columbia River (LCR) steelhead occur within the DPS: 17 winter-run populations and six summer-run populations, within the Cascade and Gorge ecological subregions.[16] The DPS also includes the progeny of ten artificial propagation programs.[17] Summer steelhead return to freshwater long before spawning. Winter steelhead, in contrast, return from the ocean much closer to maturity and spawn within a few weeks. Summer steelhead spawning areas in the Lower Columbia River are found above waterfalls and other features that create seasonal barriers to migration. Where no temporal barriers exist, the winter-run life history dominates.

It is likely that genetic and life history diversity has been reduced as a result of pervasive hatchery effects and population bottlenecks. Spatial structure remains relatively high for most populations. Out of the 23 populations, 16 are considered to have a “low” or “very low” probability of persisting over the next 100 years, and six populations have a “moderate” probability of persistence (Ford 2011b; Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a). All four strata in the DPS fall short of the WLC-TRT criteria for viability (NMFS 2013).

Baseline persistence probabilities were estimated to be “low” or “very low” for three out of the six summer steelhead populations that are part of the LCR steelhead DPS, moderate for two, and high for one, the Wind, which is considered viable. Thirteen of the 17 LCR winter steelhead populations have “low” or “very low” baseline probabilities of persistence, and the remaining

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[16] The White Salmon and Little White Salmon steelhead populations are part of the Middle Columbia steelhead DPS and are addressed in a separate species-level recovery plan, the Middle Columbia River Steelhead Distinct Population Segment ESA Recovery Plan NMFS. 2009a. Middle Columbia River steelhead distinct population segment ESA recovery plan. November 30.

[17] In 2007, the release of Cowlitz Hatchery winter steelhead into the Tilton River was discontinued; in 2009, the Hood River winter steelhead program was discontinued; and in 2010, the release of hatchery winter steelhead into the Upper Cowlitz and Cispus rivers was discontinued. In 2011, NMFS recommended removing these programs from the DPS. A Lewis River winter steelhead program was initiated in 2009, and in 2011, NMFS proposed that it be included in the DPS NMFS. 2011a. 5-year review: summary and evaluation of Lower Columbia River Chinook, Columbia River chum, Lower Columbia River coho, and Lower Columbia River steelhead. National Marine Fisheries Service, editor, Portland, Oregon.

four are at "moderate" probability of persistence (Lower Columbia Fish Recovery Board 2010a; NMFS 2013; ODFW 2010a).

Abundance and Productivity: The "low" to "very low" baseline persistence probabilities of most LCR steelhead populations reflects low abundance and productivity (NMFS 2013). All of the populations increased in abundance during the early 2000s, generally peaking in 2004. Most populations have since declined back to levels within one standard deviation of the long term mean. Exceptions are the Washougal summer-run and North Fork Toutle winter-run, which are still higher than the long term average, and the Sandy, which is lower. In general, the populations do not show any sustained dramatic changes in abundance or fraction of hatchery origin spawners since the 2005 status review (Ford 2011b). Although current LCR steelhead populations are depressed compared to historical levels and long-term trends show declines, many populations are substantially healthier than their salmon counterparts, typically because of better habitat conditions in core steelhead production areas (Lower Columbia Fish Recovery Board 2010a; NMFS 2013).

Since the last status evaluation, all populations increased in abundance during the early 2000s, generally peaking in 2004. Abundance of most populations has since declined back to levels close to the long-term mean. Exceptions are the Washougal summer and North Fork Toutle winter populations, for which abundance is higher than the long-term average, and the Sandy, for which abundance is below the long-term average. The North Fork Toutle winter steelhead population appears to be experiencing an increasing trend dating back to 1990, which is likely partially the result of recovery of habitat since the eruption of Mt. St. Helens in 1980. In general, the LCR steelhead populations do not show any sustained, dramatic changes in abundance since the previous status review (Ford 2011a).

Table 42. Abundance estimates for LCR steelhead populations (NWFSC 2015; ODFW 2010b; Streamnet 2014; WDFW 2010a; WDFW 2010b; WDFW 2011).

Stratum (Run)	Population	Years	HOR(1)	NOR(2)	Recovery Target(3)
Cascade (Winter)	Upper Cowlitz and Cispus	2009-2013	614	535	
	Lower Cowlitz	2009	4559	400	500
	Tilton	2009-2013	256	251	200
	Coweeman	2009-2013	181	483	500
	South Fork Toutle	2009-2013	5	466	600
	North Fork Toutle	2009-2013	99	530	600
	Kalama	2009-2013	433	900	600
	North Fork Lewis	2009-2013	2,126		400
	East Fork Lewis	2009-2013	0	418	500
	Washougal	2009-2013	203	368	350
	Clackamas	2004-2008	682	1,669	10,655
	Sandy	2002-2006	0	769	1,510
Cascade (Summer)	Kalama	2009-2013	334	518	500
	North Fork Lewis	2009-2013	10,508		
	East Fork Lewis	2009-2013	114	916	500
	Washougal	2009-2013	605	704	500
Gorge (Winter)	Lower Gorge				1,104
	Upper Gorge	2009-2013		41	322
	Hood	2003-2007	380	438	1,633
Gorge (Summer)	Wind	2009-2013	42	866	1,000
	Hood	2003-2007	239	241	1,988

(1) Hatchery Origin (HOR) spawners.

(2) Natural Origin (NOR) spawners.

The Columbia River Compact, a joint effort between ODFW and WDFW, monitors salmon, steelhead, and other species abundance and harvest limits in the Columbia River basin. The Compact publishes Joint Staff Reports describing management guidelines for seasonal fisheries in the mainstem Columbia River and expectations for annual salmon and summer steelhead returns (http://wdfw.wa.gov/fishing/crc/). For the years 2009 through 2013, the average return of wild winter steelhead to the Columbia River mouth was 15,931, of which 9,393 were LCR steelhead DPS. Adding the estimated abundance of summer steelhead (Table 42) to this the average we would expect an annual return of roughly 11,117 naturally produced adult LCR steelhead. Also from Table 42, we would expect an annual return of roughly 23,000 hatchery produced adult LCR steelhead.

NWFSC publishes juvenile abundance estimates each year in the annual memorandum estimating percentages of listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. Numbers for 2015 are not available at this time; however the average outmigration for the years 2010 through 2014 is shown in Table 43 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Table 43. Average estimated outmigration for ESA-listed LCR steelhead (2010-2014).

Origin	Outmigration
Natural	447,659
Listed hatchery intact adipose	2,428
Listed hatchery adipose clipped	1,025,729

<u>Limiting Factors</u> include (NMFS 2013; NOAA Fisheries 2011):

- Degraded estuarine and nearshore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and recruitment of large wood, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Reduced access to spawning and rearing habitat mainly as a result of tributary hydropower projects and lowland development
- Avian and marine mammal predation in the lower mainstem Columbia River and estuary
- Hatchery-related effects
- An altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity
- Reduced access to off-channel rearing habitat in the lower Columbia River
- Reduced productivity resulting from sediment and nutrient-related changes in the estuary
- Juvenile fish strandings that result from ship wakes
- Contaminants affecting fish health and reproduction

4.2.17.8.2 Middle Columbia River Steelhead

Middle Columbia River steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834).

Spatial Structure and Diversity***:*** Middle Columbia River (MCR) steelhead includes all naturally-spawned steelhead populations below natural and artificial impassable barriers in streams from above the Wind River, Washington, and the Hood River, Oregon (exclusive), upstream to, and including, the Yakima River, Washington, excluding steelhead from the Snake River basin; and progeny of seven artificial propagation programs. The IC-TRT identified 17 extant populations in this DPS (NMFS 2011g). The populations fall into four major population groups: the Yakima River Basin (four extant populations), the Umatilla/Walla-Walla drainages (three extant and one extirpated populations); the John Day River drainage (five extant populations) and the Eastern Cascades group (five extant and two extirpated populations) (Ford 2011b; NMFS 2009a).

Abundance and Productivity: Returns to the Yakima River basin and to the Umatilla and Walla Walla Rivers have been higher over the most recent brood cycle, while natural origin returns to the John Day River have decreased. There have been improvements in the viability ratings for some of the component populations, but the MCR steelhead DPS is not currently meeting the viability criteria (adopted from the IC-TRT) in the MCR steelhead recovery plan (NMFS 2009a). In addition, several of the factors cited by Good et al. (2005d) remain as concerns or key uncertainties. Natural origin spawning estimates of populations have been highly variable with respect to meeting minimum abundance thresholds. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

The species' populations are generally well below the ICTRT's abundance thresholds for viability. Only the Deschutes (Eastside), Fifteen Mile Creek, and the North Fork John Day populations have recent 10-year averages that exceed the thresholds; the other 14 extant populations are all below the thresholds (though some are not far below) and a few represent only fractions of the numbers needed for viability (see Table 44). On a positive note, the most recent 20-year productivity averages are showing greater-than-replacement levels in all populations for which we have data. Moreover, from the year 2009 through the year 2013, the five-year average return to the ESU was 26,851 adult fish (NWFSC 2015); of these, approximately 9 percent were of hatchery origin.

Table 44. Annual escapement estimates of adult MCR steelhead for the years 2009-2012 (NWFSC 2015).

	Hatchery Origin				Natural Origin			
Population	**2009**	**2010**	**2011**	**2012**	**2009**	**2010**	**2011**	**2012**
Klickitat	842	822	302	293	391	120	296	179
Fifteenmile Creek	0	0	33	27	395	814	383	530
Deschutes River - eastside	145	226	340	506	1,662	1,385	1,466	1,949
Deschutes River - westside	93	110	62	25	328	893	1,175	1,206
Lower Mainstem John Day	778	230	419	654	3,546	1,121	2,197	3,436
Upper Mainstem John Day	31	23	22	21	732	739	1,062	1,035
North Fork John Day	163	91	59	94	3,909	2,931	2,906	4,589
Middle Fork John Day	85	55	74	70	2,029	1,765	3,618	3,424
South Fork John Day	73	13	19	42	1,758	418	915	2,057
Umatilla River	586	926	631	778	2,344	3,702	3,879	3,111
Walla Walla River	18	50	33	50	861	1,615	1,628	1,211
Touchet River	137	182	70	83	279	828	470	293
Satus Creek	11	85	46	78	1,042	2,745	2,278	1,877
Toppenish Creek	7	19	16	29	692	620	801	696
Naches River	9	62	42	65	1,114	2,138	1,965	1,657
Upper Yakima	0	19	11	15	216	366	365	354
Totals	2,976	2,911	2,179	2,830	21,300	22,202	25,404	27,603
Averages	Hatchery Origin Average =			2,724	Natural Origin Average =			24,127

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of ESA-listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. The averages of the five most recent projections for the MCR juvenile outmigration are displayed below in Table 45.

Table 45. Recent five-year average projected outmigrations for MCR steelhead (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	Outmigration
Natural	609,458
Listed Hatchery: Adipose Clipped*	341,721
Listed Hatchery: Intact Adipose*	422,853

*When the above species was listed, NMFS included certain artificially propagated (hatchery-origin) populations in the listing. Some of those listed fish have had their adipose fins clipped at their respective hatcheries and some have not.

Limiting Factors include (NMFS 2009a; NOAA Fisheries 2011):

- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas, fish passage, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, tributary hydro system activities, and development
- Mainstem Columbia River hydropower–related impacts
- Degraded estuarine and nearshore marine habitat
- Hatchery-related effects
- Harvest-related effects
- Effects of predation, competition, and disease

4.2.17.8.3 Upper Columbia River Steelhead

Upper Columbia River steelhead were originally listed as endangered in 1997 (62 FR 43937), after their status was reviewed, they were reclassified to threatened on January 5, 2006 and then reinstated to endangered status per U.S. District Court decision in June 2007 (62 FR 43937).

Spatial Structure and Diversity: Upper Columbia River (UCR) steelhead 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 were identified by the IC-TRT in the same upriver tributaries as for UC spring-run Chinook salmon (i.e., Wenatchee, Entiat, Methow, and Okanogan) and, similarly, no major population groupings were identified due to the relatively small geographic area involve (Ford 2011b; NMFS 2011g). All extant populations are considered to be at high risk of extinction (Ford 2011b). With the exception of the Okanogan population, the Upper Columbia populations rated as "low" risk for spatial structure. The "high" risk ratings for SS/D are largely driven by chronic high levels of hatchery spawners within natural spawning areas and lack of genetic diversity among the populations. The proportions of hatchery origin returns in natural spawning areas remain extremely high across the DPS, especially in the Methow and Okanogan River populations. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

Abundance and Productivity: UCR steelhead populations have increased in natural origin abundance in recent years, but productivity levels remain low. The modest improvements in natural returns in recent years are probably primarily the result of several years of relatively good natural survival in the ocean and tributary habitats.

A review of data from the past several years indicates that natural steelhead abundance has declined or remained low in the major river basins occupied by this species since the early

1990s. However, returns of both hatchery and naturally produced steelhead to the upper Columbia have increased somewhat in recent years. Priest Rapids Dam is below the UCR steelhead production areas. The average 1997 to 2001 returns, counted at the Priest Rapids fish ladder, were approximately 12,900 steelhead. The average for the five years from 1992 through 1996 was 7,800 fish. In 2004 and 2005, it is estimated that totals of 18,727 and 12,143 UCR steelhead (respectively) returned to their spawning grounds (Fish Passage Center 2005; Pacific Coastal Salmon Recovery Fund 2007). However, returns to the upper Columbia are composed primarily of hatchery-origin fish. The percentage of the run over Priest Rapids of natural origin fish increased to over 25 percent in the 1980s then dropped to less than 10 percent by the mid-1990s. The median percent wild for 1997 to 2001 was 17 percent (Good et al. 2005a). More recent data show that these trends have continued. Moreover, from the year 2009 through the year 2013, the five-year average escapement of UCR steelhead was 2,728 naturally produced adult fish and 7,936 hatchery propagated adult fish (Table 46).

Table 46. Average numbers of adult UCR steelhead (NWFSC 2015).

Population	Years	Total	Hatchery Origin	Natural Origin
Entiat River	2009-2013	777	574	203
Methow River	2009-2013	4,438	3,640	798
Okanogan River	2009-2013	2,346	2,117	229
Wenatchee River	2009-2013	3,103	1,605	1,498
Total		10,664	7,936	2,728

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of ESA-listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. The averages of the five most recent projections for the UCR steelhead juvenile outmigration are displayed below in Table 47.

Table 47. Recent five-year average projected outmigrations for UCR steelhead (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	Outmigration
Natural	286,452
Listed Hatchery: Adipose Clipped*	658,692
Listed Hatchery: Intact Adipose*	175,528

*When the above species was listed, NMFS included certain artificially propagated (hatchery-origin) populations in the listing. Some of those listed fish have had their adipose fins clipped at their respective hatcheries and some have not.

Limiting Factors include (NOAA Fisheries 2011; Upper Columbia Salmon Recovery Board 2007):

- Mainstem Columbia River hydropower–related adverse effects
- Impaired tributary fish passage
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development.
- Effects of predation, competition, and disease mortality: Fish management, including past introductions and persistence of non-native (exotic) fish species continues to affect habitat conditions for listed species.
- Hatchery-related effects
- Harvest-related effects

4.2.17.8.4 Upper Willamette River Steelhead

Upper Willamette River steelhead were listed as threatened in 1999 (64 FR 14517) and when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834).

Spatial Structure and Diversity: Upper Willamette River (UWR) steelhead 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 the Calapooia River. One stratum and four extant populations of UWR steelhead occur within the DPS. 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, the WLC-TRT recognized that 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. Hatchery summer-run steelhead that are released in the subbasins are from an out-of-basin stock, not part of the DPS. Additionally, stocked summer steelhead that have become established in the McKenzie River were not considered in the identification of historical populations (ODFW and NMFS 2011).

Abundance and Productivity: Since the last status review in 2005, UWR steelhead initially increased in abundance but subsequently declines and current abundance is at the levels observed in the mid-1990s when the DPS was first listed. The DPS appears to be at lower risk than the UWR Chinook salmon ESU, but continues to demonstrate the overall low abundance pattern that was of concern during the last status review. 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. Overall, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

The majority of the UWR winter steelhead run return to freshwater in January through April, pass Willamette Falls from mid-February to mid-May, and spawn in March through June. Adult winter-run steelhead are counted at the Willamette Falls fishway ladder where the counts begin in November and end mid-May of the following year (Table 48). The number of winter-run steelhead passing over Willamette Falls during the winter of 2013 was 4,944 and the most recent five-year average is only at 6,030.

Table 48. Upper Willamette winter-run steelhead abundance (ODFW 2014b).

Year	Natural-origin Spawners
2009	2,813
2010	7,337
2011	7,441
2012	7,616
2013	4,944
Average	6,030

It is difficult to accurately estimate juvenile UWR steelhead abundance during the coming year. However, the average estimated outmigration (2010 to 2014) of naturally-produced smolts is 215,847 (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b). As with other species, it is reasonable to assume that this figure could be substantially higher when other juvenile life stages are included. In addition, non-listed juvenile rainbow trout and unlisted juvenile steelhead occur in the same areas as the listed UWR steelhead; and it is very difficult to distinguish between them.

Limiting Factors include (NOAA Fisheries 2011; ODFW and NMFS 2011):

- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood recruitment, and stream flow have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Degraded water quality and altered temperature as a result of both tributary dams and the cumulative impacts of agriculture, forestry, and urban development
- Reduced access to spawning and rearing habitats mainly as a result of artificial barriers in spawning tributaries
- Hatchery-related effects: impacts from the non-native summer steelhead hatchery program

- Anthropogenic introductions of non-native species and out-of-ESU races of salmon or steelhead have increased predation and competition on native UWR steelhead.

4.2.17.8.5 Snake River Basin Steelhead

Snake River steelhead were listed as threatened in 1997 (62 FR 43937), when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834).

Spatial Structure and Diversity: Snake River (SR) Basin steelhead 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. The IC-TRT identified 24 populations in five major groups (Ford 2011b; NMFS 2011g). The IC-TRT has not assessed the viability of this species. The relative proportion of hatchery fish in natural spawning areas near major hatchery release sites is highly uncertain. There is little evidence for substantial change in ESU viability relative to the previous BRT and IC-TRT reviews. Overall, therefore, the new information considered does not indicate a change in the biological risk category since the last status review (Ford 2011b).

Abundance and Productivity: The level of natural production in the two populations with full data series and the Asotin Creek index reaches is encouraging, but the status of most populations in this DPS remains highly uncertain. Population-level natural origin abundance and productivity inferred from aggregate data and juvenile indices indicate that many populations are likely below the minimum combinations defined by the IC-TRT viability criteria.

The longest consistent indicator of steelhead abundance in the Snake River basin is derived from counts of natural-origin steelhead at the uppermost dam on the lower Snake River (Lower Granite Dam). According to these estimates, the abundance of natural-origin steelhead at the uppermost dam on the Snake River has declined from a 4-year average of 58,300 in 1964 to a 4-year average of 8,300 ending in 1998. In general, steelhead abundance declined sharply in the early 1970s, rebuilt modestly from the mid-1970s through the 1980s, and declined again during the 1990s. From the year 2004 through the year 2009, the five-year average return to the ESU was 162,323 adult fish (Ford 2011); of these, approximately 90 percent were of hatchery origin (Pacific Coastal Salmon Recovery Fund 2007).

With a few exceptions, annual estimates of steelhead returns to specific production areas within the Snake River are not available. Overall, from the year 2010 through the year 2014, the five-year average escapement of SR steelhead was 46,336 naturally produced adult fish and 139,528 hatchery propagated adult fish (Table 49).

Table 49. Estimated numbers of adult Snake River steelhead (ODFW/WDFW 2014).

Run Year	A-run		B-run		Total	
	Hatchery	Wild	Hatchery	Wild	Hatchery	Wild
2009-10	260,095	39,759	19,048	4,480	279,143	44,239
2010-11	128,132	34,362	35,324	10,478	163,457	44,839
2011-12	120,643	35,471	19,526	4,680	140,169	40,151
2012-13	67,128	20,786	15,881	5,387	83,009	26,173
2013-14	74,000	25,058	6,802	2,278	31,860	76,278
Average	130,000	31,087	19,316	5,461	139,528	46,336

Juvenile abundance estimates are published each spring in an annual memorandum estimating percentage of ESA-listed Pacific salmon and steelhead smolts arriving at various locations in the Columbia River basin. The averages of the five most recent projections for the SR steelhead juvenile outmigration are displayed below in Table 50.

Table 50. Recent five-year average projected outmigrations for SR steelhead (Dey 2012; Ferguson 2010; Zabel 2014a; Zabel 2014b).

Origin	Outmigration
Natural	1,399,511
Listed Hatchery: Adipose Clipped	3,075,195
Listed Hatchery: Intact Adipose	971,028

Limiting Factors include (NMFS 2011c; NMFS 2011g):

- Mainstem Columbia River hydropower–related adverse effects
- Impaired tributary fish passage
- Degraded freshwater habitat: Floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development
- Impaired water quality and increased water temperature
- Related harvest effects, particularly for B-run steelhead
- Predation
- Genetic diversity effects from out-of-population hatchery releases

4.2.17.8.6 Puget Sound Steelhead

Puget Sound steelhead were listed as threatened under the ESA on May 11, 2007 (72 FR 26722) and affirmed as threatened on August 15, 2011 (76 FR 50448).

Spatial Structure and Diversity: Steelhead populations can be divided into two basic reproductive ecotypes, based on the state of sexual maturity at the time of river entry (summer or winter) and duration of spawning migration (Burgner et al. 1992)). The Puget Sound (PS) DPS includes all naturally spawned anadromous winter-run and summer-run steelhead populations in streams in the river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington, bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive), as well as the Green River natural and Hamma Hamma winter-run steelhead hatchery stocks. Non-anadromous ''resident'' *O. mykiss* occur within the range of PS steelhead but are not part of the DPS due to marked differences in physical, physiological,

The PS Steelhead TRT has completed a set of simple population viability analyses (PVAs) for these draft populations and MPGs within the DPS. No new estimates of productivity, spatial structure and diversity of PS steelhead have been made available since the 2007 review, when the BRT concluded that low and declining abundance and low and declining productivity were substantial risk factors for the species (USDC 2007). Loss of diversity and spatial structure were judged to be "moderate" risk factors. Since the listing of this species, this threat has not changed appreciably (Ford 2011b).

Abundance and Productivity: The BRT considered the major risk factors facing PS steelhead to be: widespread declines in abundance and productivity for most natural steelhead populations in the ESU, including those in Skagit and Snohomish rivers (previously considered to be strongholds); the low abundance of several summer-run populations; and the sharply diminishing abundance of some steelhead populations, especially in south Puget Sound, Hood Canal, and the Strait of Juan de Fuca (Hard et al. 2007). For all but a few putative PS steelhead populations, estimates of mean population growth rates obtained from observed spawner or redd counts are declining—typically 3 to 10 percent annually—and extinction risk within 100 years for most populations in the DPS is estimated to be moderate to high, especially for draft populations in the putative South Sound and Olympic MPGs. Most populations within the DPS continue downward trends in estimated abundance, a few sharply so. Extinction risk within 100 years for most populations in the DPS is estimated to be moderate to high, especially for populations in the South Sound and Olympic MPGs.

For the most recent 5-year period of escapement estimates (2009 through 2013), run size is 14,615 spawners (Table 51).

Table 51. Abundance of PS steelhead spawner escapements (natural-origin and hatchery production combined) from 2009-2013.

Populations	Run	Years	Spawners[a]			Expected Number of Outmigrants[b]
			Total	NOR	HOR	
Nooksack River [c]	Winter	2009-2012	1,472	1,472	0	167,440
Samish River [c]	Winter	2009-2013	748	748	0	85,085
Skagit River [c]	Summer/Winter	2009-2013	6,385	6,066	319	726,294
Stillaguamish River [c]	Winter	2009-2013	387	313	74	44,021
Snohomish-Skykomish Rivers [c]	Winter	2009-2013	2,387	1,933	454	271,521
Cedar River [d]	Winter	2009-2012	1	1	0	114
Green River [c]	Winter	2009-2013	621	590	31	70,639
Puyallup/Carbon River [c]	Winter	2009-2013	386	317	69	43,908
White River [c]	Winter	2009-2013	603	603	0	68,591
Nisqually River [c]	Winter	2009-2013	478	421	57	54,373
East and South Hood Canal [c]	Winter	2009-2013	139	139	0	15,811
Skokomish River [c]	Winter	2009-2013	602	602	0	68,478
West Hood Canal [c]	Winter	2009-2013	226	226	0	25,708
Dungeness River [e]	Winter	2009-2013	26	26	0	2,958
Sequim/Discovery Bay Independent Tributaries [d]	Winter	2009-2013	22	22	0	2,503
Strait of Juan de Fuca Independent Tributaries [d]	Winter	2009-2013	132	132	0	15,015
DPS average			14,615	13,621	994	1,668,371

a Geometric mean of post fishery spawners.

b Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant.

c Source: Unpublished data, Robert Leland, WDFW, Nov. 24, 2014; cited in NMFS (2015).

d Source: https://fortress.wa.gov/dfw/score/score/species/steelhead.jsp?species=Steelhead

e Source: Point No Point Treaty Council et al. 2013

Steelhead are most abundant in the northern Puget Sound, with the Skagit and Snohomish rivers supporting the two largest winter-run steelhead populations. Hood Canal and Strait of Juan de Fuca populations are generally small with their populations averaging fewer than 400 natural-origin spawners annually.

Juvenile PS steelhead abundance estimates come from the escapement data. For the species, fecundity estimates range from 3,500 to 12,000; and the male to female ratio averages 1:1

(Pauley et al. 1986a). By applying a conservative fecundity estimate of 3,500 eggs to the expected escapement of females (half of the escapement of both natural-origin and hatchery-origin spawners – 7,308 females), 25.58 million eggs are expected to be produced annually. With an estimated survival rate of 6.5 percent (Ward and Slaney 1993), the DPS should produce roughly 1.7 million natural outmigrants annually.

Limiting factors include (NOAA Fisheries 2011):

- Widespread declines in adult abundance (total run size), despite significant reductions in harvest in recent years.
- Threats to diversity posed by use of two hatchery steelhead stocks (Chambers Creek and Skamania) inconsistent with wild stock diversity throughout the DPS.
- Declining diversity in the DPS, including the uncertain but weak status of summer-run fish in the DPS.
- A reduction in spatial structure for steelhead in the DPS.
- Reduced habitat quality through changes in river hydrology, temperature profile, downstream gravel recruitment, and reduced movement of large woody debris.
- Increased flood frequency and peak flows during storms, reduced groundwater-driven summer flows in the lower reaches of many rivers and their tributaries in Puget Sound where urban development has occurred, has resulted in gravel scour, bank erosion, and sediment deposition.
- Dikes, hardening of banks with riprap, and channelization, which have reduced river braiding and sinuosity, have increased the likelihood of gravel scour and dislocation of rearing juveniles.

4.2.17.8.7 Northern California Steelhead

Northern California steelhead were listed as threatened on June 7, 2000 (65 FR 36074), and when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834).

Spatial Structure and Diversity: The range of Northern California (NC) steelhead DPS is defined to include all naturally spawned populations of steelhead in California coastal basins from Redwood Creek in Humboldt county southward to, but not including the Russian River (71 FR 834-862). It includes the basins of all the rivers and streams tributary to the Pacific Ocean between these two streams, including those in Humboldt, Trinity, Mendocino, Sonoma, Lake, Glenn, Colusa, and Tehama counties. Comprehensive geographic distribution information of areas currently occupied is not available for this DPS, but NC steelhead remain widely distributed (Williams et al. 2011).

Abundance and Productivity: NMFS has recognized the decline of NC steelhead populations for more than a dozen years, with available historical NC steelhead abundance data summarized and published in the NMFS west coast steelhead status review by Busby et al. (1996c). Prior to 1960, estimates of abundance specific to this ESU were available from dam counts in the upper Eel River (Cape Horn Dam–annual average of 4,400 adult steelhead in the 1930s), the South Fork Eel River (Benbow Dam–annual average of 19,000 adult steelhead in the 1940s), and the Mad River (Sweasey Dam–annual average of 3,800 adult steelhead in the 1940s). In the mid-1960s, estimates of steelhead spawning populations for many rivers in this ESU totaled 198,000. By the mid 1990's, the only available estimates for this area were counts at Cape Horn Dam on the Eel River where an average of 115 total and 30 wild adults were reported (Busby et al. 1996c). The most recent status review update by Williams et al. (2011) reports a mixture of patterns in population trend information, with more populations showing declines than increases. However, given the preponderance of significant negative trends in the available data, there is concern that steelhead populations in this ESU may not be self-sustaining.

NC steelhead abundance and productivity has only recently been monitored. In 2009, CDFW began studies to determine population-level abundance estimates of NC steelhead. Three streams/rivers have fish counting facilities: Caspar Creek (weir), Pudding Creek (fish ladder), and South Fork Noyo River (dam). From these studies, we estimate that the NC steelhead DPS has an annual abundance of 3,607 adults (Table 52).

Table 52. Geometric mean abundances of NC steelhead spawners from 2006-2012 escapements by population.

Stratum	Waterbody	Years	Abundance	Expected Number of Outmigrants[a]
Northern Coastal	Elk Creek[b]	2011	59	6,711
	Little River[b]	2009-2013	13	1,479
	Mattole River[c]	2007-2011	17	1,934
	Redwood Creek[g]	2010-2013	610	69,388
	Freshwater Creek[h]	2009-2013	85	9,669
North Mountain-Interior	Eel River[e]	2008-2011	332	37,765
North-Central Coastal	Big River[b]	2009-2012	249	28,324
	Caspar Creek[b]	2009-2013	34	3,868
	Cottoneva Creek[b]	2009-2010, 2012	25	2,844
	Hare Creek[b]	2010-2011	2	228
	Juan Creek[b]	2012	39	4,436
	Noyo River[b]	2009-2012	407	46,296
	SF Noyo River[b]	2009-2013	101	11,489
	Pudding Creek[b]	2009-2012	27	3,071
	Ten Mile River[b]	2010-2012	290	32,988
	Usal Creek[b]	2009-2012	52	5,915
	Wages Creek[b]	2009-2011	58	6,598
Central Coastal	Albion River[b]	2009-2012	32	3,640
	Big Salmon Creek[b]	2009, 2012	101	11,489
	Brush Creek[b]	2009-2012	6	683
	Garcia River[b]	2009-2012	366	41,633
	Gualala River[b]	2006-2010	1,066	121,258
	Navarro River[b]	2009-2012	315	35,831
Total			4,286	487,533

[a]Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant

[b](Gallagher et al. 2013; Gallagher and Wright 2011; Gallagher and Wright 2012)

[c](Group 2011)

[e]Counts at Van Arsdale Fisheries Station (http://www.pottervalleywater.org/files/VAFS_fish_counts.csv), Harris and Thompson 2014

[g] (Metheny and Duffy 2014)

[h] (Ricker et al. 2014)

Both adult and juvenile abundance data is limited for this DPS. While we currently lack data on naturally-produced juvenile NC steelhead, it is possible to make rough estimates of juvenile abundance from the available adult return data. Juvenile NC steelhead abundance estimates come from the escapement data. For the species, fecundity estimates range from 3,500 to 12,000; and the male to female ratio averages 1:1 (Pauley et al. 1986a). By applying a conservative fecundity estimate of 3,500 eggs to the expected escapement of females (half of the escapement of spawners – 2,143 females), 7.5 million eggs are expected to be produced annually. With an estimated survival rate of 6.5 percent (Ward and Slaney 1993), the DPS should produce roughly 487,533 natural outmigrants annually (Table 52).

Limiting Factors: Land management activities such as timber harvest, agriculture, and mining have resulted in significant instream habitat degradation for NC steelhead, and continue to contribute to poor habitat conditions (65 FR 36074). It is known that dams on the Mad River and Eel River block large amounts of habitat historically used by NC steelhead (Busby et al. 1996c; Spence et al. 2008). Also, the proportion of hatchery returns compared to wild stocks in recent returns to the Mad and Eel river basins have exposed their respective wild population to genetic introgression and the potential for deleterious interactions between native stock and introduced steelhead (Williams et al. 2011). Historical hatchery practices at the Mad River hatchery are of particular concern, and included out-planting of non-native Mad River hatchery fish to other streams in the DPS and the production of non-native summer steelhead (65 FR 36074). The conclusion of the 2005 status review (Good et al. 2005a) echoes that of previous reviews. Abundance and productivity in this DPS are of most concern, relative to NC steelhead spatial structure (distribution on the landscape) and diversity (level of genetic introgression). The lack of data available also remains a risk because of uncertainty regarding the condition of some stream populations.

4.2.17.8.8 Central California Coast Steelhead

The Central California Coast steelhead DPS was listed as a threatened species on August 18, 1997 (62 FR 43937) and was reaffirmed on January 5, 2006 (71 FR 834).

Spatial Structure and Diversity: On August 18, 1997, NMFS listed Central California Coast (CCC) steelhead—both natural and some artificially-propagated fish—as a threatened species (62 FR 43937). NMFS promulgated updated 4(d) protective regulations for CCC steelhead on January 5, 2006 (71 FR 834). The section 4(d) protections (and limits on them) apply to natural and hatchery CCC steelhead with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed.

The CCC steelhead DPS includes winter-run steelhead populations from the Russian River (Sonoma County) south to Aptos Creek (Santa Cruz County) inclusive and eastward to Chipps Island (confluence of the Sacramento and San Joaquin rivers) and including all drainages of San

Francisco, San Pablo, and Suisun bays. Two artificial propagation programs were listed as part of the DPS—Scott Creek/Kingfisher Flat Hatchery (includes San Lorenzo River production) and Don Clausen Fish Hatchery (includes Coyote Valley Fish Facility production) winter-run steelhead hatchery stocks (Table 53).

Table 53. Approximate annual releases of hatchery CCC steelhead (J. Jahn, pers. comm., July 2, 2013; cited in NMFS 2015).

Artificial propagation program	Adipose Fin-Clipped
Scott Creek/Kingfisher Flat Hatchery	3,220
San Lorenzo River	19,125
Don Clausen Fish Hatchery	380,338
Coyote Valley Fish Facility	246,208
Total Annual Release Number	648,891

Historically, approximately 70 populations of steelhead existed in the CCC steelhead DPS (Spence et al. 2008; Spence et al. 2012). Many of these populations (about 37) were independent, or potentially independent, meaning they had a high likelihood of surviving for 100 years absent anthropogenic impacts (Bjorkstedt et al. 2005). The remaining populations were dependent upon immigration from nearby CCC steelhead DPS populations to ensure their viability (Bjorkstedt et al. 2005; McElhany et al. 2000a).

Abundance and Productivity: While historical and present data on abundance are limited, CCC steelhead numbers are substantially reduced from historical levels. A total of 94,000 adult steelhead were estimated to spawn in the rivers of this DPS in the mid-1960s, including 50,000 fish in the Russian River – the largest population within the DPS (Busby et al. 1996). Near the end of the 20th Century, McEwan (2001) estimated the wild run population in the Russian River Watershed was between 1,700 and 7,000 fish. Abundance estimates for smaller coastal streams in the DPS indicate low but stable levels with recent estimates for several streams (Lagunitas, Waddell, Scott, San Vicente, Soquel, and Aptos creeks) of individual run sizes of 500 fish or less (62 FR 43937). CCC steelhead have experienced a serious decline in abundance and long-term population trends suggest a negative growth rate. This indicates the DPS may not be viable in the long term. Data from the 2008/09 and 2009/2010 adult CCC steelhead returns indicate a decline in returning adults across their range compared to other recent returns (e.g., 2006/2007, 2007/2008) (Jeffrey Jahn, NMFS, personal communication, August 2011). For more detailed information on trends in CCC steelhead abundance, see Busby et al. (1996c), Good et al. (2005a), Spence et al. (2008), and Williams et al. (2011).

Both adult and juvenile abundance data is limited for this DPS. While we currently lack data on naturally-produced juvenile CCC steelhead, it is possible to make rough estimates of juvenile abundance from the available adult return data. Juvenile CCC steelhead abundance estimates come from the escapement data. All returnees to the hatcheries do not contribute to the natural population and are not used in this calculation. For the species, fecundity estimates range from 3,500 to 12,000; and the male to female ratio averages 1:1 (Pauley et al. 1986a). By applying a conservative fecundity estimate of 3,500 eggs to the expected escapement of females (half of the escapement of natural-origin spawners – 715 females), 2.5 million eggs are expected to be produced annually. With an estimated survival rate of 6.5 percent (Ward and Slaney 1993), the DPS should produce roughly 162,549 natural outmigrants annually (Table 54).

Table 54. Geometric mean abundances of CCC steelhead spawners from 2006-2012 escapements by population.

Stratum	Waterbody	Years	Abundance		Expected Number of Outmigrants[ab]
			Natural	Hatchery	
Northern Coastal	Austin Creek[c]	2010-2012	63	-	7,166
	Lagunitas Creek[d]	2009-2013	71	-	8,076
	Walker Creek[e]	2007-2010	29	-	3,299
Interior	Dry Creek[c]	2011-2012	33	-	3,754
	Russian River[f]	2008-2012	230	3,451	26,163
Santa Cruz Mountains	Aptos Creek[g]	2007-2011	249	-	28,324
	San Gregario Creek[h]	2006-2007	23	-	2,616
	San Lorenzo Creek[g]	2007-2011	310	319	35,263
	Scott Creek[i]	-	179	96	20,361
	Soquel Creek[g]	2007-2011	230	-	26,163
Central Coastal	Napa River[j]	2009-2012	12	-	1,365
Total			1,429	3,866	162,549

[a] Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant

[b] Based upon natural-origin spawner numbers

[c] Source: http://www.scwa.ca.gov/fisheries-monitoring/

[d] (Ettlinger et al. 2012; Jankovitz 2013)

[e] Source: http://marinwater.org/documents/1_WalkerCreekReportandRefs_March2010.pdf

[f] Natural abundance: (Manning and Martini-Lamb 2012);

Hatchery abundance source: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=44269&inline=true

[g] Source: http://scceh.com/LinkClick.aspx?fileticket=dRW_AUu1EoU%3D&tabid=1772

[h] (Atkinson 2010)

[i] **Williams et al. (2011)**
[j] **(Koehler and Blank 2012)**

Limiting factors: Some loss of genetic diversity has been documented and attributed to previous among-basin transfers of stock and local hatchery production in interior populations in the Russian River (Bjorkstedt et al. 2005). Reduced population sizes and fragmentation of habitat in San Francisco streams has likely also led to loss of genetic diversity in these populations. DPS populations that historically provided enough steelhead immigrants to support dependent populations may no longer be able to do so, placing dependent populations at increased risk of extirpation. However, because CCC steelhead have maintained a wide distribution throughout the DPS, roughly approximating the known historical distribution, CCC steelhead likely possess a resilience that could slow their decline relative to other salmonid DPSs or ESUs in worse condition. The most recent status update concludes that steelhead in the CCC steelhead DPS remain "likely to become endangered in the foreseeable future" Williams et al. (2011), as new and additional information available since the previous status review (Good et al. 2005a) does not appear to suggest a change in extinction risk. On August 15, 2011, NMFS chose to maintain the threatened status of the CCC steelhead DPS (76 FR 50447).

4.2.17.8.9 California Central Valley Steelhead

The NMFS originally listed California Central Valley steelhead as threatened in 1998; this status was reviewed and retained on January 5, 2006 (71 FR 834).

Spatial Structure and Diversity: On March 19, 1998, NMFS listed California Central Valley (CCV) steelhead—both natural and some artificially-propagated fish—as a threatened species (63 FR 13347). On January 5, 2006, NMFS reaffirmed the threatened status of the CCV steelhead and applied the DPS policy to the species because the resident and anadromous life forms of *O. mykiss* remain "markedly separated" as a consequence of physical, ecological and behavioral factors, and therefore warranted delineation as a separate DPS and promulgated 4(d) protective regulations for CCV steelhead (71 FR 834). The section 4(d) protections (and limits on them) apply to natural and hatchery CCV steelhead with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed. On August 15, 2011, NMFS completed another 5-year status review of CCV steelhead and recommended that the CCV steelhead DPS remain classified as a threatened species (NMFS 2011l).

The CCV steelhead DPS includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Two artificial propagation programs were listed as part of the DPS—Coleman National Fish Hatchery and Feather River Hatchery winter-run steelhead hatchery stocks (Table 55).

Table 55. Expected annual CCV steelhead hatchery releases (California Hatchery Scientific Review Group 2012).

Artificial Propagation Program	Clipped Adipose Fin
Nimbus Hatchery (American River)	439,490
Feather River Hatchery (Feather River)	273,398
Coleman NFH (Battle Creek)	715,712
Mokelumne River Hatchery (Mokelumne River)	172,053
Total Annual Release Number	1,600,653

CCV steelhead historically were well-distributed throughout the Sacramento and San Joaquin rivers (Busby et al. 1996c). Although it appears CCV steelhead remain widely distributed in Sacramento River tributaries, the vast majority of historical spawning areas are currently above impassable dams. At present, all CCV steelhead are considered winter-run steelhead (McEwan and Jackson 1996), although there are indications that summer steelhead were present in the Sacramento River system prior to the commencement of large-scale dam construction in the 1940s (IEPSPWT 1999). McEwan and Jackson (1996) reported that wild steelhead stocks appeared to be mostly confined to upper Sacramento River tributaries such as Antelope, Deer, and Mill creeks and the Yuba River. However, naturally spawning populations are also known to occur in Butte Creek, and the upper Sacramento mainstem, Feather, American, Mokelumne, and Stanislaus rivers (CALFED 2000). Incidental catches and observations of steelhead juveniles also have occurred on the Tuolumne and Merced Rivers during fall-run Chinook salmon monitoring activities, indicating that steelhead are widespread, if not abundant, throughout accessible streams and rivers in the Central Valley (Good et al. 2005a).

Abundance and Productivity: Steelhead counts at the Red Bluff Diversion Dam (RBDD) have declined from an average annual count of 11,187 adults for the ten-year period beginning in 1967, to an average annual count 2,202 adults in the 1990s (McEwan and Jackson 1996). Estimates of the adult steelhead population composition in the Sacramento River (natural origin versus hatchery origin) have also changed over this time period; through most of the 1950s, Hallock et al. (1961) estimated that 88 percent of returning adults were of natural origin, and this estimate declined to 10 to 30 percent in the 1990s (McEwan and Jackson 1996). Furthermore, the California Fish and Wildlife Plan estimated a total run size of about 40,000 adults for the entire Central Valley, including San Francisco Bay, in the early 1960s (CDFG 1965). In 1991 and 1992, this run was probably less than 10,000 fish based on dam counts, hatchery returns and past spawning surveys (McEwan and Jackson 1996).

Both adult and juvenile abundance data is limited for this DPS. While we currently lack data on naturally-produced juvenile CCV steelhead, it is possible to make rough estimates of juvenile abundance from the available adult return data. Juvenile CCV steelhead abundance estimates come from the escapement data (Table 56). All returnees to the hatcheries do not contribute to the natural population and are not used in this calculation. For the species, fecundity estimates range from 3,500 to 12,000; and the male to female ratio averages 1:1 (Pauley et al. 1986a). By applying a conservative fecundity estimate of 3,500 eggs to the expected escapement of females (half of the escapement of natural-origin spawners – 687 females), 2.4 million eggs are expected to be produced annually. With an estimated survival rate of 6.5 percent (Ward and Slaney 1993), the DPS should produce roughly 156,293 natural outmigrants annually. In addition, hatchery managers could produce approximately 1.6 million ESA-listed hatchery juvenile CCV steelhead each year (Table 55).

Table 56. Abundance geometric means (2002-2010) for adult CCV steelhead natural- and hatchery-origin spawners.

Population	Years	Natural-origin Spawners	Hatchery-origin Spawners[a]	Expected Number of Outmigrants[bc]
American River[d]	2002-2005	308	1,326	35,035
Antelope Creek[e]	2007	140	-	15,925
Battle Creek[e]	2006-2010	377	1,396	42,884
Bear Creek[f]	2008-2009	119	-	13,536
Cottonwood Creek[f]	2008-2009	27	-	3,071
Clear Creek[e]	2005-2009	276	-	31,395
Cow Creek[f]	2008-2009	2	-	228
Feather River	2006-2010	-	504	-
Mill Creek[e]	-	15	-	1,706
Mokelumne River[e]	2006-2010	110	133	12,513
Total		1,374	3,359	156,293

a All hatchery-origin spawner data (2006-2010) from (California Hatchery Scientific Review Group 2012)

b Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant

c Based upon number of natural-origin spawners

d (Hannon and Deason 2005)

e (California Hatchery Scientific Review Group 2012)

f Teubert et al. (2011)

Limiting Factors: The status of CCV steelhead appears to have worsened since the 2005 status review (Good et al. 2005a), when the BRT concluded that the DPS was in danger of extinction. New information available since Good et al. (2005a) indicates an increased extinction risk (Williams et al. 2011). Steelhead have been extirpated from most of their historical range in this region. Habitat concerns in this DPS focus on the widespread degradation, destruction, and blockage of freshwater habitat within the region, and water allocation problems. Widespread hatchery production of introduced steelhead within this DPS also raises concerns about the potential ecological interactions between introduced and native stocks. Because the CCV steelhead population has been fragmented into smaller isolated tributaries without any large source population, and the remaining habitat continues to be degraded by water diversions, the population remains at an elevated risk for future population declines.

4.2.17.8.10 South-Central California Coast Steelhead

South-Central California Coast steelhead were listed as threatened in 1997, when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834).

Spatial Structure and Diversity: On August 18, 1997, NMFS listed South-Central California Coast (SCCC) steelhead—only natural-origin fish—as a threatened species (62 FR 43937). The BRT evaluated the viability and extinction risk of naturally spawning populations and found high risks to abundance, productivity, and the diversity of the SCCC DPS and expressed particular concern for this DPS's connectivity and spatial structure. NMFS promulgated 4(d) protective regulations for SCCC steelhead on January 5, 2006 (71 FR 834). The section 4(d) protections (and limits on them) apply to natural and hatchery SCCC steelhead with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed.

SCCC steelhead occupy rivers from the Pajaro River (Santa Cruz County, California), inclusive, south to, but not including, the Santa Maria River (San Luis Obispo County, California). Most rivers in this DPS drain from the San Lucia Mountain range, the southernmost section of the California Coast Ranges. Many stream and rive mouths in this area are seasonally closed by sand berms that form during the low water flows of summer.

Abundance and Productivity: In the mid-1960s the California Department of Fish and Wildlife estimated an annual run size of 17,750 adult steelhead spawning in this coastal DPS. The CDFG estimate, however, is just a midpoint number in the SCCC steelhead's abundance decline—at the point the estimate was made, there had already been a century of commercial harvest and coastal development. Current SCCC steelhead abundance is still not well known. Multiple short-term studies using different methodologies have occurred over the past decade.

Table 57. Geometric mean abundances of SCCC steelhead spawners from 2001-2012 escapements by population.

Stratum	Waterbody	Years	Abundance	Expected Number of Outmigrants[a]
Interior Coast Range	Pajaro River[b]	2007-2011	35	3,981
	Salinas River[c]	2011-2013	21	2,389
Carmel River Basin	Carmel River[d]	2009-2013	318	36,173
Big Sur Coast	Big Sur River[e]	2010	11	1,251
	Garrapata Creek[f]	2005	17	1,934
San Luis Obispo Terrace	Arroyo Grande Creek[g]	2006	18	2,048
	Chorro Creek[h]	2001	2	228
	Coon Creek[i]	2006	3	341
	Los Osos Creek[h]	2001	23	2,616
	San Simeon Creek[j]	2005	4	455
	Santa Rosa Creek[k]	2002-2006	243	27,641
Total			695	79,057

[a] Expected number of outmigrants=Total spawners*50% proportion of females*3,500 eggs per female*6.5% survival rate from egg to outmigrant

[b] Source: http://scceh.com/LinkClick.aspx?fileticket=dRW_AUu1EoU%3D&tabid=1772

[c] Kraft et al. 2013

[d] Sources: http://www.mpwmd.dst.ca.us/fishcounter/fishcounter.htm and http://www.mpwmd.dst.ca.us/wrd/lospadres/lospadres.htm.

[e] (Allen and Riley 2012)

[f] Garrapata Creek Watershed Council 2006

[g] Source: http://www.coastalrcd.org/zone1-1a/Fisheries%20Studies/AG_Steelhead_Report_Draft-small.pdf

[h] Source: http://www.coastalrcd.org/images/cms/files/MB%20Steelhead%20Abund%20and%20Dist%20Report.pdf

[i] City of San Luis Obispo 2006

[j] Baglivio (2012)

[k] Stillwater Sciences et al. (2012)

Both adult and juvenile abundance data is limited for this DPS. While we currently lack data on naturally-produced juvenile SCCC steelhead, it is possible to make rough estimates of juvenile abundance from the available adult return data. The estimated average adult run size is 695 (Table 57). Juvenile SCCC steelhead abundance estimates come from the escapement data. For the species, fecundity estimates range from 3,500 to 12,000; and the male to female ratio averages 1:1 (Pauley et al. 1986a). By applying a conservative fecundity estimate of 3,500 eggs to the expected escapement of females (half of the escapement of spawners – 348 females), 1.2

million eggs are expected to be produced annually. With an estimated survival rate of 6.5 percent (Ward and Slaney 1993), the DPS should produce roughly 79,057 natural outmigrants annually.

<u>Limiting Factors</u>: NMFS' most recent status reviews for SCCC steelhead (NMFS 2011l; Williams et al. 2011) identified habitat destruction and degradation as serious ongoing risk factors for this DPS. Urban development, flood control, water development, and other anthropogenic factors have adversely affected the proper functioning and condition of some spawning, rearing, and migratory habitats in streams designated as critical habitat. Urbanization has resulted in some permanent impacts to steelhead critical habitat due to stream channelization, increased bank erosion, riparian damage, migration barriers, and pollution (Good et al. 2005a). Many streams within the DPS have dams and reservoirs that mute flushing stream flows, withhold or reduce water levels suitable for fish passage and rearing, physically block upstream fish passage, and retain valuable coarse sediments for spawning and rearing. In addition, some stream reaches within the DPS' designated critical habitat may be vulnerable to further perturbation resulting from poor land use and management decisions.

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 and conference report.

Table 58. Summary of Species Effect Determinations for Training and Testing Activities in the Action Area

Species/DPS/ESU		Species Effect Determinations		
		Training (Navy)	**Testing (Navy)**	**NMFS Final Determination**
MARINE MAMMALS				
North Pacific right whale		NLAA	NLAA	NLAA
Humpback whale		LAA	LAA	**LAA**
Blue whale		LAA	LAA	**LAA**
Fin whale		LAA	LAA	**LAA**
Sei whale		NLAA	LAA	**LAA**
Western North Pacific gray whale		NLAA	NLAA	NLAA
Sperm whale		LAA	LAA	**LAA**
Southern Resident killer whale		LAA	NLAA	**LAA**
Guadalupe fur seal		LAA	LAA	**LAA**
SEA TURTLES				
Green sea turtle		NLAA	NLAA	NLAA
Loggerhead sea turtle		NLAA	NLAA	NLAA
Olive Ridley sea turtle		NLAA	NLAA	NLAA
Leatherback sea turtle		LAA	LAA	**LAA**
FISH				
Chinook Salmon	Puget Sound ESU	LAA	NLAA	**LAA**
	Upper Columbia River spring-run ESU	LAA	LAA	**LAA**
	Lower Columbia River ESU	LAA	LAA	**LAA**
	UpperWillamette River ESU	LAA	LAA	**LAA**
	Snake River spring/summer-run ESU	LAA	LAA	**LAA**
	Snake River fall-run ESU	LAA	LAA	**LAA**
	California Coastal ESU	LAA	LAA	**LAA**
	Central Valley spring-run ESU	LAA	LAA	**LAA**
	Sacramento River winter-run	LAA	LAA	**LAA**
Coho Salmon	Lower Columbia ESU	LAA	LAA	**LAA**
	Oregon coast ESU	LAA	LAA	**LAA**
	Southern Oregon/Northern California coast ESU	LAA	LAA	**LAA**
	Central California Coast	LAA	LAA	**LAA**
Chum Salmon	Hood Canal summer-run ESU	LAA	NLAA	**LAA**
	Columbia River ESU	LAA	LAA	**LAA**

Species/DPS/ESU		**Species Effect Determinations**		
		Training (Navy)	**Testing (Navy)**	**NMFS Final Determination**
Sockeye Salmon	Ozette LakeESU	NLAA	NLAA	**NLAA**
	Snake River ESU	LAA	LAA	**LAA**
Steelhead	Puget Sound DPS	LAA	NLAA	**LAA**
	Upper Columbia River DPS	LAA	LAA	**LAA**
	Middle Columbia River DPS	LAA	LAA	**LAA**
	Lower Columbia River DPS	LAA	LAA	**LAA**
	Upper Willamette River DPS	LAA	LAA	**LAA**
	Snake River Basin DPS	LAA	LAA	**LAA**
	Northern California DPS	NLAA	NLAA	**LAA**
	California Central Valley DPS	NLAA	NLAA	**LAA**
	Central California Coast DPS	NLAA	NLAA	**LAA**
	South-Central California Coast DPS	NLAA	NLAA	**LAA**
	Southern California DPS	NLAA	NLAA	**NLAA**
Rockfish	Bocaccio–Puget Sound/Georgia Basin DPS	LAA	NLAA	**LAA**
	Canary–Puget Sound/Georgia Basin DPS	LAA	NLAA	**LAA**
	Yelloweye –Puget Sound/Georgia Basin DPS	LAA	NLAA	**LAA**
Pacific Eulachon–Southern DPS		LAA	NLAA	**LAA**
Green Sturgeon –Southern DPS		NLAA	NLAA	NLAA

Table 59. Summary of Critical Habitat Effect Determinations for Training and Testing Activities in the Action Area

Species/DPS/ESU		**Critical Habitat Effect Determinations**		
		Training (Navy)	**Testing (Navy)**	**NMFS Final Determination**
MARINE MAMMALS				
North Pacific right whale		NE	NE	NE
Humpback whale		--	--	--
Blue whale		--	--	--
Fin whale		--	--	--
Sei whale		--	--	--
Western North Pacific gray whale		--	--	--
Sperm whale		--	--	--
Southern Resident killer whale		NE	NE	NE
Guadalupe fur seal		--	--	--
SEA TURTLES				
Green sea turtle		NE	NE	NE
Loggerhead sea turtle		NE	NE	NE
Olive Ridley sea turtle		NE	NE	NE
Leatherback sea turtle		NE	NE	NE
FISH				
Chinook Salmon	Puget Sound ESU	NLAA	NLAA	NLAA
Chum Salmon	Hood Canal summer-run ESU	NLAA	NLAA	NLAA
Steelhead	Puget Sound DPS[1]	NE	NE	NE
Rockfish	Bocaccio–Puget Sound/Georgia Basin DPS	NE	NE	NLAA
	Canary–Puget Sound/Georgia Basin DPS	NE	NE	NLAA
	Yelloweye –Puget Sound/Georgia Basin DPS	NE	NE	NLAA
Green Sturgeon –Southern DPS		NLAA	NLAA	NLAA

1. If proposed critical habitat remains as freshwater and estuarine habitats only then the critical habitat determination is a no effect. However, if the final listing changes to include marine nearshore areas, then the conclusion will be a may affect, not likely to adversely affect similar to Puget Sound Chinook and Hood Canal summer-run chum.

LAA = may affect, Likely to Adversely Affect;

NLAA = may affect, Not Likely to Adversely Affect;

NE = No Effect (critical habitat designated [in or out of the Action Area])

-- = critical habitat not designated

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).

5.1 Climate Change

The latest Assessment Synthesis Report from the Working Groups on the Intergovernmental Panel on Climate Change (IPCC) concluded climate change is unequivocal (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 ^{o}C per decade over the period 1971 to 2010) (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-19th century has been greater than the mean rate during the previous two millennia (IPCC 2014). Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012a). Further, ocean acidity has increased by 26 percent since the beginning of the industrial era (IPCC 2014) 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. Notably, leatherback sea turtles were predicted to gain core habitat area, whereas loggerhead sea turtles and blue whales were predicted to experience losses in available core habitat. McMahon and Hays (2006) predicted increased ocean temperatures would expand the distribution of leatherback sea turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean. 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. Willis-Norton et al. (2015) acknowledge there would be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback sea turtles in the eastern south Pacific.

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).

Changes in global climatic patterns are expected to have profound effects on coastlines worldwide, potentially having significant consequences for the species considered in this Opinion that are partially dependent on terrestrial habitat areas (i.e., sea turtles). For example, rising sea levels are projected to inundate some sea turtle nesting beaches (Caut et al. 2009; Wilkinson and Souter 2008), change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson and Souter 2008). The loss of nesting beaches may have catastrophic effects on global sea turtle populations if they are unable to colonize new beaches, or if new beaches do not provide the habitat attributes (e.g., sand depth, temperature regimes, refuge) necessary for egg survival. Additionally, increasing temperatures in sea turtle nests, as is expected with climate change, alters sex ratios, reduces incubation times (producing smaller hatchlings), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2010; Fuentes et al. 2009c; Glen et al. 2003). All of these temperature related impacts have the potential to significantly impact sea turtle reproductive success and ultimately, long-term species viability. Poloczanska et al. (2009b) noted that extant marine turtle species have survived past climatic shifts, including glacial periods and warm events, and therefore may have the ability to adapt to ongoing climate change (e.g., by finding new nesting beaches). However, the authors also suggested since the current rate of warming is very rapid, expected changes may outpace sea turtles' ability to adapt.

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. 2009a). 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. 2009a). 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). 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. 2009a). 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 60).

Table 60. 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 most recent report by the IPCC has found that 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 (www.climatechange2013.org/images/uploads/WGIAR5-SPM_Approved27Sep2013.pdf).

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). For sea turtles, warming ocean temperatures may extend poleward the habitat which they can utilize (Poloczanska et al. 2009a). 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 sea turtle species (Poloczanska et al. 2009a). 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 2002b; 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 2002b; 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. An example of this is the altered sex ratios observed in sea turtle populations worldwide (Fuentes et al. 2009a; Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008). This does not appear to have yet affected population viabilities through reduced reproductive success, although nesting and emergence dates of days to weeks in some locations have changed over the past several decades (Poloczanska et al. 2009a). 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). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form or if the beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut et al. 2009). Storms may also cause direct harm to sea turtles, causing "mass" strandings and mortality (Poloczanska et al. 2009a). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchling), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2010; Fuentes et al. 2009c). Additionally, green sea turtle hatchling size also appears to be influenced by incubation temperatures, with smaller hatchlings produced at higher temperatures (Glen et al. 2003). An air temperature increase of 3° C is likely to exceed the thermal threshold of most clutches, leading to death (Hawkes et al. 2007). Smaller individuals likely experience increased predation (Fuentes et al. 2009b). Climatic anomalies influencing the Marianas Islands include El Niño/Southern Oscillation (ENSO) and La Niña events (Giese and Carton 1999; Mantua and Hare 2002a; NOAA 2005a; NOAA 2005b; Sugimoto et al. 2001; Trenberth 1997). Recent research egg and hatchling mortality of leatherback turtles in northwest Costa Rica were affected by climatic variability (precipitation and air temperature) driven by the El Niño Southern Oscillation (ENSO). Drier and warmer conditions associated with El Niño increased egg and hatchling mortality (Santidrián Tomillo et al. 2012). The fourth assessment report of the IPCC projects a warming and drying in Central America and other regions of the World (IPCC 2007). Using projections from an ensemble of global climate models contributed to the Intergovernmental Panel on Climate Change report, Santidrián et al. (2012) projected that egg and hatchling survival will rapidly decline in the region over the next 100 years, due to warming and drying in northwestern Costa Rica. Warming and drying trends may threaten the survival of sea turtles.

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. 1999a). Sei whales are estimated to have been reduced to 20 percent (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977b). In addition, 9,500 blue whales were reported killed by commercial whalers in the North Pacific between 1910 and 1965 (Ohsumi and Wada. 1972); 46,000 fin whales between 1947and 1987 (Rice 1984); and 25,800 sperm whales (Barlow et al. 1997). North Pacific right whales once numbered 11,000 animals but commercial whaling has now reduced their population to 29 to 100 animals (Wada 1973).

5.3 Fisheries

Listed salmon are incidentally caught in several fisheries that operate in the NWTT Action Area, including groundfish fisheries that operate off the coasts of Washington; fisheries for Pacific salmon that operate under the Pacific Salmon Treaty; salmon fisheries that are managed by the U.S. Pacific Fisheries Management Council under the Pacific Coast Management Plan; salmon fisheries managed by the U.S. Fraser River Panel; commercial ocean salmon troll fisheries that operate off the coasts of Oregon and Washington; and subsistence, commercial, and recreational fisheries for Pacific salmon that operate in the Columbia River. These fisheries incidentally capture endangered and threatened salmon. Endangered Species Act section 7 consultation standards aim to limit the impact of ocean salmon fisheries on ESA-listed stocks. For example, the maximum age-3 impact rate for 2015 ocean salmon fisheries on Sacramento River winter Chinook is 19 percent (PFMC 2015).

The whiting fishery, which is a component of the groundfish fisheries, were expected to incidentally capture not more than 11,000 Chinook salmon per year and have been estimated to have caught an average of 7,281 each year from 1991 to 2005 (NMFS 2006b). The bottom trawl component of the groundfish fishery was expected to capture between 6,000 and 9,000 Chinook salmon each year, with 5,000 to 8,000 of these salmon captured in the Vancouver and Columbia catch areas. On average, the bottom trawl groundfish fisheries captured 11,320 Chinook salmon, 40 coho salmon, and 13 chum salmon from 2002 to 2004 (NMFS 2006b).

Biological opinions that NMFS has issued on these fisheries concluded that the fisheries were not likely to jeopardize the continued existence of endangered or threatened salmon that were likely to be captured in the fisheries. Biological opinions on the effects of these fisheries on southern resident killer whales, which rely on salmon for food, concluded that fishery-related removals of salmon were not likely to jeopardize the continued existence of southern resident killer whales.

Entrapment and entanglement in commercial fishing gear is one of the most frequently documented sources of human-caused mortality in large whale species and sea turtles. For example, in 1978, Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherback turtles, loggerheads, olive ridleys and hawksbills, were captured annually by Japanese tuna longliners in the Western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data and questionnaires, Nishimura and Nakahigashi (1990) estimated that for every 10,000 hooks in the Western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42 percent.

NMFS has observed 3,251 sets, representing approximately 3,874,635 hooks (data from February 1994 through December 31, 1999). The observed entanglement rate for sperm whales would equal about 0.31 whales per 1,000 sets or 0.0002 per 1,000 hooks. At those rates, we would expect about 200 sperm whales entanglements per 1,000 sets. However, only one sperm whale has been entangled in this gear; as a result, NMFS believes that the estimated entanglement rate substantially overestimates a sperm whale's actual probability of becoming entangled in this gear and the potential hazards longline gear poses to sperm whales.

5.4 Vessel Strike

The NWTT Action Area is home to major shipping facilities in Puget Sound. Figure 7 displays one month of commercial ship automated identification system position data for the Washington portion of the Action Area.

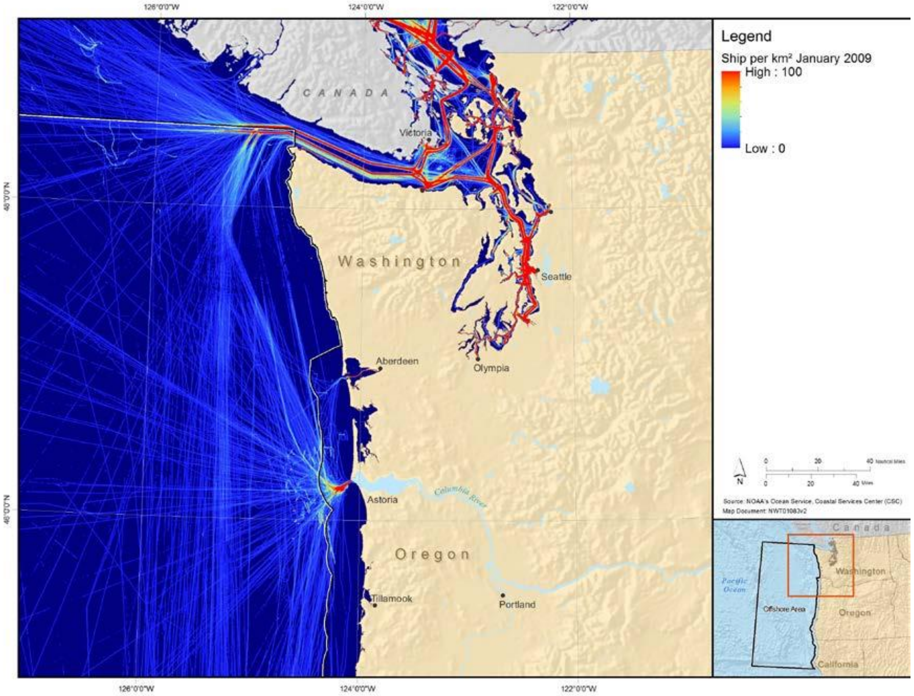

Figure 7. Commercial Ship Automated Identification System Position Data for Washington State (January 2009)

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. The number of observed physical injuries to humpback whales as a result of ship collisions has increased in Hawaiian waters (Glockner-Ferrari et al. 1987). On the Pacific coast, a humpback whale is probably killed about every other year by ship strikes (Barlow *et al.* 1997). From 1996 to 2002, eight humpback whales were reported struck by vessels in Alaskan waters. In 1996, a humpback whale calf was found stranded on Oahu with evidence of vessel collision (propeller cuts; NMFS unpublished data). From 1994 to 1998, two fin whales were presumed to have been killed in ship strikes.

Based on the data available from Douglas et al. (2008c), Jensen and Silber (2004a), and Laist et al. (2001), there have been at least 25 incidents in which marine mammals are known to have been struck by ships in the Puget Sound region and southwestern British Columbia. The marine mammals that were involved in almost half of these incidents died as a result of the strike and they suffered serious injuries in four of those strikes.

Virtually all of the rorqual whale species have been documented to have been hit by vessels. This includes blue whales (Berman-Kowalewski et al. 2010; Calambokidis 2012; Van Waerebeek et al. 2007), fin whales (as recently as November 2011 in San Diego) (Douglas et al. 2008a; Van Waerebeek et al. 2007), sei whales (Felix and Waerebeek 2005; Van Waerebeek et al. 2007), Bryde's whales (Felix and Waerebeek 2005; Van Waerebeek et al. 2007), minke whales (Van Waerebeek et al. 2007), and humpback whales (Douglas et al. 2008a; Lammers et al. 2003; Van Waerebeek et al. 2007). For example, in April 2013 (at Burien, Washington) and June 2013 (at Ocean City, Washington) two stranded fin whales that had been struck by vessels brought the total to nine known fin whales having stranded in Washington after being struck by vessels in approximately the last decade (Schorr et al. 2013).

Fin whales were struck most frequently, accounting for almost 30 percent of the total number of incidents and two-thirds of the incidents in which the whale died as a result of the collision. Northern resident killer whales were struck slightly less frequently, although a cluster of ship strikes in 2006 accounted for four of the six ship strikes involving this population of killer whales. Humpback whales were third in frequency, followed by southern resident killer whales, offshore killer whales, and blue whales. About two-thirds (17 out of the 25) of the incidents occurred in waters off British Columbia, although the locations were variable.

The adult male southern resident killer whale (L98) that was killed in a collision with a tug boat in 2006 may have reduced the demographic health of this killer whale population. At population sizes between 75 and 90 individuals, we would expect southern resident killer whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox 2007)—including stochastic sex determination (Lande et al. 2003)—and the effects of phenomena interacting with environmental variability. Although the small number of adult male southern resident killer whales might represent a stable condition for this species, it might also reflect the effects of stochastic sex determination. If the latter is the case, the death of L98 in a population with a smaller percentage of males would increase the imbalance of male-to-female gender ratios in this population and increase the population's probability of further declines in the future.

Allen et al. (2012) recorded the noises from 24 ships ranging in length from 10.4 m to 294.1 m at hydrophone depths of 5, 15, and 25 m and calculated source levels to characterize the three-dimensional acoustic environment a mysticete would encounter during a whale/ship approach. Results indicated that mysticetes near the sea surface may experience greater difficulty localizing oncoming ships than in deeper waters as a combined result of lower SLs at the surface in shallow locations, bow null effect acoustic shadow zones, and masking from ambient noise. As a consequence, the range of detection for a ship may be too close for a mysticete to execute a successful avoidance maneuver.

5.5 Water Quality Degradation

Between 2000 and 2006, counties in Puget Sound increased by 315,965 people or by more than 50,000 people per year, with associated increases in the area of impervious surface and population density per square mile of impervious surface in the Puget Sound region (PSAT 2007). Between 1991 and 2001, the area of impervious surface in the Puget Sound basin increased 10.4 percent (PSAT 2007). By 2001, impervious surface covered 7.3 percent of the Puget Sound region below 1,000 feet elevation; in some counties and watersheds in the region, this area was substantially higher.

Over the same time interval, about 190 square miles of forest (about 2.3 percent of the total forested area of the Puget Sound basin) was converted to other uses. In areas below 1,000 feet elevation, the change was more dramatic: 3.9 percent of total forest area was converted to other uses. By 2004, about 1,474 fresh and marine waters in Puget Sound were listed as "impaired waters" in Puget Sound. Fifty-nine percent of these waters tested were impaired because of toxic contamination, pathogens, low dissolved oxygen or high temperatures. Less than one-third of these impaired waters have cleanup plans in place. Chinook salmon from Puget Sound have 2-to-6 times the concentrations of PCBs in their bodies as other Chinook salmon populations on the Pacific Coast. Because of this contamination, the Washington State Department of Health issued consumption advisories for Puget Sound Chinook (PSAT 2007).

The quality of water in the Puget Sound Basin and aquatic biota those water support have been affected by a range of forestry, agricultural, and urban development practices. The chemical quality of surface water in the foothills and mountains is generally suitable for most uses. However, the physical hydrology, water temperature, and biologic integrity of streams have been influenced to varying degrees by logging (Ebbert et al. 2000).

Because of development, many streams in the Puget Lowlands have undergone changes in structure and function with a trend toward simplification of stream channels and loss of habitat (Ebbert et al. 2000). Sources of contaminants to lowland streams and lower reaches of large rivers are largely nonpoint because most major point sources discharge directly to Puget Sound. Compared with that in small streams in the Puget Lowlands, the quality of water in the lower reaches of large rivers is better because much of the flow is derived from the forested headwaters.

More than half of the agricultural acreage in the basin is located in Whatcom, Skagit, and Snohomish Counties. Agricultural land use consists of about 60 percent cropland and 40 percent pasture. Livestock produce a large amount of manure that is applied as fertilizer to cropland, some- times in excess amounts, resulting in runoff of nitrogen and phosphorus to surface water and leaching of nitrate to ground water. Runoff from agricultural areas also carries sediment, pesticides, and bacteria to streams (Ebbert et al. 2000). Pesticides and fumigant-related compounds are present, usually at low concentrations, in shallow ground water in agricultural areas.

Heavy industry is generally located on the shores of the urban bays and along the lower reaches of their influent tributaries, such as Commencement Bay and the Puyallup River in Tacoma and Elliott Bay and the Duwamish Waterway in Seattle. High-density commercial and residential development occurs primarily within and adjacent to the major cities. Development in recent years has continued around the periphery of these urban areas but has trended toward lower density. This trend has resulted in increasing urban sprawl in the central Puget Sound Basin.

Urban land-use activities have significantly reduced the quality of streams in the Puget Sound Basin (Ebbert et al. 2000). Water-quality concerns related to urban development include providing adequate sewage treatment and disposal, transport of contaminants to streams by storm runoff, and preservation of stream corridors.

Water availability has been and will continue to be a major, long- term issue in the Puget Sound Basin. It is now widely recognized that ground-water withdrawals can deplete streamflows (Ebbert et al. 2000), and one of the increasing demands for surface water is the need to maintain instream flows for fish and other aquatic biota.

Pollutants founds in Puget Sound Chinook salmon have found their way into the food chain of the Sound. Harbor seals in southern Puget Sound, which feed on Chinook salmon, have PCB levels that are seven times greater than those found in harbor seals from the Georgia Basin. Concentrations of polybrominated diphenyl ether (also known as PBDE, a product of flame retardants that are used in household products like fabrics, furniture, and electronics) in seals have increased from less than 50 parts per billion in fatty tissue to more than 1,000 ppb over the past 20 years (PSAT 2007).

Water quality appears poised to have larger-scale effects on the marine ecosystem of the Puget Sound – Georgia Basin as evidenced by the intensity and persistence of water stratification in the basin. Historically, Puget Sound was thought to have an unlimited ability to assimilate waste from cities, farms and industries in the region and decisions about human occupation of the landscape were based on that belief. More recent data suggests that the marine ecosystems of the basin have a much more limited ability to assimilate pollution, particularly in areas such as Hood Canal, south Puget Sound, inner Whidbey basin and the central Georgia Basin. In these areas, as strong stratification has developed and persisted, the respective water quality has steadily decreased. As waters become more stratified, through weather, climate or circulation changes, they become even more limited in their ability to assimilate pollution.

The presence of high levels of persistent organic pollutants, such as PCBs, DDT, and flame–retardants have also been documented in southern resident killer whales (Herman et al. 2005;

Ross et al. 2000; Ylitalo et al. 2001). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Kakuschke and Prange. 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007). Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

5.6 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, dredging, construction; oil, gas, and mineral exploration in offshore areas; 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).

Andrew et al. (2002) compared ocean ambient sound from the 1960s to the 1990s from a receiver off the California coast. The data showed an increase in ambient noise of approximately 10 dB in the frequency ranges of 20 to 80 Hz and 200 to 300 Hz, and about 3 dB at 100 Hz over a 33-year period. Each 3 dB increase is noticeable to the human ear as a doubling in sound level. A possible explanation for the rise in ambient noise is the increase in shipping noise. There are approximately 11,000 supertankers worldwide, each operating approximately 300 days per year, each producing constant broadband noise at typical source levels of 198 dB (Hildebrand 2004). Generally the most energetic regularly operated sound sources are seismic airgun arrays from approximately 90 vessels with typically 12 to 48 individual guns per array, firing about every 10 seconds (Hildebrand 2004).

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. The oil and gas industry conduct seismic surveys to search for new hydrocarbon deposits. In addition, research geologists conduct seismic surveys to study plate tectonics as well as other topics in marine geology. 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.

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, dredging, construction; oil, gas, and mineral exploration in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities (Richardson et al. 1995d).

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 2003b; 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 2003b). Commercial fishing vessels, cruise ships, transport boats, airplanes, helicopters and recreational boats all contribute sound into the ocean (NRC 2003b). The military uses sound to test the systems of Navy vessels as well as for naval operations. 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 2003b).

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@1m, 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 1000 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). As noted previously, 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).

Bassett et al. (2012b) recorded vessel traffic over a period of just under a year as large vessels passed within 20 km of a hydrophone site located at Admiralty Inlet in Puget Sound. During this period there were 1,363 unique Automatic Identification System transmitting vessels recorded. In 2014, there were over 5,300 cargo, cruise, or fishing vessels docking at one of the major ports in Puget Sound. In addition to these port calls resulting in approximately 10,600 annual vessel transits, there is the routine ferry, recreational, and other vessel traffic from commercial activities such as whale watching in the Inland Waters portion of the NWTT Study Area. Because Navy vessels are much fewer in number, they are a small component of overall vessel traffic and vessel noise in most areas where they operate; this is especially the case in the Study Area (see (Mintz and Filadelfo 2011) concerning a general summary for the U.S. EEZ).

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).

Erbé (2002b) recorded underwater noise of whale-watching boats in the popular killer whale-watching region of southern British Columbia and northwestern Washington State. Source levels ranged from 145 to 169 dB re 1 Pa - 1 m and increased as the vessel's speed increased. Based on sound propagation models, she concluded that the noise of fast boats would be audible to killer whales over 16 km, would mask killer whale calls over 14 km, would elicit behavioral response over 200 m, and would cause a temporary threshold shifts of 5 dB within 450 m after 30 to 50 minutes of exposure. She concluded that boats cruising at slow speeds would be audible and would cause masking at 1 km, would elicit behavioral responses at 50 m, and would result in temporary threshold shifts at 20 m.

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).

The engines associated with the boats in their study produced sounds in the 0.5 to 8.0 KHz range at source levels comparable to those of killer whale vocalizations. They concluded that those boats in their study that travelled at their highest speeds proximate to killer whales could make enough noise to make hearing difficult for the whales.

In addition to the disturbance associated with the presence of vessel, the vessel traffic affects the acoustic ecology of southern resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of southern resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15 percent during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2009a) reported that southern resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales needs to increase the duration of their vocalization to avoid masking by the boat noise.

Commercial shipping traffic is a major source of low frequency (5 to 500 Hz) human generated sound in the world's oceans (NRC 2003b; 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. Within the Action Area identified in this Opinion, the vessel sound inside the western half of the Strait of Juan de Fuca and off the Washington coast comes from cargo ships (86 percent), tankers (6 percent), and tugs (5 percent) (NMFS 2008d citing Mintz and Filadelfo 2004a, 2004b)).

(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. (2012a) 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 (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.

Measurements taken at San Nicolas Island identified seasonal differences in ocean ambient levels due to seasonal changes in wind driven waves, biological sound production, and shipping route changes (McDonald et al. 2006a). The strongest seasonal signal at the San Nicolas South site was attributed to blue whale singing (Burtenshaw et al. 2004a) which had a broad peak near 20 Hz in the spectral data (because fin whales occur in the area throughout the year, the seasonal difference was attributed to blue whales, which only occur in the areas seasonally). When the band of fin whale calls were excluded, the average February 2004 ambient pressure spectrum level was 10 to 14 dB higher than the February 1965 and 1966 levels over the 10 to 50 Hz band. Above 100 Hz, there was a 1 to 2 dB difference between the two sets of February noise data (McDonald et al. 2006a). Afterward, the increase per year was significantly less, about 0.2dB/yr.

5.7 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 recent study of whale watch activities worldwide has 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 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 then, 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). 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. In January 2002, NMFS also published an official policy on human interactions with wild marine mammals which states that: "*NOAA Fisheries cannot support, condone, approve or authorize activities that involve closely approaching, interacting or attempting to interact with whales, dolphins, porpoises, seals or sea lions in the wild. This includes attempting to swim with, pet, touch or elicit a reaction from the animals.*"

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.

The number and proximity of vessels, particularly whale-watch vessels in the areas occupied by southern resident killer whales, represents a source of chronic disturbance for this population. 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.

5.8 Navy Training and Testing Activities in the Action Area

The Navy has been conducting exercises in the NWTT Action Area for over 60 years. In terms of surface combatant ships, currently there are two aircraft carriers and five Navy destroyers home-ported at naval facilities within Puget Sound. Monitoring in conjunction with Navy exercises to determine the effects of active sonar and explosives on marine mammals was initiated in 2010 as part of the MMPA regulations that allowed NMFS to issue LOAs for Navy military readiness activities in the NWTRC. Stranding data has been collected by researchers in the NWTRC for approximately 30 years as well as by NMFS for roughly 22 years. Though not all dead or injured marine mammals can be accounted for, if marine mammals were being harmed by the Navy training exercises in the NWTRC with any regularity, evidence of that harm would have been detected over the 30-year period.

5.8.1 Summary of Activities Under Previous LOAs and Biological Opinions

Here we provide information regarding the Navy's activities under the 2010 to 2011, 2011 to 2012 and the 2012 to 2015 LOAs and biological opinions. This information was provided by the Navy in annual exercise reports, monitoring reports, and the ESA consultation request package. We also reviewed the classified reports to verify compliance with previous ESA biological opinions and Incidental Take Statements.

5.8.1.1 *Activities Conducted Through May 2014*

In the annual exercise reports, the Navy reported that no individual category of authorized system(s) or training event(s) exceeded authorized quantities (to include sonar and explosives) within the NWTRC (Navy 2011; Navy 2012a). The Navy did not provide the number of exercises, expended/detonated rounds, or the number of IEER events in the unclassified reports (Navy 2011; Navy 2012a). Neither individual testing activities nor activities under the Keyport LOA or Opinion are reported herein, but are also below authorized take quantities.

5.8.1.2 *Activities Not Conducted*

The Navy did not conduct the following activities under the 2010 to 2011, 2011 to 2012, or to date per the 2012 to 2015 LOAs and biological opinions:

- Sinking exercises (SINKEX)
- Installation of the Portable Undersea Tracking Range (PUTR);
- Installation of the underwater training minefield (non-explosive).

While no EOD underwater detonations (UNDETs) were undertaken through 2012, there were a few, low level EOD UNDETs in 2013 to 2014 (source: Navy 2013 and 2014 annual exercise reports). The 2013 NWTRC Annual Exercise Report included one UNDET (November 15, 2012, one 1.5 lbs NEW charge). The 2014 NWTRC Annual Exercise Report included seven UNDETs (all reduced charge weight events, <1 ounce each) [August 28, 2013; three 1 oz NEW charges; and April 2, 2014, four 1 oz NEW charges]. A total of three EOD UNDET events with eight underwater explosions occurred within NWTRC from November 2013 through April 2014. The maximum NEW was 1.5 lb. Most (88 percent) were 1-oz charges.

5.8.1.3 *Estimated Take of ESA-listed Species Through May 2014*

The Navy's classified annual NWTRC Exercise Report contains the list of authorized systems used for the reporting year through May 2014. Of the sonar systems authorized under NMFS Final Rule and LOA for the NWTRC, one system reported annual use (May 2012 to May 2013) higher than the average annual use authorized in the LOA and incidental take statement of the

2012 to 2015 biological opinion. This occurred over a one-week period in September 2012 during an additional unscheduled training event in the offshore waters of the NWTRC, and accounted for over 116 percent of the annual authorization for this system as reported in the classified NWTRC Exercise Report. However, in terms of the five-year authorization from the NWTRC Final MMPA Rule, the system in question is still only at 38 percent of total authorized use after three of five years. Accordingly, for purposes of the now-superseded 2010 and 2012 biological opinions, NMFS has determined that this estimated annual exceedance would not result in a change to those earlier conclusions that the Navy training activities in the NWTRC are not likely to jeopardize listed species or likely to destroy or adversely modify critical habitat. NMFS will consider the estimated take from these activities as well as those identified in the Navy's May 2014 annual report, as part of the environmental baseline for this consultation.

Table 61 contains annual NWTRC estimated post-calculation annual potential exposures through May 2014 as well as cumulative species specific estimates for the entire period through May 2014. Post-calculation comparisons may not be indicative of actual exposures based on real-world short and long-term spatial movements of various species and their relative occurrence within the NWTRC. Additionally, post-calculated takes are relative to the model-predicted values derived for the 2010 programmatic biological opinion and MMPA rule/LOA.

During the 2012 to 2013 reporting period, based on calculations using the earlier model, humpback whale takes exceeded the 15 annual take estimates (19 post-calculated) by four exposures potentially resulting in behavioral harassment. Of the 19 exposures, 19 were from active sonar and zero from explosive sources. Fin whale estimated takes (57) were also exceeded by four exposures potentially resulting in behavioral harassment. Of the 61 exposures to fin whales, 60 were from sonar and 1 from explosive sources. Sperm whale takes were exceeded by19 exposures (130 post-calculation vs. 111 annual authorized). Of The 130 exposures, 129 were from sonar and 1 from explosive sources. All post-calculated explosive exposures were from 76-mm GUNEX only.

Due to difficulty in determining particular stock densities of killer whales, all stocks of killer whales were combined for NWTRC modeling exposures. This included offshore, transient, and southern resident killer whale stocks. There was no Navy modeled exposure to killer whales from explosives. Most (86 percent) of modeled exposures to killer whales in general were from surface ship sonar (12 Level B/behavioral harassment exposures) and 14 percent from DICAS sonobuoys (2 Level B/behavioral harassment exposures). Likelihood of offshore exposure to surface ship sonar by southern resident killer whales is low given southern resident killer whale summer inshore Puget Sound preference, and winter nearshore transient movements along the Washington-Oregon-northern California coasts. Therefore, the Navy assumed that any May 2012 to May 2014 exposures based solely on the post-calculation, would be to offshore or transient stocks of killer whales and not to southern resident killer whales.

5.8.1.4 *Training Use May 2013 to May 2014*

The Navy's 2014 classified annual NWTRC Exercise Report contains the list of authorized systems and their reported use from 2 May 2013 to 1 May 2014. Of the sonar systems and explosives authorized under NMFS' NWTRC Final Rule, Letters of Authorization (LOA), and biological opinions (BO) for the NWTRC during this period, no sonar system exceeded any authorized amount and there was no offshore explosive use reported.

5.8.1.5 *Exposures during the current year and predicted through November 2015*

Table 61 contains NWTRC estimated post-calculation annual potential exposures from May 2013 to May 2014; cumulative species-specific estimates for the period from November 2010 through May 2014; and an estimate of total percentage of exposures compared to the amount authorized over 5-years if in the final year there was 100 percent system use.

The post-calculation estimates for this period indicate annual species-specific estimates of exposure for the period May 2013 to May 2014 range from zero percent of annual authorization to four percent of the annual authorization. In terms of cumulative potential exposures under the NWTRC Final Rule from November 2010 through May 2014, species-specific exposures range from 25 to 43 percent of total five-year authorization at the end of the fourth of five years.

In terms of quantity of potential exposures to Endangered Species Act (ESA) cetacean species, of the six ESA species (blue whale, fin whale, humpback whale, sei whale, sperm whale, Southern Resident killer whale), there was potentially only one exposure to sperm whales.

If the Navy were to use 100 percent of all sonar system and explosive use authorized under the NWTRC Final Rule, LOA, and the 2012 Opinion, the cumulative species-specific exposure estimates would be between 36 to 59 percent of total authorized over the five-year period. Takes of all ESA species would be below 50 percent of the authorized maximum. Navy post calculations for remaining years will not be comparable to previous years due to the change in modeling criteria; however relative percentages might be appropriate for comparison to previous performance.

Table 61. Navy Post-calculation Annual Potential Exposures from November 2010 through May 2014 and Cumulative Totals

Species	Authorized Level B Harassment			Predicted Level B, Harassment Takes Based on Reported Training								Percent of Predicted Takes vs. Authorized
	Nov 2010-May 2011	May 2011 - May 2012	May 2012-May 2015	November 2010 – May 2011		May 2011 – May 2012		May 2012 – May 2013		May 2013 – May 2014		Nov 2010 – May 2014
				Level B/ Harass	% Potential Takes Compared to Authorized	Level B/ Harass	% Potential Takes Compared to Authorized	Level B/ Harass (explosives)	% Potential Takes Compared to Authorized	Level B/ Harass (explosives	% Potential Takes Compared to Authorized	
Humpback whale	15	15	15	0	0%	2	13%	19 (0)	**127%**	0	0%	**35%**
Fin whale	40	69	57	0	0%	8	12%	61 (1)	**107%**	0	0%	**31%**
Blue whale	11	17	16	0	0%	2	18%	16	100%	0	0%	**30%**
Sei whale	1	1	1	0	0%	0	0%	1	100%	0	0%	**25%**
Sperm whale	84	127	111	1	1%	18	14%	130 (0)	**117%**	**1**	**1%**	**35%**

5.8.1.6 *Monitoring*

The Navy provided the following summary (also see Table 62) of monitoring related to the NWTRC training activities accomplished from November 2011 through May 2014. This information was provided as part of the *Comprehensive Marine Species Monitoring Report for The Navy's Northwest Training Range Complex 2011-2014*. U.S. Pacific Fleet, Pearl Harbor, Hawaii. Final July 1, 2014:

1) Long-term fixed passive acoustic monitoring is an effective way to determine seasonal species-specific occurrence of vocalizing and potentially foraging animals. It does not account for non-vocalizing animals. Passive acoustic monitoring can also be used to record natural and anthropogenic sounds leading to better assessment of ambient noise conditions.

By the summer of 2014, Navy funded passive acoustic monitoring will have been ongoing off the coast of Washington State for close to 10 years. Navy research funding and reporting occurred from 2004 to 2010 (Kerosky et al. 2013; Oleson and Hildebrand 2012; Oleson et al. 2009; Rice et al. 2015; Trickey et al. 2015). Under NWTRC compliance monitoring from 2011 to 2014, over 27,000 hours of passive acoustic data has been collected from two passive acoustic devices on the shelf and slope (DoN 2011; DoN 2012a; DoN 2013a).

Specific passive acoustic monitoring observations include:

- Future National Marine Fisheries Service (NMFS) and Navy adaptive management should be conducted with an eye toward reviewing the relevance of continued data collection. Toward that end, the Navy in 2014 to 2015 will focus on analysis from just one NWTRC passive device (slope site).

- Passive acoustic monitoring confirmed that highly infrequent low levels of Navy active mid-frequency sonar were detected by the two fixed monitoring sites off the Washington coast. From 2008 to 2013, passive sensors only detected four to seven days per year from temporally separated mid-frequency sonar events lasting at most a few hours in duration. This is consistent with the a) overall low level of at-sea sonar training in the NWTRC as compared to other Navy range complexes, and b) the general tendency for unit-level sonar and explosive training to occur further offshore, sometimes >50 miles.

- Passive acoustic monitoring has the potential, via expanded analysis tools, to begin addressing the possible impacts of anthropogenic sources on marine mammal vocalization and echolocation, with the assumptions that changes in vocalizations and echolocation rates are indicative of behavioral changes. However, this kind of analysis is better suited for those areas where the Navy in-water training occurs more frequently such as Southern California or Hawaii vice the more limited Navy in-water training within the NWTRC.

2) Satellite tracking tags can be an effective indicator of marine mammal distribution and movement patterns at short (days-weeks) and long time scales (months-year) (DoN 2011; DoN 2012a; DoN 2013a; Mate 2013; Schorr et al. 2013).

The Navy believes for future NWTRC tagging efforts from 2014 forward, longer term tags are preferred for continued monitoring. Long term tags will not only provide information on baleen whale distributions in terms of local bathymetric features, but also allow determination of percentages of time individuals spend within and outside of the NWTRC.

In particular, certain offshore sub-areas within NWTRC are more likely to have in-water Navy training events as compared to the rest of the NWTRC. Therefore, comparisons of baleen whale residence times and area restricted searches (potential foraging metric) in sub-areas of the NWTRC can be valuable in comparing potential baleen whale interactions or lack of interactions with Navy training events.

Specific tagging observations include:

- Navy funded gray whale projects from 2011 to 2013 in NWTRC provided valuable distribution information. To support the need for longer term tracking of additional cetacean species following the success of the gray whale tagging, the Navy funded a new large scale tagging effort for blue and fin whales. The focus of this study will be movement patterns and residency pattern of blue and fin whales along the U.S. West Coast, including within NWTRC. This project was funded in spring 2014 for a planned summer 2014 field season.

3) Finally, as the Navy prepares for future study question-based monitoring within the Pacific Northwest, the Navy funded a new study in the spring of 2014 to model offshore movements of Southern Resident killer whales. Work will be performed by scientists affiliated with NMFS' Northwest Fisheries Science Center (NWFSC).

The first year of this project occurred from fall 2014 through spring 2015 and involved the: a) deployment of 15 bottom-mounted acoustic monitoring devices, b) purchase of four (4) satellite tracking tags for eventual attachment to Southern Resident killer whales, c) and development of a new state-space model to predict Southern Resident killer whale offshore movement and habitat. Model development will be started concurrently with the 2014 to 2015 field data collection. Previously collected NWFSC passive acoustic and tagging data from the past two years of offshore Southern Resident effort will be used to initiate model development. Continued field efforts under the NWTT monitoring plan will continue for the fall 2015 to spring 2016 season.

Table 62. Monitoring Plan Metrics Accomplished in the NWTRC through May 2014

Metric	November 2011 to May 1, 2014
Navy Funded Opportunistic/Fully Funded Marine Mammal Tagging	1) Multiple Cetacean species: Ten (10) Andrews-style LIMPET (Low Impact Minimally Percutaneous External Transmitter) tags were purchased by the Navy and supplied to researchers at Cascadia Research Collective for use within a collaborative study of marine mammal movement patterns within offshore waters of Washington State. All ten (10) Navy-funded tags were deployed from 2011 to 2012 within offshore waters of Washington State. Satellite tags deployed during field efforts were associated with grants research from National Marine Fisheries Service's Alaska Regional Office and Southwest Fisheries Science Center, and a collaborative project with Washington Department of Fish and Wildlife. In total, over 21 tags were attached (see Schorr et al. (2013) and Section 2.2 in this report). 2) Gray whale: 19 Telonics ST-15 ultra-high frequency location only tags and Wildlife Computer Spot-5 tags were purchased by the Navy and Navy funded associated field work by Oregon State University. Tags were attached to Pacific Coast Feeding Group of gray whales from October to November 2012, and in October 2013 (see Mate (2013), Mate et al. (2014), and Section 2.2 in the report). 3) Baleen (blue and fin) whales: Up to 24 location-only SPOT-5 tags and 4 newly designed Advance Dive Behavior tags were planned for attachment in summer 2014. While tagging location will be in Southern California, the goal is to study and document blue and fin whale movements along the entire U.S. West Coast including Navy range areas like the Southern California Range Complex and NWTRC (See Section 2.2 in the report). 4) Pinniped: Started in late 2013, with multiple tags and visual observations to be done on pinnipeds around and adjacent to select Navy waterfront facilities within Puget Sound (see Section 2.4 of the report).
Deploy Two Long-term Passive Acoustic Monitoring (PAM) Devices	Two (2) high-frequency acoustic recording packages (HARP) from Scripps Institute of Oceanography were funded by Navy for deployment at offshore Washington State locations monitored under previous Navy Research funding from 2004 to 2010. Under the NWTRC monitoring plan and associated U.S. Pacific Fleet funding, the two devices have been in place from November 2011 to present. Through March 2013, the last series analyzed so far, over 27,000 hours of passive acoustic data have been analyzed for baleen whale calls; toothed whale calls/whistles/echolocation clicks; and anthropogenic sounds. These devices have been continuously maintained and data analyzed for the duration of this period. Analysis confirmed detection of four baleen whale species (blue whales, fin whales, gray whales, humpback whales); and seven toothed whale species. Ship and boat noise was common anthropogenic sound at both sites.

Future New Study Question Projects starting in fall 2014:

"What is the distribution, residency time, and spatial extent of Southern Resident killer whale winter movements off the coasts of Washington, Oregon, and northern California, and relationship of this movement to NWTRC?"

Navy-funded study in spring of 2014 on "Modeling the Distribution of Southern Resident Killer Whales in the Pacific Northwest."

5.8.1.7 *Navy Compliance Monitoring For the NWTRC 2014-2015*

For the fifth and final year of Navy Compliance Monitoring within the NWTRC (May 2, 2014 to November 2015), the Navy with NMFS concurrence during annual adaptive management meetings is restructuring the NWTRC monitoring metrics, so that at the end of the final year of monitoring (May 2, 2014 to May 1, 2015) there is an end focus on marine mammal tagging versus continued passive acoustic data collection.

Given continued fishery interaction and high shelf currents leading to equipment difficulties, along with the renewed focus on baleen whale tagging and new Southern Resident killer whale research, the Navy will only report on deployment and data analysis from one long-term bottom-mounted passive acoustic device, the slope HARP-QC discussed in Section 2.3 of the report. The shelf HARP-CE will be retrieved. Instead, a greater focus will be placed on results from the U.S. West Coast blue and fin whale tag as it relates to the NWTRC/NWTT, and on the start of the new Southern Resident killer whale study by NWFSC (Table 63).

Table 63. Navy NWTRC Compliance Monitoring For Year 5 Compared To Preceding Effort.

Monitoring Technique	Implementation and Transition	
	Year 4 (May 2, 2013-May 1, 2014)	Year 5 (May 2, 2014-November 2015)
Marine Mammal Tagging	Initiate contracting with focus on baleen whale tagging to prioritize blue whales and fin whales. Report results on FY13 funded study and resulting tagging through May 2014.	Purchase additional tags and continue collecting tag track data on blue whales and fin whales. Tag attachment will start in the summer of 2014 in Southern California. Tag tracks will be displayed for movements along the entire US West Coast including NWTRC Annual reporting of progress.
Passive Acoustic Monitoring	Present data analysis from two Navy funded offshore passive acoustic monitoring devices	Continue deployment of one (1) bottom-mounted passive acoustic device at the slope site (HARP-QC discussed in Section 2.3 of the report) Annual reporting of detections.
2014-2015 Study Question New Start	-	Begin new study: " Modeling the Distribution of Southern Resident Killer Whales in the Pacific Northwest " discussed in Section 3.1 of the report

In addition to offshore projects involving long-term passive acoustic monitoring and opportunistic or directed marine mammal tagging, the Navy from 2012 to 2014 also funded and conducted several marine mammal studies within Puget Sound for multiple species (e.g., DoN 2013c; Jeffries 2013).

The Navy assessed that long-term fixed passive acoustic monitoring is an effective way to determine seasonal species-specific occurrence of vocalizing and potentially foraging animals. It does not account for non-vocalizing animals. Passive acoustic monitoring can also be used to record natural and anthropogenic sounds leading to better assessment of ambient noise conditions. Passive acoustic monitoring has potential via expanded analysis to begin addressing possible impacts of anthropogenic sources on marine mammal vocalization and echolocation, with the assumptions that changes in vocalizations and echolocation rates are indicative of behavioral changes. However, the Navy concluded that this kind of analysis is better suited for those areas where the Navy in-water training occurs more frequently such as Southern California or Hawaii vice the more limited Navy in-water training within the NWTRC.

In the report the Navy also concluded that satellite tracking tags can be an effective indicator of marine mammal distribution and movement patterns at short (days to weeks) and long time scales (months to years). Longer term tag tracks are needed in order to better determine baleen whale distributions in terms of bathymetric features, and to determine what percentage of time individuals spend within the NWTRC and outside of the NWTRC.

Finally, the Navy is beginning to transition NWTRC compliance monitoring away from strictly metric-based accomplishments (i.e., number of devices deployed, number of tags attached), to a more region-specific and species-specific format. To that end, two new ecological based studies have been initiated in 2014. One study using passive acoustic tools, satellite tagging, and advanced modeling will attempt to refine predictions of offshore occurrence and locations for Southern Resident killer whales. Another new study using satellite location tags and advanced modeling will detail long-term blue and fin whale occurrence, migration, and local residency patterns along the U.S. West Coast including within and outside of NWTRC.

5.9 Scientific Research and Permits

Scientific research permits issued by the NMFS currently authorize studies listed species in the North Pacific Ocean, some of which extend into portions of the Action Area for the proposed project. Authorized research on ESA-listed whales includes 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 whales by harassment, with none resulting in mortality. Sea turtle research includes capture, handling, restraint, tagging, biopsy, blood sampling, lavage, ultrasound, and tetracycline injection. Table 64 describes the cumulative number of takes for each listed species in the Action Area authorized in scientific research permits.

Table 64. Authorized permitted takes of listed whales, pinnipeds, and sea turtles in the Pacific Ocean under the Endangered Species Act and the Marine Mammal Protection Act.

Species	2009-2014 lethal take	2009-2014 sub-lethal take
Blue Whale	0	107,785
Fin Whale	0	154,771
Western North Pacific Gray Whale	0	0
Sei Whale	0	41,745
Humpback Whale	0	372,260
North Pacific Right Whale	0	5,762
Sperm Whale	0	142,700
Main Hawaiian Island Insular False Killer Whale	0	9,195
Guadalupe Fur Seal	0	1,125
Hawaiian Monk Seal	142	23,437
Green Turtle	0	7,545
Hawksbill Turtle	0	1,085
Leatherback Turtle	0	1,178
Loggerhead Turtle	0	519
Olive Ridley Turtle	0	2,198

5.10 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 NWTT Action Area. The Action Area includes the inshore and offshore portions of the Northwest Training Range Complex (NWTRC) in Northern California, Oregon, and Washington State, the Naval Undersea Warfare Center (NUWC) Keyport Range Complex in and around Puget Sound, and the Southeast Alaska Acoustic Measurement Facility (SEAFAC) in the Western Behm Canal, Alaska. (See Figure 2, Figure 3, and Figure 4).

Loss of natural habitat as a result of population growth and urbanization is a constant threat to the birds, mammals, fish, reptiles, amphibians and invertebrates in the Georgia Basin-Puget Sound region. Although killer whales in British Columbia are assessed as vulnerable by the Conservation Data Centre in British Columbia, there is great concern about the status of the southern resident killer whale population that resides in the Georgia Basin-Puget Sound region. Recent studies have revealed high persistent organic pollution levels in the tissues of this population. There is also concern about recent mortalities in the population, a reduction in food (prey) availability and increasing stress from whale watchers and boaters.

Sixty-four of the vertebrate species that are native to Puget Sound are considered at some risk of extinction within the Sound, including one out of four native reptile species, 18 percent of the freshwater fish species, 15 percent of all native amphibian species, 12 percent of all native mammal species, and 12 percent of the native breeding bird species. Forty-one of the 298 vertebrates that are native to the Georgia Basin are either threatened, endangered, or candidates for these designations, including white sturgeon, Vancouver Island marmot, Oregon spotted frog, and sharp-tailed snake. Fourteen of the 41 species of freshwater fish that are native to the Georgia Basin and 10 mammal species are considered at risk of population collapses, declines, or extinction within the Georgia Basin. The Canadian government is examining 30 other species that are native to the Georgia Basin for potential as endangered species.

Southern resident killer whales were listed as endangered because of their exposure to the various stressors that occur in the range of the species and in the Action Area for this consultation. Exposure to those stressors resulted in the species' decline from around 200 individuals to about 67 individuals in the 1970s and the species' apparent inability to increase in abundance above the 75 to 90 individuals that currently comprise this species. The phenomena that are impairing the ability of the species to increase abundance potentially increase the extinction risk of southern resident killer whales and could amplify the potential consequences of human-related activities on this species. Based on their population size and population ecology (that is, slow-growing mammals that give birth to single calves with several years between births), we assume that southern resident killer whales would have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities that result in the death or injury of individual whales (for example, ship strikes or entanglement) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) *as well as* endogenous threats resulting from the small size of their population. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer southern resident killer whales remain in these circumstances, the greater their extinction probability becomes. (Williams et al. 2014b) indicated that northern resident killer whales showed moderate (severity score 2 to 4) responses to the presence of the large ships that use Johnstone Strait in summer months, but behavioral responses were best explained by combinations of time (Year and Month), age of the animal, number of ships (CAR, COL and TUG) and the broadband noise level received by the whale (RL_rms).

NMFS has consistently concluded that the various fisheries that incidentally capture endangered or threatened salmon or steelhead in the Action Area are not likely to jeopardize the continued existence of those species. However, the effects of the fisheries combined with the effects of water quality degradation in the Puget Sound – Georgia Basin region on Puget Sound Chinook salmon, Hood canal summer-run chum salmon, and Puget Sound steelhead are not known but have increased the extinction risks of other endangered or threatened anadromous fish species (for example, delta smelt in the San Francisco estuary).

Thus far under the current MMPA rule (2010 to 2015), the Navy's training activities on the NWTRC have resulted in estimated take that are well below the five-year levels evaluated in the previous Biological Opinion for the Northwest Training Range Complex. There have not been any vessel strikes of any species during training activities in the NWTRC during the five-year period.

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), whereas others result in more indirect (e.g., a fishery that impacts prey availability) or non-lethal (e.g., whale watching, anthropogenic sound) impacts. 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 many of the species in this Opinion are wide ranging and subject to stressors in locations well beyond the NWTT Action Area.

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 15, 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 despite the potential negative impacts of the *Environmental Baseline*. Therefore, while 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 the suite of conditions described in the *Environmental Baseline* is 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 jeopardy analysis relies 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.

The adverse modification analysis considers the impacts on the conservation value of designated critical habitat. This biological opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 CFR 402.02, which was invalidated by Gifford Pinchot Task Force v. United States Fish and Wildlife Service, 378 F.3d 1059 (9th Cir. 2004), amended by 387 F.3d 968 (9th Cir. 2004). Instead, we have relied upon the statutory provisions of the ESA to complete our analysis with respect to critical habitat.

6.1 Stressors Associated with the Proposed Action

The potential stressors (risks) to ESA-listed species that we analyzed based on the training and testing activities the Navy proposes to conduct in the NWTT Action Area are summarized in Table 65.

Table 65. 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 (non-impulse or impulse), 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 and testing 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 testing 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 and testing 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 and testing activities throughout the Action Area, contributing both airborne and underwater sound to the ocean environment. Aircraft used in training and testing generally have 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,

Stressor	Description of Stressor
	passenger vessels, and oil tankers (all over 65 ft. [20 m] in length), is heaviest near and between the major shipping ports.
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.
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 10–15 knots, and submarines generally operate at speeds in the range of 8–13 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, but greater activity would be expected near ports than in other areas of the Action Area. 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 and testing activities are separated into four categories: (1) fixed-wing aircraft, (2) rotary-wing aircraft, (3)

Stressor	Description of Stressor
	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.
Entanglement (fiber optic cables and guidance wires, and decelerators/parachutes)	The only type of cable expended during training and testing 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 and testing 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.
Ingestion (munitions and military expended materials other than munitions)	The only munitions small enough for a marine mammal or sea turtles 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 sea turtles. Many different types of explosive munitions

Stressor	Description of Stressor
	can result in fragments that are expended at sea during training and testing 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).
Secondary (explosion byproducts, metals, chemicals, sedimentation, and transmission of marine mammal diseases and parasites)	Secondary stressors associated with some training and testing 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

Stressor	Description of Stressor
	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 and testing activities involving ship hulks, targets, ordnance, munitions, and other military expended materials. Several training and testing 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 NWTT Action Area

Table 66 below provides an overview of training and testing activities levels by stressor. Stressors that are likely to adversely affect ESA-listed species are bolded.

Table 66. Overview of Training and Testing Activity Levels by Stressor

Components	Area	Number of Training and Testing Activities in the Action	
		Training	Testing
Acoustic Stressors			
Sonar and other active sources (hours)	**Offshore Area**	**551**	**977**
	Inland Waters	**407**	**5,448**
	W. Behm Canal	**0**	**2,762**
Sonar and other active sources (items)	**Offshore Area**	**1,616**	**943**
	Inland Waters	**0**	**1,308**
	W. Behm Canal	**0**	**0**
Explosives (items)	**Offshore Area**	**142**	**148**
	Inland Waters	**42**	**0**
	W. Behm Canal	**0**	**0**
Weapons firing, launch, and impact noise	Offshore Area	See Analysis	
	Inland Waters		
	W. Behm Canal		
Activities including vessel noise	Offshore Area	1,108	138
	Inland Waters	310	582
	W. Behm Canal	0	60
Activities including aircraft noise	Offshore Area	8,140	80
	Inland Waters	117	20
	W. Behm Canal	0	0
Energy Stressors			
Activities including electromagnetic devices	Offshore Area	0	0
	Inland Waters	1	0
	W. Behm Canal	0	0
Physical Disturbance and Strike Stressors			
Activities including vessels	Offshore Area	1,116	158
	Inland Waters	310	602
	W. Behm Canal	0	60
Activities including in-water devices	Offshore Area	493	134
	Inland Waters	1	628
	W. Behm Canal	0	0
Military Expended Materials	Offshore Area	198.028	3,922
	Inland Waters	3,085	513
	W. Behm Canal	0	0
Entanglement Stressors			
Fiber Optic Cables and Guidance Wires	Offshore Area	0	131
	Inland Waters	1	245
	W. Behm Canal	0	0

Components	Area	Number of Training and Testing Activities in the Action	
		Training	**Testing**
Decelerator/Parachutes	Offshore Area	8,952	1,210
	Inland Waters	0	40
	W. Behm Canal	0	0
Ingestion Stressors			
Military expended materials from munitions	Offshore Area	183,374	1,946
	Inland Waters	3,042	6
	W. Behm Canal	0	0
Military expended materials other than munitions	Offshore Area	9,654	2,057
	Inland Waters	43	630
	W. Behm Canal	0	0
Secondary Stressors			
Habitat (sediments and water quality; air quality)	Offshore Area	Qualitative	
	Inland Waters		
	W. Behm Canal		
Prey	Offshore Area	Qualitative	
	Inland Waters		
	W. Behm Canal		

6.1.2 Summary of Effect Determinations By Stressor

Table 67 and Table 68 below summarize 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 (See Section 4.2) 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 (See Sections 6.7 through 6.10). Stressors from training and testing activities (See Sections 2.1 and 2.2) that were determined to have No Effect (NE) or were Not Likely to Adversely Affect (NLAA) are not discussed further in this Opinion. The Navy's determinations of No Effect are provided for reference although they did not request consultation on those stressors in relation to the listed species or critical habitat. We concurred with Navy's NLAA determinations for all stressors except its determinations for explosives and salmonids, which during consultation, we determined Likely to Adversely Affect (LAA) species. They Navy subsequently changed their determinations in an amendment to the biological evaluation. The stressors that were determined to be LAA and the contributing training and testing activities are likely to result in take of one or more ESA-listed species.

Table 67. 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	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	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Humpback whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Blue whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Fin whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Sei whale	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Western North Pacific gray whale	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Sperm whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Southern Resident killer whale	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Guadalupe fur seal	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
SEA TURTLES																
Leatherback sea turtle	**LAA**	NE	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Fish																
Chinook Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Coho Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Chum Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Sockeye Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Steelhead	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Rockfish	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Pacific Eulachon – Southern DPS	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Green Sturgeon	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Table 68. Summary of NMFS' Effect Determinations by Stressor - Testing Activities

Common Name	Overall ESA Determination	Effect Determinations by Stressor (TESTING 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	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	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Humpback whale	**LAA**	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Blue whale	**LAA**	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Fin whale	**LAA**	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Sei whale	NLAA	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Western North Pacific gray whale	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Sperm whale	**LAA**	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Southern Resident killer whale	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
Guadalupe fur seal	**LAA**	**LAA**	NLAA	NE	NLAA	NLAA	NE	NE	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA
SEA TURTLES																
Leatherback sea turtle	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Fish																
Chinook Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Coho Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Chum Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA
Sockeye Salmon	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Common Name	**Overall ESA Determination**	**Effect Determinations by Stressor (TESTING 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**	**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**	
Steelhead	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Rockfish	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Pacific Eulachon – Southern DPS	**LAA**	NLAA	**LAA**	NLAA	NLAA	NE	NE	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Green Sturgeon	NLAA	NLAA	NLAA	NLAA	NLAA	NE	NE	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 NWTT Action Area, it is carried forward in our effects analysis.

6.2.1 Vessel Strike

There has never been a vessel strike to a whale during any of the training or testing activities proposed in the NWTT Action Area. There has been only one whale strike in the Pacific Northwest by the Navy since such records have been kept (June 1994 to present). In August 2012, a San Diego homeported DDG at-sea about 35 nm west of Coos Bay, Oregon struck a whale while transiting to San Diego after visiting Seattle for a Fleet Week celebration. The whale (believed to be a minke whale) was last seen swimming away from the location. The fate of the animal is unknown and although no blood or other obvious indications of injury to the whale were detected, this does not negate the possibility that there may have been serious internal injury to the whale resulting from the encounter. Given the only documented Naval vessel whale strike in the Pacific Northwest since 1994 did not occur as part of training and testing and was not believed to be an ESA-listed whale, strikes of ESA-listed whales associated with NWTT activities is not reasonably certain to occur. The training and testing activities proposed that involve vessel movement are described in Chapter 2 (Description of Proposed Action and Alternatives) of the Final EIS/OEIS. Activities involving vessel movement would be widely dispersed throughout the Action Area.

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. 2008b; 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).

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 testing 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 operate at the slowest speed possible consistent with either transit needs or training or testing 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 Navy vessel to strike a cetacean during the course of training and testing activities in NWTT Action Area, we do not believe this is reasonably certain 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 employs minimization measures to reduce the likelihood for a surface vessel to strike a large whale.

Given the low abundance of leatherback sea turtles in the Action Area (particularly in inland waters where vessel traffic is most concentrated), the low number of vessels associated with NWTT activities (relative to commercial traffic in the area and relative to the amount of Navy vessel traffic in other training ranges where strikes have been documented such as the HSTT Study Area), the use of strike mitigation measures, the potential for the bow wave of large vessels to preclude a vessel strike, and the lack of documented cases of Navy vessels in the NWTT Action Area striking a leatherback sea turtle, the likelihood of a Navy vessel associated with NWTT activities to strike a leatherback sea turtle is so low as to be discountable. Consequently, NMFS has determined that the likelihood of vessel strike during training and testing over the five-year period of the MMPA rule and continuing into the reasonably foreseeable future is sufficiently low to be discountable and therefore is not discussed further in this Opinion.

6.2.2 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 sea turtles 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 but no TTS exposures of ESA-listed marine mammals are predicted by Navy modeling efforts. Marine mammals, sea turtles, 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, sea turtles, 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, an increase in the likelihood of injury from significant disruption of breeding, feeding, or sheltering for ESA-listed marine mammals and sea turtles. Therefore, the effects of weapons firing, launch, and impact noise on ESA-listed marine mammals, sea turtles, and ESA-listed fish would not rise to the level of take and any behavioral impacts would be insignificant.

6.2.3 Effects of Aircraft Noise

Studies have shown that aircraft presence and operation can result in changes in behavior of cetaceans (Arcangeli and Crosti 2009; Holt et al. 2009b; 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 NWTT Action Area 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 and sea turtles.

We did not estimate 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. Nevertheless, we assume any ESA-listed species that occur in the Action Area during training exercises and testing activities that involve aircraft are likely to be exposed to minor acoustic stimuli associated with aircraft traffic.

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.

6.2.3.1 ***Mysticetes***

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.

6.2.3.2 ***Odontocetes***

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).

6.2.3.3 ***Sea Turtles***

Based on sea turtle sensory biology (Bartol et al. 1999b; Ketten and Bartol 2005; Ketten and Bartol 2006; Lenhardt et al. 1994a; Ridgway et al. 1969), sound from low flying aircraft could be heard by a sea turtle at or near the surface. Turtles might also detect low flying aircraft via visual cues such as the aircraft's shadow. Hazel et al. (2007) suggested that green turtles rely more on visual cues than auditory cues when reacting to approaching water vessels. This suggests that sea turtles might not respond to aircraft overflights based on noise alone. Based on limited morphological and scientific study, ESA-listed fish are likely less sensitive to above-water acoustic noise, such as that originating from aircraft, than marine mammals.

6.2.3.4 ***Conclusion***

In conclusion, the low number of aircraft flights, typical altitudes of flights, sporadic occurrence of flights, limited duration of flights, and the lack of substantial sound propagation into the water column from aircraft indicates there is a low probability of exposing marine mammals, sea turtles, and ESA-;listed fish to aircraft noise at perceivable levels. In the event an ESA-listed species was exposed to aircraft noise, it would likely result in temporary behavioral responses. These behavioral responses would not increase the likelihood of injury from significantly disrupting breeding, feeding, or sheltering and would not rise to the level of take. Therefore, the effects of aircraft noise on ESA-listed species is insignificant and not likely to adversely affect them.

6.2.4 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 ± SE) 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 1985b; 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 1991a; 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 1985b; Kruse 1991a; 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.

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 1991a). 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 1991a). 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. Typical Navy training and testing activities 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.

For surface vessels, the set of variables that help determine whether marine mammals are likely to be disturbed include: (1) the number of vessels in a marine mammal's perceptual field and the animal's assessment of the risks associated with those vessels; (2) the distance between vessels and marine mammals; (3) the vessel's speed and path; (4) the predictability of the vessel's path; (5) noise associated with the vessel and the rate at which the engine noise increases; (6) the type of vessel; and (7) the behavioral state of the animal. Because of the number of vessels involved in U.S. Navy training exercises and testing 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 exercises, unit- and intermediate-level exercises, and testing 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). As noted previously, 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 µPa^2-s. Bassett et al. (2010) measured mean sound pressure levels at Admiralty Inlet from commercial shipping at 117 dB re 1 µPa with a maximum exceeding 135 dB re 1 µPa on some occasions.

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)).

Bassett et al. (2012b) recorded vessel traffic over a period of just under a year as large vessels passed within 20 km of a hydrophone site located at Admiralty Inlet in Puget Sound. During this period there were 1,363 unique Automatic Identification System transmitting vessels recorded. Navy vessels, given they are much fewer in number, are a small component of overall vessel traffic and vessel noise in most areas where they operate and this is especially the case in the

Action Area (see Mintz and Filadelfo (2011)) concerning a general summary for the U.S. Exclusive Economic Zone). In addition, 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).

6.2.4.1 ***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 react 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 NWTT Action Area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004) 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. 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Nowacek et al. 2004; Terhune and Verboom 1999). Although this may minimize potential disturbance from passing ships, it does increase the whales' vulnerability to potential ship strike. The regulated approach distance for North Atlantic right whales is 500 yards (yd.) (457 m) (NMFS 1997a).

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 2010a).

6.2.4.2 ***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.

Wursig et al. (1998) reported most *Kogia* species and beaked whales react negatively to vessels by quick diving and other avoidance maneuvers. Cox et al. (2006) noted very little information is available on the behavioral impacts of vessels or vessel noise on beaked whales. A single observation of vocal disruption of a foraging dive by a tagged Cuvier's beaked whale documented when a large noisy vessel was opportunistically present, suggests that vessel noise may disturb foraging beaked whales (Aguilar Soto et al. 2006). Tyack et al. (2011b) noted the result of a controlled exposure to pseudorandom noise suggests beaked whales would respond to vessel noise at similar received levels to those noted previously for mid-frequency sonar.

Most delphinids react neutrally to vessels, although both avoidance and attraction behavior is known (Hewitt 1985a; Wursig et al. 1998). Avoidance reactions include a decrease in resting behavior or change in travel direction (Bejder et al. 2006a). Incidence of attraction includes harbor porpoises approaching a vessel and common, rough-toothed, and bottlenose dolphins bow riding and jumping in the wake of a vessel (Norris and Prescott 1961; Shane et al. 1986) (Ritter 2002; Wursig et al. 1998). A study of vessel reactions by dolphin communities in the eastern tropical Pacific found that populations that were often the target of tuna purse-seine fisheries (spotted, spinner and common dolphins) show evasive behavior when approached; however populations that live closer to shore (within 100 nm; coastal spotted and bottlenose dolphins) that are not set on by purse-seine fisheries tend to be attracted to vessels (Archer et al. 2010a; Archer et al. 2010b). Pirotta et al. (2015) quantified the effect of boat disturbance on bottlenose dolphin foraging and determined that foraging activity was more affected by boat presence than noise level.

Killer whales, the largest of the delphinids, are targeted by numerous small whale-watching vessels in the Pacific Northwest. For the 2012 season, it was reported that 1,590 vessel incidents were possible violations of the federal vessel approach regulations or MMPA and ESA laws as well (Eisenhardt 2013). Research suggests that whale-watching distances may be insufficient to prevent behavioral disturbances due to vessel noise (Noren et al. 2009). In 2012, there were 79 U.S. and Canadian commercial whale watch vessels in the Haro Strait region (Eisenhardt 2013). These vessels have measured source levels that ranged from 145 to 169 dB re 1 µPa at 1 m. The sound they produce underwater has the potential to result in behavioral disturbance, interfere with communication, and affect the killer whales' hearing (Erbe 2002b). Killer whales foraged significantly less and traveled significantly more when boats were within 328 ft. (100 m) (Kruse 1991b; Lusseau et al. 2009; Trites and Bain 2000; Williams and Noren 2009; Williams et al. 2002b). These short-term feeding activity disruptions may have important long-term population-level effects (Lusseau et al. 2009; Noren et al. 2009). The reaction of the killer whales to whale-watching vessels may be in response to the vessel pursuing them, rather than to the noise of the vessel itself, or to the number of vessels in their proximity.

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).

6.2.4.3 ***Conclusion***

We recognize that Navy vessels almost certainly 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). 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).

We considered the research and reports cited above and conclude that ESA-listed marine mammals, sea turtles, ESA-listed fish are likely to either not react or exhibit an avoidance behavior. 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 and turtles. 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.

Behavioral disruptions of ESA-listed species resulting from the presence of vessels or submarines are expected to be temporary. Animals are expected to resume their migration, feeding, or other behaviors with no significant impact to their breeding, feeding or sheltering. Marine mammals react to vessels in a variety of ways and seem to be generally influenced by the activity the marine mammal is engaged in when a vessel approaches (Richardson et al. 1995d). Some respond negatively by retreating or engaging in antagonistic responses while other animals ignore the stimulus altogether (Terhune and Verboom 1999; Watkins 1986). Given the short duration of vessel noise stressors, the infrequency of this stressor, and the temporary nature of biological responses of marine mammals, sea turtles and ESA-listed fish to this stressor, these ESA-listed species are either not likely to respond or are not likely to respond in ways that might be adverse (the responses might represent an approach or attentive movement, a small change in orientation in the waters, etc.).

In conclusion, ESA-listed species are either not likely to respond to vessel noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding or sheltering. Therefore, the effects of noise from Navy vessels on ESA-listed species would not rise to the level of take and is considered insignificant.

6.2.5 Effects of Fiber Optic Cables, Guidance Wires, and Decelerators/Parachutes from Entanglement

Expended materials from U.S. Navy training and testing may pose a risk of entanglement or ingestion to marine mammals, sea turtles, and ESA-listed fish in the NWTT Action Area. These interactions could occur at the sea surface, in the water column, or on the seafloor. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with military expended materials could result in negative sub-lethal effects, mortality, or result in no impact.

Expended materials from Navy training and testing may include the following: fiber optic cables, guidance wires, parachutes, 55 gallon drums, sonobuoy tubes, unrecovered aircraft stores (e.g., fuel tanks, carriages, dispensers, racks), ship hulks, 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 sea turtles 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 NWTT Action Area, 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, fish (e.g., salmon, eulachon, rockfish), and sea turtles. Bottom feeding animals have an increased likelihood of encountering expended materials because they may find them during feeding long after the training or testing event has occurred. If encountered, foraging animals may ingest the item or become entangled by it.

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 or sea turtle 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, sea turtle) 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 presumably 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.

6.2.6 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, sea turtles, 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 expected 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 is limited to sperm whales. 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, sea turtles, and ESA-listed fish 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 increased likelihood of injury from significant disruption of 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 occur 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 increased likelihood of injury from significant disruption of 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, 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.

6.2.7 Effects of Electromagnetic Devices

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 exercises and testing 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 being exposed to electromagnetic energy at sufficient intensities to create a biologically relevant response is so low as to be discountable. Therefore, electromagnetic energy from mine neutralization equipment is not likely to adversely affect ESA-listed cetaceans and will not be considered further in this Opinion.

Sea turtles use geomagnetic fields to navigate at sea, and therefore changes in those fields could impact their movement patterns (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b). Turtles in all life stages orient to the earth's magnetic field to position themselves in oceanic currents; this helps them locate seasonal feeding and breeding grounds and to return to their nesting sites (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b). Experiments show that sea turtles can detect changes in magnetic fields, which may cause them to deviate from their original direction (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b). Sea turtles also use nonmagnetic cues for navigation and migration, and these additional cues may compensate for variations in magnetic fields. Experimental studies show that hatchlings and juvenile turtles are sensitive to the earth's natural magnetic field and they can distinguish magnetic inclination in different places during their migration routes (Lohmann 1991; Luschi et al. 2007). Sea turtles can distinguish magnetic differences lighter than 9 milliteslas (or 9,000 microteslas (μT)) (Lohmann et al. 2001; Lohmann et al. 1999).

If located in the immediate area (within about 650 ft. [200 m]) where electromagnetic devices are being used, sea turtles could deviate from their original movements. Even if detectable, the electromagnetic devices used in training exercises and testing activities are not expected to cause more than a short-term behavioral disturbance to sea turtles because of the: (1) relatively low intensity of the magnetic fields generated (0.2 μT [or 200 nT] at 200 m [656.2 ft.] from the source), (2) very localized potential impact area, and (3) temporary duration of the activities (hours). Impacts of exposure to electromagnetic stressors are not expected to result in substantial changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment, and are not expected to result in population-level impacts. Therefore, the effects of electromagnetic energy from mine neutralization equipment are insignificant and not likely to adversely affect ESA-listed sea turtles.

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. 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.

We assume that ESA-listed fish most likely avoid sources of strong electromagnetic fields and that any potential behavioral responses would be very minor and only occur very near the source. Therefore, we conclude that ESA-listed fish are not likely to be adversely affected by electromagnetic devices during Navy training and testing activities.

6.2.8 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. Therefore, the likelihood of striking a marine mammal, sea turtle, or ESA-listed fish is so unlikely as to be discountable. Any ESA-listed marine mammal, sea turtle, or fish species is likely to exhibit behavioral avoidance of in-water devices in the event they are within range to detect them. Any such avoidance behavior would be of short duration and intensity that it will not increase the likelihood of injury from disruption of breeding, feeding, or sheltering and would not rise to the level of take.

6.2.9 Secondary Stressors

This section analyzes potential impacts to marine mammals, sea turtles, and ESA-listed fish exposed to stressors indirectly through impacts to their habitat or prey or through the introduction of parasites or disease. Stressors from Navy training and testing activities could pose indirect impacts to marine mammals via habitat or prey. These include (1) explosives and byproducts, (2) metals, (3) chemicals, and (4) transmission of disease and parasites. The first three 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.

Metals are introduced into seawater and sediments as a result of training and testing activities involving the destruction of ship hulks, 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 is 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. 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). Polychlorinated biphenyls are present in target vessels used in sinking exercises but the Environmental Protection Agency considers the quantity introduced from these Navy exercises to be within the standards of the Marine Protection, Research, and Sanctuaries Act. Rapid dilution of introduced chemicals would occur and toxic concentrations are unlikely to be encountered in seawater (Navy 2013).

The primary vector through which parasites of disease would be transferred to new locations and the ESA-listed species there would be through the deployment of marine mammals used by the Navy's Marine Mammal Systems. Pinnipeds and dolphins used in the Marine Mammal Program are under exceptional care and control. Navy animals receive regular veterinarian care, including predeployment exams, regular deworming, and regional screening for specific pathogens of interest (Navy 2013). The animals are fed restaurant-quality fish to minimize the likelihood of parasite ingestion and animal waste is collected and managed to control the potential spread of parasites. Prior to animal deployment Navy personnel observe the surrounding area and if wild marine mammals are spotted animal deployment is delayed. Contact between Navy animals and wild animals is minimized to the greatest extent possible. In the 40 years the Marine Mammal Program has been operating there has been no known disease or parasite transmissions from Navy animals to wild animals (Navy 2013). Given the exceptional care Navy animals receive, the waste disposal protocols, the minimal time Navy animals are in contact with wild animals, and the 40 years of their use without incident, it is unlikely parasites or diseases will be transferred to ESA-listed species.

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, sea turtle 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, other chemicals, or parasites/disease resulting from NWTT activities are so unlikely as to be considered discountable. Therefore, secondary stressors from NWTT activities are not likely to adversely affect ESA-listed species.

6.2.10 Indirect Effects of Injury and Mortality of Chinook Salmon from Explosions on Southern Resident Killer Whales

Southern Resident killer whale have the potential to be adversely affected in the NWTT Action Area. The injury and/or mortality of Chinook salmon that would result from explosions could result in some level of harm to Southern Resident killer whales by reducing prey availability if the extent of take of salmon is large enough. A reduction in availability of salmon may cause animals to forage for longer periods, travel to alternate locations, or abandon foraging efforts. Hilborn et al. (2012) reported that information from Southern Resident killer whales in the summer indicates a heavy reliance on Chinook salmon. As Chinook salmon abundance declines in the fall, the diet data show that chum salmon and other species become more important. There are little winter diet data, but the data that do exist also suggest the importance of Chinook salmon (Hilborn et al. 2012).

The age distribution of Chinook salmon consumed by Southern Resident killer whales does not match the predicted age distribution of Chinook salmon thought to be available to killer whales. Instead, Southern Resident killer whales consume greater proportions of the larger (older) fish (particularly the 4- and 5-year-olds) relative to the overall numbers of Chinook salmon present (Hilborn et al. 2012). Accordingly, we assessed the injury and mortality of Chinook salmon from Navy training and testing activities and the percent reduction of the adult Puget Sound Chinook salmon. The only activity we determined may result in injury of death of ESA-listed Puget Sound Chinook salmon was detonations during training and testing in both inland and offshore waters. The percent of adult Puget Sound Chinook salmon that we estimated would be killed or injured by these detonations is listed in Table 69. Details on the methodology used to estimate this level of impact is in Section 6.11.

Table 69. The percent of ESA-listed adult Puget Sound Chinook salmon that would be affected (injured or killed) by Navy detonations.

Species	Location	Natural	Hatchery: adipose intact	Hatchery: adipose clip
Chinook	Inland	0.0077	0.0001	0
	Offshore	0.0050	0.0051	0
	Total	0.0127	0.0052	0

As described in further detail in Section 6.11, this extent of injury and mortality of Chinook salmon is extremely low and would not be detectable above baseline conditions and assessed levels of take of Chinook salmon from activities including commercial and recreational fisheries. Therefore, the extent of effects to Southern Resident killer whales from this adverse impact is not anticipated to result in significant behavioral effects and would not likely increase the risk of injury or mortality for whales currently in the population. We conclude that the indirect effects to Southern Resident killer whales from this percent reduction in availability of Puget Sound Chinook salmon to be insignificant, would not rise to the level of take, and is not 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 testing 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 NWTT Final EIS/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.

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).

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 and testing 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). Prior experimental work has also demonstrated the post-mortem presence of bubbles following decompression in laboratory animals can occur as a result of invasive investigative procedures (Stock et al. 1980).

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. 2005b; 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. 2005b; 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 8 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).

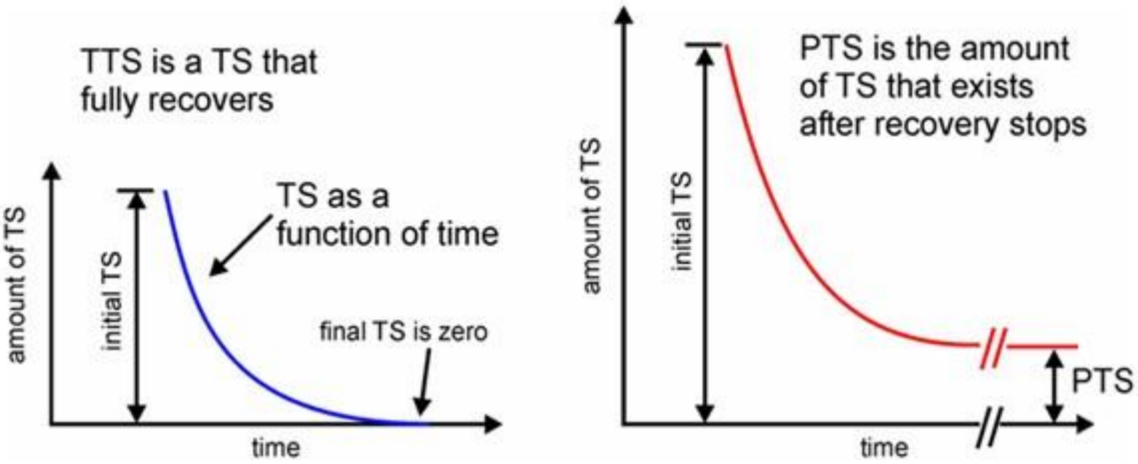

Figure 8. Two Hypothetical Threshold Shifts, Temporary and Permanent

Hearing loss, or auditory fatigue, in marine mammals has been studied by a number of investigators (Finneran et al. 2005b; 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. 2014a; 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. 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. 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. 2007). 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.

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 and others (Richardson et al. 1995d). More recent reviews (Ellison et al. 2012; Nowacek et al. 2007; Southall et al. 2009) (Southall et al. 2007b) 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. 2013b).

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. 1996a) 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. 2004). 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 exercises 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 1975a). 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 1975a).

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. 2005). 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 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.

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 and testing activities in the NWTT Action Area. 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. However, to our knowledge, empirical data has not confirmed this to be the case. 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. 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

2003a; 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 2003a; 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 2011j; NMFS 2011k; NMFS 2011m). 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), 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 and not the important variable, which is 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 and testing 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 and testing 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 and testing 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 and testing 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 and testing 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 NWTT Action Area. 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.

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 9). 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).

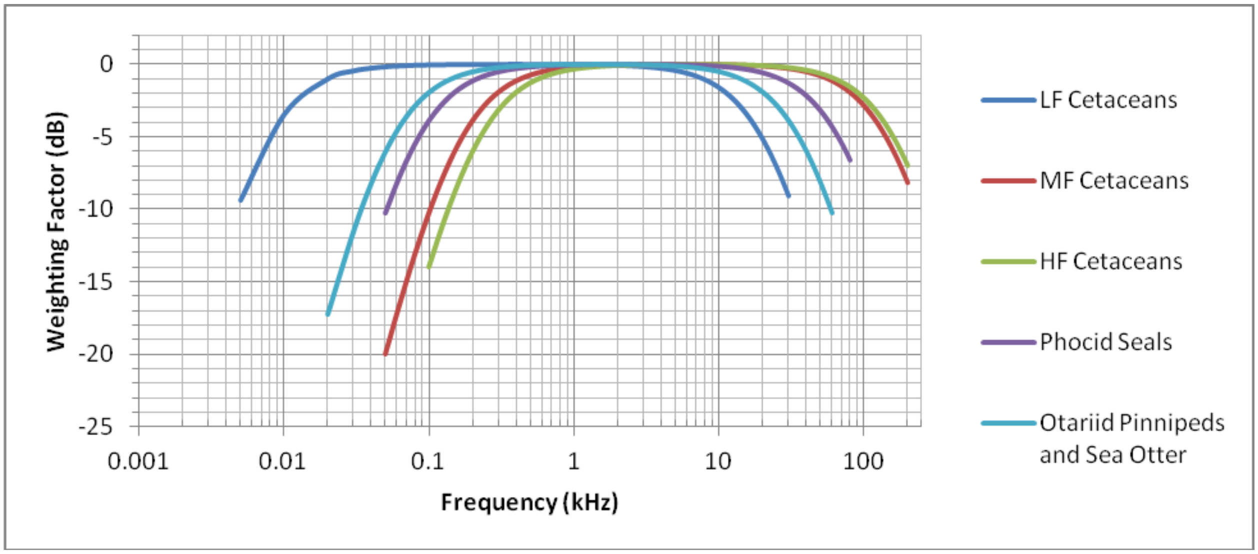

Figure 9. Type I Auditory Weighting Functions Modified from the Southall et al. (2007) M-Weighting Functions

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 10. 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 10) 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 10) 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.

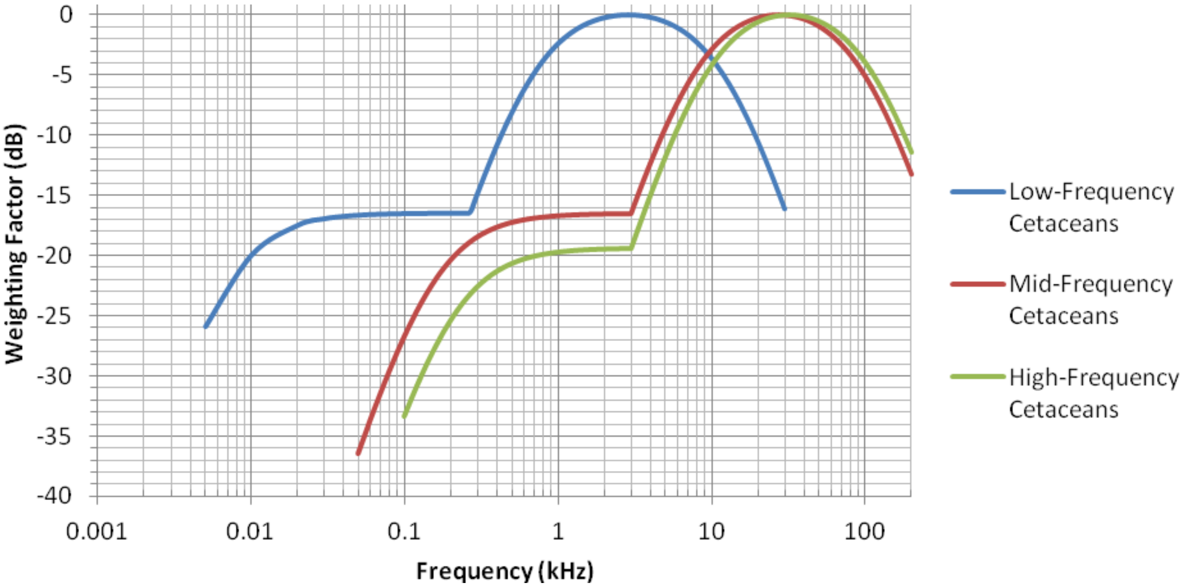

Figure 10. Type II Weighting Functions for Low-, Mid-, and High-Frequency Cetaceans

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 70) 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 70. 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)
High-Frequency Cetaceans	Porpoises and *Kogia* spp.	152 dB re 1 µPa^2-s SEL (Type II weighting)	172 dB re 1 µPa^2-s SEL (Type II weighting)
Phocid Seals (underwater)	Northern Elephant & Harbor Seals	183 dB re 1 µPa^2-s SEL (Type I weighting)	197 dB re 1 µPa^2-s SEL (Type I 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)
Mustelidae (underwater)	Sea Otters		

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

Temporary threshold shift involves no tissue damage, is by definition temporary, and therefore is not considered injury. 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. 2005b; 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. (2012d) 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 71 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 71. Approximate Maximum Ranges to the Onset of Temporary Threshold Shift for Four Representative Sonar 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
High-frequency cetaceans	2,170-7,570	4,050-15,350	5,430-19,500	90	180-190	260-950	< 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
Phocid seals & Manatees	70-1,720	200-3,570	350-4,850	< 50	100	150	< 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 2008e). 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).

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 testing (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 72 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 72. 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
High-Frequency Cetaceans	100	20	10
Phocid Seals	80	10	<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 10, to predict PTS.

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 73. 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 73. 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)	187 dB re 1 μPa^2-s (mid-freq weighting)			

Group	Species	Onset TTS	Onset PTS	Onset Slight GI Tract Injury	Onset Slight Lung Injury	Onset Mortality
		or 224 dB Peak SPL	or 230 dB Peak SPL			
High-Frequency Cetaceans	Porpoises and *Kogia* spp.	146 dB re 1 µPa^2-s (mid-freq weighting) or 195 dB Peak SPL	161 dB re 1 µPa^2-s (mid-freq weighting) or 201 dB Peak SPL			
Phocid Seals (In-Water)	Harbor, bearded, hooded common, spotted, ringed, harp, ribbon and gray seals	177 dB re 1 µPa^2-s (phocid weighting) or 212 dB Peak SPL	192 dB re 1 µPa^2-s (phocid weighting) or 218 dB Peak SPL			

Equations:

(1) $$\boxed{-39.1M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-\sec}$$

(2) $$\boxed{=91.4M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-\sec}$$

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, DRm = 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

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 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 behaviors that would qualify as harassment (as that term is defined by the MMPA applicable to military readiness activities, such as the Navy's testing and training with mid-frequency active sonar) at a given received level of sound. This effects analysis assumes that the potential consequences of 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 harassment 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 a significant behavioral response. The behavioral response function applied to mysticetes (Figure 11) differs from that used for odontocetes (Figure 12) 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.

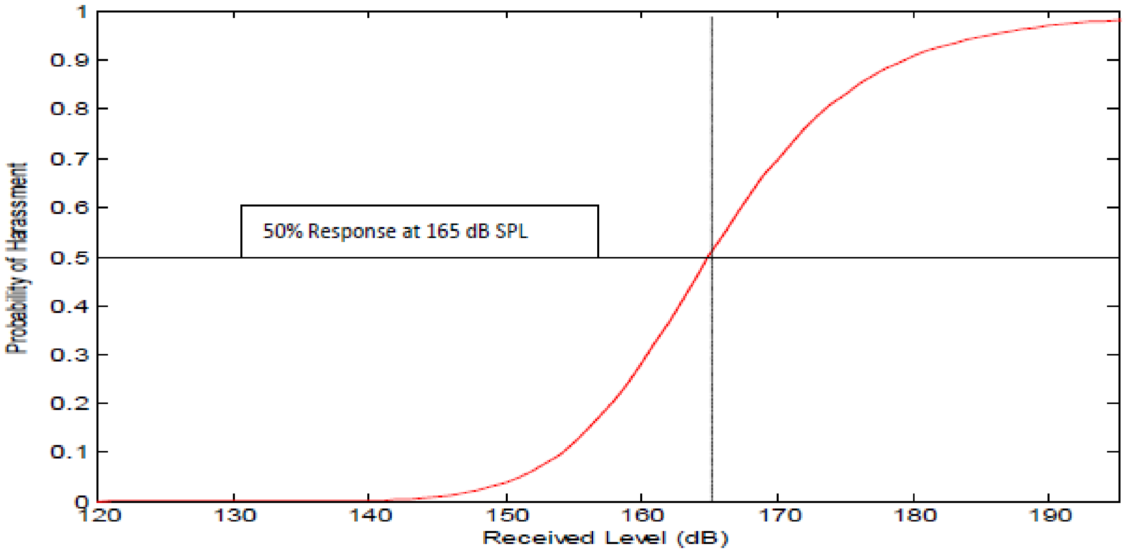

Figure 11. Behavioral response function applied to mysticetes.

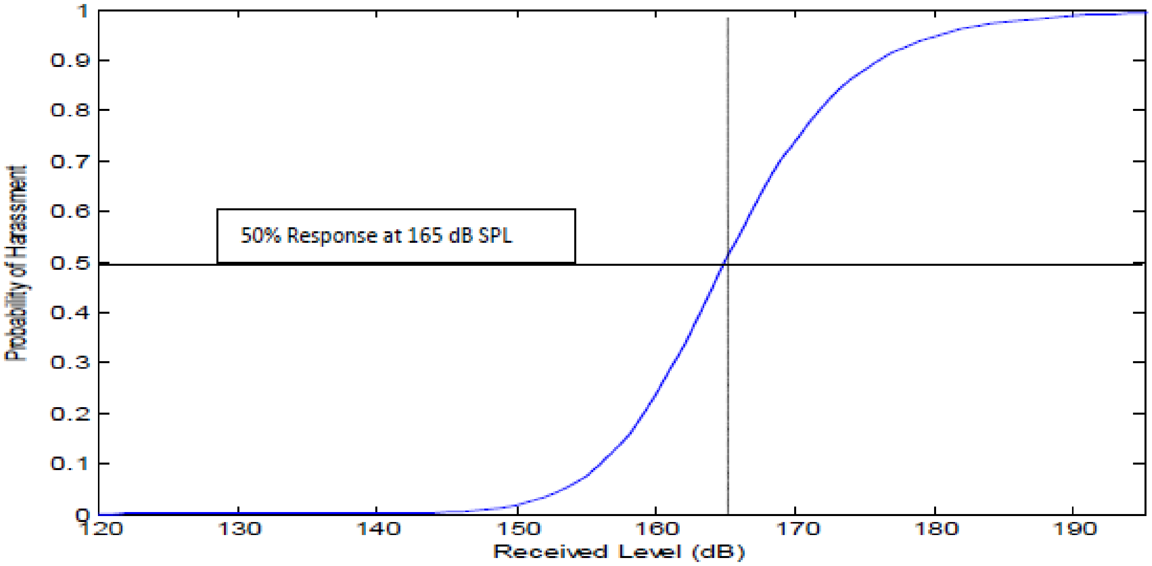

Figure 12. Behavioral response function applied to odontocetes and pinnipeds.

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 behavioral response function, are shown in Table 74 and Table 75 respectively.

Table 74. 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 Increment	Approximate Distance (m)	Behavioral Harassment % from SPL Increment	Approximate Distance (m)	Behavioral Harassment % from SPL Increment
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%

The range to 120 dB re 1 μPa varies by system, but can exceed 107 miles (172 km) for the most powerful hull mounted sonar; however, only a very small percentage of animals would be predicted to react at received levels between 120 and 130 dB re 1 μPa.

Table 75. Range to Received Sound Pressure Level (SPL) in 6-dB Increments and Percentage of Behavioral Harassments for Mid-Frequency and High Frequency Cetaceans 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 Increment**	**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 76. 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 76) are applied to the received sound level before being compared to the threshold.

Table 76. 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)
Phocid seals (underwater)	Hawaiian monk seal	SPL: BRF (Type I Weighting)	172 dB re 1μPa2 -s SEL (Type I Weighting)
Otariid seals (underwater)	Guadalupe fur seal	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 or testing activity, criteria are applied to predict the number of animals that may have a behavioral reaction. For events with multiple explosions, the behavioral threshold used in this analysis is 5 dB less than the TTS onset threshold (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.

6.4 Risk Assessment Framework - Sea Turtles

Direct injury to sea turtles from non-explosive sound sources, such as sonar, is unlikely because of relatively lower peak pressures and slower rise times than potentially injurious sources such as explosions. Nonexplosive sources also lack the strong shock waves that are associated with explosions. Therefore, primary blast injury and barotrauma would not result from exposure to non-impulsive sources such as sonar, and are only considered for explosive detonations.

The potential for trauma in sea turtles exposed to explosive sources has been inferred from tests of submerged terrestrial mammals exposed to underwater explosions (Ketten et al. 1993; Richmond et al. 1973; Yelverton et al. 1973). The effects of an underwater explosion on a sea turtle depend upon several factors, including size, type, and depth of both the animal and the explosive, depth of the water column, and distance from the charge to the animal. Smaller sea turtles would generally be more susceptible to injury. The compression of blast-sensitive, gas-containing organs when a sea turtle increases depth reduces likelihood of injury to these organs. The location of the explosion in the water column and the underwater environment determines whether most energy is released into the water or the air and influences the propagation of the blast wave.

6.4.1 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue impacts to sea turtles is primary blast injury and barotrauma after exposure to the shock waves of high-amplitude impulsive sources, such as explosions. Primary blast injuries result from the initial compression of a body exposed to the high pressure of a blast or shock wave. Primary blast injury is usually limited to gas-containing structures (e.g., lung and gut) and the pressure-sensitive components of the auditory system (discussed below) (Craig and Hearn 1998; SG 1991), although additional injuries could include concussive brain damage and cranial, skeletal, or shell fractures (Ketten 1995). 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 on the severity of the trauma. Rupture of the lung may introduce air into the vascular system, producing air blockages that can restrict oxygen delivery to the brain and heart. Although often secondary in life-threatening severity to pulmonary blast trauma, the gastrointestinal tract can also suffer bruising and tearing from blast exposure, particularly in air-containing regions of the tract. Potential traumas include internal bleeding, bowel perforation, tissue tears, and ruptures of the hollow abdominal organs. Although hemorrhage of solid organs (e.g., liver, spleen, and kidney) from blast exposure is possible, rupture of these organs is rarely encountered. Non-lethal injuries could increase a sea turtle's risk of predation, disease, or infection.

6.4.2 Auditory Trauma

Components of the auditory system that detect smaller or more gradual pressure changes can also be damaged when overloaded at high pressures with rapid rise times. Rupture of the tympanic membrane, while not necessarily a serious or life-threatening injury, may lead to permanent hearing loss (Ketten 1993; Ketten 1995). No data exist to correlate the sensitivity of the tympanic membrane and middle and inner ear to trauma from shock waves from underwater explosions (Viada et al. 2008).

The specific impacts of bulk cavitation (the collapse of air spaces created by explosive detonations) on sea turtles are unknown. The presence of a sea turtle within the cavitation region created by the detonation of small charges could annoy, injure, or increase the severity of the injuries caused by the shock wave. The area of cavitation from a large charge, such as those used in ship shock trials, is expected to result in mortality for smaller animals (Craig Jr. and Rye 2008). An animal located at (or near) the cavitation closure depth would be subjected to a short duration ("water hammer") pressure pulse; however, direct shock wave impacts alone would be expected to cause auditory system injuries and could cause internal organ injuries.

6.4.3 Hearing Loss

Hearing loss could effectively reduce the distance over which sea turtles can detect biologically relevant sounds. Both auditory trauma (a direct injury discussed above) and auditory fatigue may result in hearing loss, but the mechanisms responsible for auditory fatigue differ from auditory trauma. Hearing loss due to auditory fatigue is also known as threshold shift, a reduction in hearing sensitivity at certain frequencies. Threshold shift is the difference between hearing thresholds measured before and after an intense, fatiguing sound exposure. Threshold shift occurs when hair cells in the ear fatigue, causing them to become less sensitive over a small range of frequencies related to the sound source to which an animal was exposed. The actual amount of threshold shift depends on the amplitude, duration, frequency, and temporal pattern of the sound exposure. No studies are published on inducing threshold shift in sea turtles; therefore, the potential for the impact on sea turtles is inferred from studies of threshold shift in other animals.

Temporary threshold shift is a hearing loss that recovers to the original hearing threshold over a period. An animal may not even be aware of a TTS. It does not become deaf, but requires a louder sound stimulus (relative to the amount of TTS) to detect a sound within the affected frequencies. Temporary threshold shift may last several minutes to several days, depending on the intensity and duration of the sound exposure that induced the threshold shift (including multiple exposures).

Permanent threshold shift is a permanent hearing loss at a certain frequency range. Permanent threshold shift is non-recoverable due to the destruction of tissues within the auditory system. 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.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication. As a result, we do not expect instances of TTS and PTS to have fitness consequences for individual turtles.

6.4.4 Auditory Masking

Auditory masking occurs when a sound prevents or limits the distance over which an animal detects other biologically relevant sounds. When a noise has a sound level above the sound of interest, and in a similar frequency band, auditory masking could occur. Any sound above ambient noise levels and within an animal's hearing range could cause masking. The degree of masking increases with increasing noise levels; a noise that is just-detectable over ambient levels is unlikely to actually cause any substantial masking, whereas a louder noise may mask sounds over a wider frequency range. In addition, a continuous sound would have more potential for masking than a sound with a low duty cycle. In the open ocean, ambient noise levels are between about 60 and 80 dB re 1 μPa (NRC 2003a), especially at lower frequencies (below 100 Hz) and inshore, ambient noise levels, especially around busy ports, can exceed 120 dB re 1 μPa.

Unlike hearing loss, which likely results in a 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.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication. As a result, sound may play a limited role in a sea turtle's environment. Therefore, the potential for masking may be limited.

6.4.5 Physiological Stress

Sea turtles may exhibit a behavioral response or combinations of behavioral responses upon exposure to anthropogenic sounds. If a sound is detected, a stress response (i.e., startle or annoyance) or a cueing response (based on a past stressful experience) can occur. Sea turtles 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, nesting, and interactions with predators all contribute to stress. Anthropogenic activities could provide additional stressors above and beyond those that occur in the absence of human activity.

Immature Kemp's ridley sea turtles show physiological responses to the acute stress of capture and handling through increased levels of the stress hormone corticosterone, along with biting and rapid flipper movement (Gregory and Schmid 2001). Though it should be noted that Kemp's ridley sea turtles are not found in the NWTT Action Area, we would expect the turtles considered in this Opinion to have a similar physiological stress response. Captive olive ridley hatchlings showed heightened blood glucose levels indicating physiological stress (Zenteno et al. 2008). Repeated exposure to stressors, including human disturbance such as vessel disturbance and anthropogenic sound, may result in negative consequences to the health and viability of an individual or population (Gregory and Schmid 2001). Factors to consider when predicting a stress or cueing response is whether an animal is naïve or has prior experience with a stressor. Prior experience with a stressor may be of particular importance as repeated experience with a stressor may dull the stress response via acclimation (Hazel et al. 2007).

6.4.6 Behavioral Reactions

Little is known about the hearing ability of sea turtles and their response to acoustic disturbance and thus analogous species for which data are available are used to estimate the potential behavioral reactions to sound. The response of a sea turtle 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). Distance from the sound source and whether it is perceived as approaching or moving away could also affect the way a sea turtle responds. Potential behavioral responses to anthropogenic sound could include startle reactions, disruption of feeding, disruption of migration, changes in respiration, alteration of swim speed, alteration of swim direction, and area avoidance.

It is also possible that behavioral reactions could lead to negative physiological consequences. For example, Garcia-Parraga et al. (2014) reported evidence of decompression sickness (DCS; e.g., gas embolism) in sea turtles following capture in trawls or gillnets, with a higher incidence of DCS when caught in deeper waters. It is possible that a sea turtle could have an extreme behavioral avoidance reaction (e.g., surfacing too quickly in an attempt to avoid noise) that could lead to DCS-like symptoms and fitness consequences. However, it should be noted that this is the first, and to our knowledge, only study that has documented DCS-like symptoms in sea turtles. Previous research has suggested sea turtles are protected against DCS through anatomical, physiological, and behavioral adaptations (Berkson 1967; Castellini 2012; Fossette et al. 2010; Lutcavage and Lutz 1997; Piantadosi and Thalmann 2004). Given this uncertainty in the available literature and the lack of evidence that this sort of extreme behavioral avoidance reaction would be expected, we do not believe such a reaction is likely to occur and we do not consider DCS in sea turtles further in this Opinion.

6.4.6.1 ***Behavioral Reactions to Impulsive Sound Sources***

Studies of sea turtle responses to sounds are limited, though a few studies examined sea turtle reactions to airguns, which produce broadband impulse sound. O'Hara and Wilcox (1990) attempted to create a sound barrier at the end of a canal using seismic airguns. They reported that loggerhead turtles kept in a 984 ft by 148 ft (300 m by 45 m) enclosure in a 10 m deep canal maintained a distance of 98 ft (30 m) from airguns fired every 15 seconds, with the strongest sound components within the 25 Hz to 1,000 Hz frequency range. McCauley et al. (2000) estimated that the received level at which turtles avoided sound in the O'Hara and Wilcox (1990) experiment was 175 to 176 dB re 1 μPa root mean square.

Moein Bartol et al. (1995) investigated the use of air guns to repel juvenile loggerhead sea turtles from hopper dredges. Sound frequencies of the airguns ranged from 100 Hz to 1,000 Hz at three levels: 175, 177, and 179 dB re 1 μPa at 1 m. The turtles avoided the airguns during the initial exposures (mean range of 24 m), but additional trials several days afterward did not elicit statistically significant avoidance. They concluded that this was due to either habituation or a temporary shift in the turtles' hearing capability.

McCauley et al. (2000) exposed caged green and loggerhead sea turtles to an approaching-departing single air gun to gauge behavioral responses. The trials showed that above a received level of 166 dB re 1 μPa root mean square, the turtles noticeably increased their swimming activity compared to non-operational periods, with swimming time increasing as air gun levels increased during approach. Above 175 dB re 1 μPa root mean square, behavior became more erratic, possibly indicating the turtles were in an agitated state (McCauley et al. 2000). The authors noted that the point at which the turtles showed erratic behavior and exhibited possible agitation would be expected to approximately equal the point at which active avoidance would occur for unrestrained turtles (McCauley et al. 2000).

No obvious avoidance reactions by free-ranging sea turtles, such as swimming away, were observed during a multi-month seismic survey using airgun arrays, although fewer sea turtles were observed when the seismic airguns were active than when they were inactive (Weir 2007). The author noted that sea state and the time of day affected both airgun operations and sea turtle surface basking behavior, making it difficult to draw conclusions from the data. Further, DeRuiter and Larbi Doukara (2012) noted diving behavior following airgun shots in loggerhead turtles, and noted a decreased dive probability with increasing distance from the airgun array.

6.4.6.2 *Behavioral Reactions to Non-Impulsive Sources*

No studies have been performed to examine the response of sea turtles to sonar. However, based on the limited range of hearing, they may respond to sources operating below 2 kHz but are unlikely to sense higher frequency sounds.

6.4.6.3 *Behavioral Reactions to Vessels*

Vessel noise and visual stimuli (vessels and shadows) could disturb sea turtles, and potentially elicit a startle response, avoidance, or other behavioral reaction. Sea turtles are frequently exposed to research, ecotourism, commercial, government, and private vessel traffic. Some sea turtles may habituate to vessel noise, and may be more likely to respond to the sight of a vessel rather than the sound of a vessel, although both may play a role in prompting reactions (Hazel et al. 2007).

6.4.6.4 *Behavioral Reactions to Aircraft and Missile Overflight*

Sea turtles may respond to both the physical presence (visual effects of the aircraft and shadows) and to the noise generated by aircraft. Helicopters may produce strong downdrafts, a vertical flow of air that becomes a surface wind, which can also affect an animal's behavior at or near the surface. In most cases, exposure of a sea turtle to fixed-wing or rotary-wing aircraft would last for only seconds as the aircraft quickly passes overhead. Animals would have to be at or near the surface at the time of an overflight to be exposed to appreciable sound levels or visual stimuli.

6.4.7 Repeated Exposures of Sea Turtles

Navy sonar systems are generally deployed from highly mobile vessels or in-water devices which do not directly target sea turtles. The typical duty cycle with most tactical anti-submarine warfare is 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 sea turtles), we would not expect repeated exposures.

Establishing a causal link between anthropogenic noise 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). 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).

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication. Therefore, repeated interruptions of a sea turtle's normal activity due to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population. To result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for lost feeding opportunities 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 foraging habitat would still be available in the environment following the cessation of acoustic exposure.

Sea turtles may habituate to, or become tolerant of, repeated exposures over time, such as ambient noise found in areas of high vessel traffic (Hazel et al. 2007). After initial avoidance reactions, loggerhead sea turtles habituated to repeated experimental exposures to airguns of up to a source level of 179 dB re 1 μPa in an enclosure. The habituation behavior was retained by the sea turtles when exposures were separated by several days (Moein Bartol et al. 1995). Individual sea turtles 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 and wide ranging life histories of the species considered in this Opinion, we do not believe this will result from Navy training and testing activities in the NWTT Action Area.

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. 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 120 dB). Given sea turtles' 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. As a result of being capable of moving outside the range of behavioral responses from acoustic stressors, being physiologically less susceptible to acoustic stressors, and relying on senses other than hearing for important biological behaviors; we do not expect any fitness consequences from any individual animals nor do we expect any population level effects from behavioral responses.

6.4.8 Criteria for Predicting Acoustic and Explosive Impacts – Sea Turtles

In this Opinion, we consider two primary categories of sound sources that the Navy used in its analyses of sound impacts on sea turtles: impulsive sources (e.g., explosives, airguns, weapons firing) and non-impulsive sources (e.g., sonar, pingers, and countermeasure devices). The Navy, in cooperation with NMFS, developed acoustic impacts criteria and thresholds for sea turtle exposures to various sound sources.

6.4.8.1 *Frequency Weighting*

Animals generally do not hear equally well across their entire hearing range. Several studies using green, loggerhead, and Kemp's ridley turtles suggest sea turtles are most sensitive to low-frequency sounds, although this sensitivity varies slightly by species and age (Bartol et al. 1999a; Bartol and Ketten 2006; Lenhardt et al. 1994b; Ridgeway et al. 1969). Sea turtles possess an overall hearing range of approximately 100 Hz to 1 kHz, with an upper limit of 2 kHz (Bartol et al. 1999a; Bartol and Ketten 2006; Lenhardt et al. 1994b; Ridgeway et al. 1969). Because hearing thresholds are frequency-dependent, an auditory weighting function was developed for sea turtles (turtle-weighting, or T-weighting). The T-weighting function simply defines lower and upper frequency boundaries beyond which sea turtle hearing sensitivity decreases. The single frequency cutoffs at each end of the frequency range where hearing sensitivity begins to decrease are based on the most liberal interpretations of sea turtle hearing abilities (10 Hz and 2 kHz). These boundaries are precautionary and exceed the demonstrated or anatomy-based hypothetical upper and lower limits of sea turtle hearing. The T-weighting function adjusts the received sound level, based on sensitivity to different frequencies, emphasizing frequencies to which sea turtles are most sensitive and reducing emphasis on frequencies outside of their estimated useful range of hearing. For example, a 160 dB re 1 μPa tone at 10 kHz, far outside sea turtle best range of hearing, is estimated to be perceived by a sea turtle as a 130 dB re 1 μPa sound (i.e., 30 dB lower). Stated another way, a sound outside of the range of best hearing would have to be more intense to have the same impact as a sound within the range of best hearing.

6.4.8.2 ***Hearing Loss – Temporary and Permanent Threshold Shift***

Whereas TTS represents a temporary reduction of hearing sensitivity, PTS represents tissue damage that does not recover and permanent reduced sensitivity to sounds over specific frequency ranges. To date, no known data are available on potential hearing impairments (i.e., TTS and PTS) in sea turtles. Sea turtles, based on their auditory anatomy (Lenhardt et al. 1985; Moein Bartol and Musick 2003; Wartzok and Ketten 1999; Wever 1978; Wyneken 2001), almost certainly have poorer absolute sensitivity (i.e., higher thresholds) across much of their hearing range than do the mid-frequency cetacean species. Therefore, applying TTS and PTS criteria derived from mid-frequency cetaceans to sea turtles provides a protective approach to estimating acoustic impacts to sea turtles (PTS and TTS data are not available for low-frequency cetaceans). Criteria for hearing loss due to onset of TTS and PTS are based on sound exposure level (for non-impulsive and impulsive sources) and peak pressure (for impulsive sources only).

To determine the sound exposure level, the turtle weighting function is applied to the acoustic exposure to emphasize only those frequencies within a sea turtle's hearing range. Multiple exposures within any 24-hour period are considered one continuous exposure for the purposes of calculating the received sound exposure level for a given individual. This conservatively assumes no recovery of hearing between exposures during a 24-hour period. The weighted sound exposure level is then compared to weighted threshold values for TTS and PTS. If the weighted exposure level meets or exceeds the weighted threshold, then the physiological impact (TTS or PTS) is assumed to occur. For impacts from exposures to impulsive sources, the metric (peak pressure or sound exposure level) and threshold level that results in the longest range to impact is used to predict impacts. Exposures are not calculated for sound sources with a nominal frequency outside the upper and lower frequency hearing limits for sea turtles.

In addition to being discussed below, thresholds for onset of TTS and PTS for impulsive and non-impulsive sounds are summarized in 6.4.8.2.2, 6.4.8.2.1, 6.4.8.2.4, and 6.4.8.2.3, respectively. As described in more detail below, it is critical to note that all turtle PTS and TTS thresholds were set too high. Therefore, quantitative impacts presented herein for PTS and TTS are conservative estimates.

Table 77. Sea Turtle Impact Threshold Criteria for Impulsive Sources

Impulsive Sound Exposure Impact	Threshold Value
Onset Mortality[1] (1% Mortality Based on Extensive Lung Injury)	$= 91.4M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right){}^{1}/_{2}\ Pa-s$
Onset Slight Lung Injury[1]	$= 39.1M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right){}^{1}/_{2}\ Pa-s$
Onset Slight Gastrointestinal Tract Injury	237 dB re 1 µPa SPL (104 psi)
Onset PTS	187 dB re 1 µPa^2 - s SEL (T^2) or 230 dB re 1 µPa Peak SPL
Onset TTS	172 dB re 1 µPa^2 - s SEL (T^2) or 224 dB re 1 µPa Peak SPL
Injury (Airguns)	190 dB re 1 µPa SPL root mean square[3]

dB: decibels, µPa: micropascals, PTS: permanent threshold shift, SEL: sound exposure level, SPL: sound pressure level, TTS: temporary threshold shift

[1] M=Mass of animals (kg) as shown for each species, DRm=depth of animal (m)

[2] (T): Turtle weighting function

[3] The time interval for determining the root mean square that which contains 90 percent of the total energy within the envelope of the pulse. This windowing procedure for impulse signals removes uncertainty about where to set the exact temporal beginning or end of the signal, which may be obscured by ambient noise.

Table 78. Sea Turtle Impact Threshold Criteria Used in Acoustic Modeling for Non-Impulse Sources

Physiological Thresholds	
Onset PTS	Onset TTS
198 dB SEL (T)	178 dB SEL (T)

dB: decibels; µPa: micropascals; PTS: permanent threshold shift; SEL: sound exposure level; SPL: sound pressure level; TTS: temporary threshold shift; (T): Turtle weighting function

Table 79 shows the average ranges to the potential effect from in-water explosions based on the thresholds for sea turtles. Some of the conservative assumptions made by the Navy for the impact modeling and criteria may cause the impact predictions to be overestimated, as follows:

- Many explosions from ordnance such as bombs and missiles actually explode upon impact with above-water targets. For this analysis, sources such as these were modeled as exploding at depths of 1 m, overestimating the amount of explosive and acoustic energy entering the water.

- For predicting TTS and PTS based on sound exposure level, the duration of an explosion is assumed to be 1 second. Actual detonation durations may be much shorter, so the actual sound exposure level at a particular distance may be lower.

- Mortality and slight lung injury criteria are based on juvenile turtle masses, which substantially increases that range to which these impacts are predicted to occur compared to the ranges that would be predicted using adult turtle masses.

- Animats are assumed to receive the full impulse of the initial positive pressure wave due to an explosion, although the impulse-based thresholds (onset mortality and onset slight lung injury)

Table 79. Range to impacts from In-Water Explosives on Sea Turtles from Representative Sources

Criteria Predicted Impact	**Impact Predicted to Occur When Sea Turtle is at this Range (m) or Closer to a Detonation**							
	Bin E-1 (0.0-0.5 lb. NEW)	**Bin E-3 (0.6-2.6 lb. NEW)**	**Bin E-4 (2.6-6.0 lb. NEW)**	**Bin E-5 (6.0-10.0 lb. NEW)**	**Bin E-8 (21.0-60.0 lb. NEW)**	**Bin E-10 (251-500 lb. NEW)**	**Bin E-11 (501-1,000 lb. NEW)**	**Bin E-12 (1000-1,651 lb. NEW)**
Onset Mortality (1% Mortality)	4	26	51	46	102	164	458	199
Onset Slight Lung Injury	17	50	130	85	179	284	816	343
Onset Slight GI Tract Injury	40	60	175	55	106	184	201	250
Permanent Threshold Shift[1]	67	196	215	162	424	873	809	1,251
Temporary Threshold Shift[1]	90	724	421	288	844	1,975	1,693	2,640
Behavioral Response	144	1,512	796	565	1,458	3,217	3,015	3,962

[1] Modeling for sound exposure level-based impulse criteria assumed explosive event durations of 1 second. Actual durations may be less, resulting in smaller ranges to impact.

Notes: (1) lb. = pound(s), m = meters, NEW = net explosive weight; (2) Ranges determined using REFMS, the Navy's explosive propagation model.

6.4.8.2.1 Temporary Threshold Shift – Non-Impulsive Sources

Based on best available science regarding TTS in marine vertebrates (Finneran et al. 2002; Southall et al. 2007c) and the lack of information regarding TTS in sea turtles, the total T-weighted sound exposure level of 178 dB re 1 micro Pascal squared second (μPa^2 -s) is used to estimate exposures resulting in TTS for sea turtles. The T-weighting function is used in conjunction with this non-impulsive criterion, which effectively provides an upper cutoff of 2 kHz. The T-weighted non-impulsive TTS threshold of 178 dB re 1 μPa^2 -s sound exposure level was inadvertently based on Type II weighted cetacean TTS data rather than Type I weighted cetacean TTS data. This resulted in incorrectly lowering the turtle TTS threshold by 17 dB. The sea turtle non-impulsive TTS threshold, based on mid-frequency cetacean data, should be 17 dB higher than 178 dB re 1 μPa^2 -s. Because an incorrectly lowered threshold was used to quantitatively analyze acoustic impacts on sea turtles, the quantitative impacts presented herein for non-impulsive TTS are conservative (i.e., over-predicted).

6.4.8.2.2 Temporary Threshold Shift – Impulsive Sources

Based on best available science regarding TTS in marine vertebrates (Finneran et al. 2005b; Finneran et al. 2000b; Finneran et al. 2002; Nachtigall et al. 2003; Nachtigall et al. 2004; Schlundt et al. 2000a) and the lack of information regarding TTS in sea turtles, the respective total T-weighted sound exposure level of 172 dB re 1 μPa^2-s or peak pressure of 224 dB re 1 µPa (23 pounds per square inch [psi]) is used to estimate exposures resulting in TTS for sea turtles. The T-weighting function is applied when using the sound exposure level-based thresholds to predict TTS.

The T-weighted impulsive TTS threshold of 172 dB re 1 μPa^2-s sound exposure level was inadvertently based on Type II weighted cetacean TTS data rather than Type I weighted cetacean TTS data. This resulted in incorrectly lowering the turtle TTS threshold. The sea turtle impulsive TTS threshold, based on Type I mid-frequency cetacean data, should be 183 dB re 1 μPa^2-s. Because an incorrectly lowered threshold was used to quantitatively analyze acoustic impacts to sea turtles, the quantitative impacts presented herein for impulsive TTS are conservative (i.e., over-predicted).

6.4.8.2.3 Permanent Threshold Shift – Non-Impulsive Sources

Since no studies were designed to intentionally induce PTS in sea turtles, levels for onset of PTS for these animals must be estimated using TTS data and relationships between TTS and PTS established in terrestrial mammals. Permanent threshold shift can be estimated based on the growth rate of a threshold shift and the level of threshold shift required to potentially become non-recoverable. A variety of terrestrial and marine mammal data show that threshold shifts up to 40 to 50 dB may be recoverable, and that 40 dB is a reasonable upper limit of a threshold shift that does not induce PTS (Southall et al. 2007b; Ward et al. 1958; Ward et al. 1959a). This analysis assumes that continuous-type exposures producing threshold shifts of 40 dB or more always result in some amount of PTS. Data from terrestrial mammal testing (Ward et al. 1958; Ward et al. 1959a) show TTS growth of 1.5 to 1.6 dB for every 1 dB increase in sound exposure level. The difference between minimum measureable TTS onset (6 dB) and the 40 dB upper safe limit of TTS yields a difference of 34 dB. When divided by a TTS growth rate of 1.6 dB TTS per dB sound exposure level, there is an indication that an increase in exposure of a 21.25 dB sound exposure level would result in 40 dB of TTS. For simplicity and conservatism, the number was rounded down to 20 dB sound exposure level. Therefore, non-impulsive exposures of 20 dB sound exposure level above those producing a TTS may be assumed to produce a PTS. The onset of TTS threshold of 195 dB re 1 μPa^2-s for sea turtles has a corresponding onset of PTS threshold of 198 dB re 1 μPa^2-s. The T-weighting function is applied when using the sound exposure level-based thresholds to predict PTS.

The T-weighted non-impulsive TTS threshold of 178 dB re 1 μPa^2-s sound exposure level was inadvertently based on Type II weighted cetacean TTS data rather than Type I weighted cetacean TTS data. This resulted in incorrectly lowering the turtle TTS threshold by 17 dB; consequently, also incorrectly lowering the sea turtle PTS threshold by 17 dB. The sea turtle non-impulsive PTS threshold, based on mid-frequency cetacean data, should be 17 dB higher than 198 dB re 1 μPa^2-s. Because an incorrectly lowered threshold was used to quantitatively analyze acoustic impacts to sea turtles, the quantitative impacts presented herein for non-impulsive PTS are conservative (i.e., overpredicted).

6.4.8.2.4 Permanent Threshold Shift – Impulsive sources

Because marine mammal and sea turtle PTS data from impulsive exposures do not exist, onset of PTS levels for these animals are estimated by adding 15 dB to the sound exposure level-based TTS threshold and 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. This results in onset of PTS thresholds of total weighted sound exposure level of 187 dB re 1 μPa^2-s or peak pressure of 230 dB re 1 μPa for sea turtles. The T-weighting function is applied when using the sound exposure level-based thresholds to predict PTS.

The T-weighted impulsive PTS threshold of 187 dB re 1 μPa^2-s sound exposure level was inadvertently based on Type II weighted cetacean TTS data rather than Type I weighted cetacean TTS data. This resulted in incorrectly lowering the turtle TTS threshold. The sea turtle impulsive PTS threshold, based on Type I mid-frequency cetacean data, should be 198 dB re 1 μPa^2-s. Because an incorrectly lowered threshold was used to quantitatively analyze acoustic impacts to sea turtles, the quantitative impacts presented herein for impulsive PTS are conservative (i.e., over-predicted).

6.4.8.3 *Mortality and Injury from Explosions*

There is a considerable body of laboratory data on actual injuries from impulsive sounds, usually from explosive pulses, obtained from tests with a variety of vertebrate species (e.g., (Goertner et al. 1994; Richmond et al. 1973; Yelverton et al. 1973)). Based on these studies, potential impacts, with decreasing likelihood of serious injury or lethality, include onset of mortality, onset of slight lung injury, and onset of slight gastrointestinal injury. In the absence of data specific to sea turtles, criteria developed to assess impacts to protected marine mammals are also used to assess impacts to protected sea turtles. These criteria are discussed below.

6.4.8.3.1 Criteria for Mortality and Slight Lung Injury

In air or submerged, the most commonly reported internal bodily injury to sea turtles from explosive detonations is hemorrhaging in the fine structure of the lungs. The likelihood of internal bodily injury is related to the received 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 is used as a metric upon which internal organ injury can be predicted. Onset mortality and onset slight lung injury are defined as the impulse level that would result in one percent mortality (most survivors have moderate blast injuries and should survive) and zero percent mortality (recoverable, slight blast injuries) in the exposed population, respectively. Criteria for onset mortality and onset slight lung injury were developed using data from explosive impacts on mammals (Yelverton and Richmond 1981).

The impulse required to cause lung damage is related to the volume of the lungs. The lung volume is related to both the size (mass) of the animal and compression of gas-filled spaces at increasing water depth. Turtles have relatively low lung volume to body mass and a relatively stronger anatomical structure compared to mammals; therefore application of the criteria derived from studies of impacts of explosions on mammals may be conservative. Table 80 provides a conservative body mass for each sea turtle species based on juvenile mass.

Juvenile body mass was selected for analysis given the early rapid growth of these reptiles (newborn turtles weigh less than 0.5 percent of maximum adult body mass). In addition, small turtles tend to remain at shallow depths in the surface pressure release zone, reducing potential exposure to injurious impulses. Therefore, use of hatchling weight would provide unrealistically low thresholds for estimating injury to sea turtles. The use of juvenile body mass rather than adult body mass was chosen to produce reasonably conservative estimates of injury.

Table 80. Species-Specific Sea Turtle Masses for Determining Onset of Extensive and Slight Lung Injury Thresholds

Common Name	Juvenile Mass (kg)	Reference
Loggerhead sea turtle	8.4	Southwood et al. (2007)
Green sea turtle	8.7	Wood and Wood (1993)
Hawksbill sea turtle	7.4	Okuyama et al. (2010)
Olive ridley sea turtle	6.3	McVey and Wibbels (1984) and Caillouet et al. (1986)
Leatherback sea turtle	34.8	Jones (2009)

The scaling of lung volume to depth is conducted because data come from experiments with terrestrial animals held near the water's surface. The calculation of impulse thresholds consider depth of the animal to account for compression of gas-filled spaces that are most sensitive to impulse injury. The impulse required for a specific level of injury (impulse tolerance) is assumed to increase proportionally to the square root of the ratio of the combined atmospheric and hydrostatic pressures at a specific depth with the atmospheric pressure at the (Goertner 1982).

Very little information exists about the impacts of underwater detonations on sea turtles. Impacts of explosive removal operations on sea turtles range from non-injurious impacts (e.g., acoustic annoyance, mild tactile detection, or physical discomfort) to varying levels of injury (i.e., non-lethal and lethal injuries) (Klima et al. 1988; Viada et al. 2008). Often, impacts of explosive events on turtles must be inferred from documented impacts on other vertebrates with lungs or other-gas containing organs, such as mammals and most fishes (Viada et al. 2008). The methods used by Goertner (1982) to develop lung injury criteria for marine mammals may not be directly applicable to sea turtles, as it is not known what degree of protection to internal organs from the shock waves is provided to sea turtles by their shell (Viada et al. 2008). However, the general principles of the Goertner model are applicable, and should provide a protective approach to assessing potential impacts on sea turtles. The Goertner method predicts a minimum primary positive impulse value for onset of slight lung injury and onset of mortality, adjusted for assumed lung volume (correlated to animal mass) and depth of the animal. These equations are shown in Table 77.

6.4.8.3.2 Criteria for Onset of Gastrointestinal Tract Injury

Without data specific to sea turtles, data from tests with terrestrial animals are used to predict onset of gastrointestinal tract injury. Gas-containing internal organs, such as lungs and intestines, were the principle damage sites from shock waves in submerged terrestrial mammals (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). Slight contusions to the gastrointestinal tract were reported during small charge tests (Richmond et al. 1973), when the peak was 237 dB re 1 μPa. Therefore, this value is used to predict onset of gastrointestinal tract injury in sea turtles exposed to explosions.

6.4.8.4 ***Criteria for Behavioral Reactions***

A sea turtle's behavioral responses to sound are assumed to be variable and context specific. For instance, a single impulse may cause a brief startle reaction. A sea turtle may swim farther away from the sound source, increase swimming speed, change surfacing time, and decrease foraging if the stressor continues to occur. For each potential behavioral change, the magnitude of the change ultimately would determine the severity of the response; most responses would be short-term avoidance reactions.

A few studies reviewed in Section 6.4.6, investigated behavioral responses of sea turtles to impulsive sounds emitted by airguns (McCauley et al. 2000; Moein Bartol et al. 1995; O'Hara and Wilcox 1990). There are no studies of sea turtle behavioral responses to sonar. Cumulatively, available airgun studies indicate that perception and a behavioral reaction to a repeated sound may occur with sound pressure levels greater than 166 dB re 1 μPa root mean square, and that more erratic behavior and avoidance may occur at higher thresholds around 175 to 179 dB re 1 μPa root mean square (McCauley et al. 2000; Moein Bartol et al. 1995; O'Hara and Wilcox 1990). When exposed to impulsive acoustic energy from an airgun above 175 dB re 1 μPa root mean square, sea turtle behavior becomes more erratic, possibly indicating the turtles were in an agitated state (McCauley et al. 2000). A received level of 175 dB re 1 μPa root mean square is more likely to be the point at which avoidance may occur in unrestrained turtles, with a comparable sound exposure level of 160 dB re 1 μPa^2 -s (McCauley et al. 2000). Airgun studies used sources that fired repeatedly over some duration. For single impulses at received levels below threshold shift (hearing loss) levels, the most likely behavioral response is assumed to be a startle response. Since no further sounds follow the initial brief impulse, the biological significance is considered to be minimal.

Behavioral responses of sea turtles to airgun exposures in caged enclosures are likely to be different than those from turtles exposed to impulsive acoustic sources from NWTT activities in the open environment. Although information regarding the behavioral response of sea turtles to acoustic stressors is generally lacking, McCauley et al. (2000) provides an indication that 175 dB re 1 μPa root mean square is a reasonable threshold criterion in the absence of more rigorous experimental or observational data. The 175 dB re 1 μPa root mean square threshold criterion for behavioral take in sea turtles may change with better available information in the future, but currently is the best available science. To assess the number of sea turtles expected to behaviorally respond to acoustic stress all turtles exposed to sound equal to, or greater than, 175 dB and less than the criterion for TTS were summed. No attempt to process these exposures or evaluate the effectiveness of mitigation measures was made, suggesting any behavioral take estimates of sea turtles from acoustic stressors are likely overestimates. We are unaware of any sea turtle response studies to non-impulsive acoustic energy; therefore, we used the same criteria as those for impulsive acoustic stressors.

6.5 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. 2014).

6.5.1 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; Løvik 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. 1996a).

Popper et al. (2014) 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.

6.5.2 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. 2014).

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. (2014) 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.

6.5.3 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.

6.5.3.1 ***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. 2007). 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 2002b; 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).

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. (1996a) 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. (2014) estimate exposure to low frequency sonar > 193 $\mathrm{dB_{rms}}$ 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.

6.5.3.2 ***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. (2014) 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 PTS, 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.

6.5.4 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. (2014) 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. 2014).

6.5.5 Physiological Stress

As with masking, a fish must first be able to detect a sound above its hearing threshold for that particular frequency and the ambient noise before a behavioral reaction or physiological stress can occur.

Stress refers to biochemical and physiological responses to increases in background sound. The initial response to an acute stimulus is a rapid release of stress hormones into the circulatory system, which may cause other responses such as elevated heart rate and blood chemistry changes. Although an increase in background sound has been shown to cause stress in humans, only a limited number of studies have measured biochemical responses by fish to acoustic stress (Remage-Healey et al. 2006; Smith et al. 2004b; Wysocki et al. 2007; Wysocki et al. 2006) and the results have varied. There is evidence that a sudden increase in sound pressure level or an increase in background noise levels can increase stress levels in fish (Popper and Hastings 2009a; Popper and Hastings 2009b). Exposure to acoustic energy has been shown to cause a change in hormone levels (physiological stress) and altered behavior in some species such as the goldfish *(Carassius auratus)* (Pickering 1981; Smith et al. 2004a; Smith et al. 2004b), but not all species tested to date, such as the rainbow trout *(Oncorhynchus mykiss*) (Wysocki et al. 2007).

Remage-Healey et al. (2006) found elevated cortisol levels, a stress hormone, in Gulf toadfish exposed to low frequency bottlenose dolphin sounds. Additionally, the toadfish' call rates dropped by about 50 percent, presumably because the calls of the toadfish, a primary prey for bottlenose dolphins, give away the fish's location to the dolphin. The researchers observed none of these effects in toadfish exposed to an ambient control sound (i.e., low-frequency snapping shrimp 'pops').

Smith et al. (2004b) found no increase in corticosteroid, a stress hormone, in goldfish exposed to a continuous, band-limited noise (0.1 to 10 kHz) with a sound pressure level of 170 dB re 1 μPa for 1 month. Wysocki et al. (2007) exposed rainbow trout to continuous band-limited noise with a sound pressure level of about 150 dB re 1 μPa for 9 months with no observed stress effects. Growth rates and effects on the trout's immune system were not significantly different from control animals held at sound pressure level of 110 dB re 1 μPa.

6.5.6 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).

6.5.6.1 ***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.

Popper et al. (2014) 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. 2014).

6.5.6.2 ***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. (2014) 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.

6.5.7 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). 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). Within the ocean environment, aggregate anthropogenic impacts have to be considered in context of natural variation and climate 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).

Most of the fish that may be exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training and testing in the NWTT Action Area. The majority of the proposed training and testing activities occur over a small spatial scale relative to the entire Action Area, have few participants, and are of a short duration (the order of a few hours or less). These periodic or episodic exposure and response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. For example, Navy sonar systems are generally deployed from highly mobile vessels or in-water devices which do not directly target ESA-listed resources. 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) (Navy 2013). 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), successive events are not done in the same location. Additionally, due to the duty cycle, duration of active transmission in a specific location, and mobility of the ESA-listed fish species considered in this Opinion, we would not expect repeated exposures.

Repeated exposure to impulsive acoustic stressors may be more likely as successive detonations could 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 the effect of repeated exposures under this scenario to not impact the fitness of individual fish. As suggested by Popper et al. (2014), 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. It also may be possible for detonations to temporarily affect refugia or foraging habitat selection. However, given these animal's mobility, we would expect individuals to temporarily select alternative refuge or forage sites nearby until the exposure levels in their initially selected location have decreased. To result in significant fitness consequences, we would have to assume that an individual fish could not compensate for lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Similarly, we would have to assume individuals could not find alternative refuge habitat nearby. There is no indication this is the case. Therefore, even temporary displacement from initially selected habitat is not expected to impact the fitness of any individual animals because we would expect equivalent habitat to be available in close proximity. 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). 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 temporary hearing loss or 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.

6.5.8 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 for this consultation. For additional information on the methodology used to develop these criteria, see Renken (2015).

6.5.8.1 ***Non-impulsive Acoustic Sources (Sonar)***

As we discussed in Section 6.5.1 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 81) 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 82 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 81. 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. (2014) 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) or fish that do not have a swim bladder (e.g., eulachon).

[3] Cannot Hear

Table 82. Predicted Range to Effect for Sonar Source bins used in NWTT. (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
LF4	0	0	0	0	0	2	0	0	2	9
LF5	0	0		0	0	0	0	0	0	1
ASW2	0	0		0	0	1	0	0	1	6
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
MF8	<<15	<15	CH	<<15	<15	CH	<<15	<15	17	361
MF9	0	0	CH	0	0	CH	0	0	0	9
MF10	0	0	CH	0	0	CH	0	0	0	0
MF11	<<6	<6	CH	<<6	<6	CH	<<6	<6	7	69
MF12	<<5	<5	CH	<<5	<5	CH	<<5	<5	5	52
ASW4	<<1	<1	CH	<<1	<1	CH	<<1	<1	1	15
M3	0	0	CH	0	0	CH	0	0	0	4

-CH = cannot hear

-MF5, MF6, and MF10 pose no risk to fish according to this assessment. Also, recall that 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.

6.5.8.2 *Explosives*

The use of explosives in the NWTT Study Area 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. 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[18].

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[18] Eulachon do not possess swim bladders and so are far less susceptible to injury and mortality from explosions than fish with swim bladders. The range from an explosive event over which damage may occur to fish without swim bladders is on the order of 100 times less than that for a fish with a swim bladder (Goertner 1978). Therefore, the range to effects values for eulachon were calculated using the same methodology as was used for the other fish species considered in this Opinion, and then each range to effect value was divided by 100 to account for the fact that eulachon do not possess a swim bladder.

Table 83. Ranges to Effects to Fish from Explosives. Ranges to effects are presented in meters.

Explosive BIN	Representative depth of charge[1]		Puget Sound Rockfish		Chum			Chinook			Coho		Sockeye		Steelhead		Eulachon
		Life Stage	Larvae	Juveniles	Juveniles in Puget Sound	Juveniles in offshore	Adult	Juveniles in Puget Sound	Juveniles in offshore	Adult	Juveniles	Adult	Juveniles	Adult	Juveniles	Adult	Adult
		Weight (g)	0.005	0.372	0.38	3.867	398.7	8.186	11.43	3019.7	14.91	1468.3	15.088	440.33	30.626	354.7	37
SWAG[3]		1% Mort	102	59	61	NA[2]	27	41	NA	22	NA	NA	NA	NA	36	27	NA
		Onset injury	184	118	131	NA	69	95	NA	62	NA	NA	NA	NA	83	69	NA
E1		1% Mort	NA	NA	NA	109	61	NA	96	50	93	54	93	60	84	62	0.82
		Onset injury	NA	NA	NA	214	140	NA	205	120	199	127	199	139	184	142	1.82
E3 off shore	30	1% Mort	NA	NA	NA	245	140	NA	214	109	209	119	208	139	193	143	1.88
		Onset injury	NA	NA	NA	445	307	NA	403	248	394	262	394	303	377	311	3.69
E3 inland	8	1% Mort	406	234	233	NA	99	158	NA	79	NA	NA	NA	NA	135	101	NA
		Onset injury	661	385	384	NA	179	268	NA	153	NA	NA	NA	NA	231	181	NA
E4	20	1% Mort	NA	NA	NA	262	152	NA	229	119	221	129	221	150	203	154	1.98
		Onset injury	NA	NA	NA	447	283	NA	396	244	384	247	383	279	356	287	3.51
E5	1	1% Mort	NA	NA	NA	160	89	NA	140	71	136	75	136	88	124	90	1.21
		Onset	NA	NA	NA	263	155	NA	231	133	228	137	228	153	208	157	2.03

Explosive BIN	Representative depth of charge[1]		Puget Sound Rockfish		Chum			Chinook			Coho		Sockeye		Steelhead		Eulachon
		Life Stage	Larvae	Juveniles	Juveniles in Puget Sound	Juveniles in offshore	Adult	Juveniles in Puget Sound	Juveniles in offshore	Adult	Juveniles	Adult	Juveniles	Adult	Juveniles	Adult	Adult
		Weight (g)	0.005	0.372	0.38	3.867	398.7	8.186	11.43	3019.7	14.91	1468.3	15.088	440.33	30.626	354.7	37
		injury															
E8	35	1% Mort	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	514	380	NA
		Onset injury	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	849	636	NA
E10	1	1% Mort	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	362	263	NA
		Onset injury	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	582	426	NA
E11	35	1% Mort	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	860	631	NA
		Onset injury	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	1404	1032	NA
E12	1	1% Mort	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	435	319	NA
		Onset injury	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	703	514	NA

[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.

[3] Initial range to effects values for SWAG charges were based on Yelverton et al. (1975). However, when comparing these modeled values to measured data from similar explosions in Puget Sound, it was determined that the modeled values likely over-estimated the range to injury and mortality for this explosive bin by a factor of 0.6 (Martin Renken, personal communication, October 20, 2015). Therefore, we multiplied modeled range to effects values estimated for each species by a factor of 0.6 (e.g., the range to effect for onset of injury of rockfish larvae was ~306m; this was reduced to 184 m).

6.6 Density Estimate Use in Exposure Estimates

The following sections discuss the density estimates used in the effects analyses.

6.6.1 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. These data were incorporated into a Geographic Information System database that includes seasonal (summer/fall and winter/spring) 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 (U.S. Department of the Navy 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 unit of 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. 2008b). 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 off California, Oregon, and Washington. 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. 2009b; 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. Details of the density derivation for each species of pinniped in the Action Area are provided in the Pacific Navy Marine Species Density Database Technical Report (U.S. Department of the Navy 2014). In summary, the methods used to derive pinniped densities involved a series of species-specific data reviews to compile the most accurate and up-to-date information available. This review was undertaken by a panel of subject matter experts, including marine mammal scientists from the Washington State Department of Fish and Wildlife, Navy, and ManTech International. Once all available information, including known haul-out sites and local abundance, had been reviewed and updated as necessary, the resulting numbers of animals were assigned to inland water areas divided into regions consistent with Jeffries et al. (Jeffries et al. 2003). The total abundance divided by the area of the region was the resultant density for each species in a given location.

6.6.2 Leatherback Sea Turtle

A quantitative analysis of impacts on a species requires data on the abundance and concentration of the species population in the potentially impacted area. The most appropriate metric for this type of analysis is density, which is the number of animals present per unit area. There is no single source of density data for every area of the world, species, and season because of the fiscal costs, resources, and effort involved in providing survey coverage to sufficiently estimate density. Therefore, to characterize the marine species density for large areas such as the Action Area, the Navy compiled data from several sources. To compile and structure the most appropriate database of marine species density data, the Navy developed a protocol to select the best available data sources based on species, area, and time (season). The resulting Geographic Information System database called the Navy Marine Species Density Database (DON 2012b) includes seasonal density values the leatherback sea turtle present within the Action Area. The species' density distribution matched the expected distribution from published literature and the NMFS stock assessments. In this Opinion, sea turtle density data were used as an input in the NAEMO in their original temporal and spatial resolution.

6.6.3 Fish

The methodologies used to estimate densities of ESA-listed fish in the Action Area are described in section 6.11.

6.7 Exposure and Response – Non-Impulsive Acoustic Stressors - Training

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 NWTT Action Area, it is discussed further in this Section. Exposure of fish to acoustic stressors was not modeled due to limited information on species distribution and density in the Action Area.

6.7.1 Exposure and Response of Marine Mammals to Non-Impulsive Acoustic Stressors – Training Activities

For this consultation, we considered exposure estimates from the Phase II NAEMO model at several output points for marine mammals and sea turtles. 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 exercises, 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. Neither sets of exposure estimates, the unprocessed or processed, were adjusted to account for standard mitigation actions that NMFS' Permits Division requires under the MMPA rule and LOA to avoid marine mammals, nor were the estimates adjusted to account for avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft. Since the processed exposure estimates represent incidental take for purposes of the ESA, we base our jeopardy analyses and determinations on these estimates. As stated above, these estimates were not adjusted to take into account the standard mitigation measures required under the MMPA rule or conducted by the Navy.

6.7.1.1 ***Blue Whale***

The NAEMO output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 84. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 84. Summary of Predicted Exposure of Blue Whales to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Submarine Mine Exercise	0.87	0%
Submarine Sonar Maintenance	18.69	7%
Surface Ship Sonar Maintenance	10.40	4%
TRACKEX – Helo	1.36	0%
TRACKEX – MPA	5.33	2%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	17.66	6%
TRACKEX – Sub	54.57	20%
TRACKEX – Surface	171.28	61%
Total Unprocessed Exposure Instances	280.16	100%

Based on the Navy's exposure models and processed instances of exposure, each year we would expect three instances, of the total 280 possible exposure instances, annually in which blue whales might be exposed to active sonar associated with training exercises and exhibit a behavioral response as a result of that exposure. We also anticipate two instances of TTS resulting from training activities. We do not expect any exposures leading to injury including PTS.

Temporary Threshold Shift (TTS) - As mentioned previously, 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 exercises 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 NWTT Action Area.

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. 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. Results from 2 years (2009 and 2010) of intensive monitoring by independent scientists and Navy observers in the Southern California (SOCAL) and Hawaii Range Complexes have recorded an estimated 161,894 marine mammals with no evidence of distress or unusual behavior observed during Navy activities, supporting that TTS is unlikely to significantly impair the ability of blue whales to communicate, forage, or breed and will not have fitness level consequences at the individual or population level.

Behavioral Responses - As discussed, 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 exercises and testing 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 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; 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.

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 experiments involving pseudo-random noise 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. 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 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. 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. (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 exercises conducted in the NWTT Action Area 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 to15 knots separated in distances when using active sonar. Thus, even during major training exercises, it is unlikely that a specific geographic area or water column would be ensonified at high levels for prolonged periods of time, which could increase the risk of significant behavioral responses or repeated disturbances in close temporal proximity. 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 exercises), 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 exercises, 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 testing and training activities, including the use of MFA sonar, have been ongoing in the NWTT Action Area for decades, and 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, has likely reached carrying capacity (Monnahan et al. 2014a). As stated previously, Navy training and testing activities in that area as with similar activities in the NWTT Action Area, 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 and testing activities in the NWTT Action Area are also likely to have not prevented the recovery of the blue whales throughout its range. The TTS and behavioral response estimates may change from year to year, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitness of affected individuals, is not expected to change.

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 these 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 ENP blue whale population has likely reached carrying capacity despite decades of Navy training and testing activities, including MFA sonar, in the Hawaii and Southern California Training Ranges.

6.7.1.2 *Fin Whale*

The NAEMO output estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in (Table 85). We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 85. Summary of Predicted Exposure of Fin Whales to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Submarine Mine Exercise	1.14	0%
Submarine Sonar Maintenance	105.92	7%
Surface Ship Sonar Maintenance	69.13	5%
TRACKEX - Helo	13.50	1%
TRACKEX - MPA	36.33	3%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	116.37	8%
TRACKEX - Sub	294.74	20%
TRACKEX - Surface	805.68	57%
Total Unprocessed Exposure Instances	1,442.82	100%

Based on the Navy's exposure models, each year we would expect 14 instances, of the total 1,443 potential instances above 120 dB SPL, annually in which fin whales might be exposed to active sonar associated with training exercises and exhibit a behavioral response to the exposure. We also expect 11 instances of TTS, but do not expect any exposures leading to injury including permanent threshold shift.

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 and testing 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 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 (1997b) 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 (1997b)). 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. Without more specific 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 and testing 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 and testing activities have not prevented fin whale numbers from increasing and potentially nearing ecosystem limits in Southern California. Since training and testing activities in the NWTT Action Area are similar to those in Southern California, we would expect the effects to be similar in the NWTT Action Area and thus not a limiting factor in the recovery of fin whales. The TTS and behavioral response estimates may change over baseline conditions from year to year, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitness of affected individuals, is not expected to change.

Non-impulsive acoustic stressors from Navy training exercises and testing activities conducted during the five-year year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of fin whales to reduce the viability or recovery of these populations. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. Therefore, we do not anticipate those behavioral responses to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.7.1.2.1 Humpback Whale

The NAEMO output estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 86. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 86. Summary of Predicted Exposure of Humpback Whales to Non-impulsive Sources during Training

Activity	**Number of Exposures > 120dB SPL**	**Percent Contribution by Activity**
Maritime Homeland Defense/Security Mine Countermeasures Integrated Exercise	0.00	0%
Submarine Mine Exercise	1.57	0%
Submarine Sonar Maintenance	232.00	18%
Surface Ship Sonar Maintenance	104.44	8%
TRACKEX - Helo	11.32	1%
TRACKEX - MPA	16.53	1%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	167.79	13%
TRACKEX - Sub	190.05	14%
TRACKEX - Surface	592.74	45%
Total Unprocessed Exposure Instances	1316.44	100%

Based on the Navy's exposure models, each year we would expect seven instances, of the total 1,316 potential instances above 120 dB SPL, annually in which humpback whales might be exposed to active sonar associated with training exercises and exhibit a behavioral response to the exposure. We also anticipate five instances of TTS, but do not expect any exposures leading to injury including permanent threshold shift.

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 exercises and testing 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. (2006b) 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 exercises and testing activities in the NWTT Action Area 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. Without more specific 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. Studies of reactions to airgun noises were inconclusive (Malme et al. 1985). Humpback whales on the breeding grounds did not stop singing in response to underwater explosions (Payne and McVay 1971). Humpback whales on feeding grounds did not alter short-term behavior or distribution in response to explosions with received levels of about 150 dB re 1 Pa/Hz at 350 Hz (Lien et al. 1993; Todd et al. 1996b). However, at least two individuals were probably killed by the high-intensity, impulse blasts and had extensive mechanical injuries in their ears (Ketten et al. 1993; Todd et al. 1996b). The explosions may also have increased the number of humpback whales entangled in fishing nets (Todd et al. 1996b). 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.

The overall abundance of humpback whales in the Pacific has continued to increase. The TTS and behavioral response estimates may change from year to year over baseline conditions, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitness of affected individuals, is not expected to change.

Acoustic stressors from Navy training and testing activities conducted during the five-year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of humpback whales to reduce the viability of these populations or adversely impact species recovery. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. Therefore, we do not anticipate those behavioral responses to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.7.1.2.2 Sei Whale

The NAEMO output estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 87. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 87. Summary of Predicted Exposure of Sei Whales to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Submarine Sonar Maintenance	5.74	6%
Surface Ship Sonar Maintenance	6.11	6%
TRACKEX – Helo	1.02	1%
TRACKEX – MPA	2.37	2%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	6.51	7%
TRACKEX – Sub	18.90	20%
TRACKEX - Surface	56.81	58%
Total Unprocessed Exposure Instances	97.46	100%

Based on the Navy's exposure models, each year we would not expect any instances annually in which sei whales might be exposed to active sonar associated with training exercises at a level that would elicit a behavioral response or potential injury such as permanent threshold shift. The 97 potential instances of exposure would be below thresholds for behavioral responses.

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 exercises and testing 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. Without more specific information, we assume the response of sei whales to TTS to be similar to that of blue whales. The TTS and behavioral response estimates may change from year to year over baseline conditions, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitness of affected individuals, is not expected to change.

Non-impulsive acoustic stressors from Navy training exercises and testing activities conducted during the five-year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of sei whales to reduce the viability of these populations or their ability to recover. We anticipate temporary behavioral responses from training activities, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses to rise to the level of take or to result in changes in reproduction, numbers, or distribution of these populations.

6.7.1.2.3 Sperm Whale

The NAEMO output estimates that sperm whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 88. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The vast majority of exposure instances are at received levels that are not likely to result in significant behavioral responses or higher level effects. In other words, the number of exposures that may result in a take is a small subset of the total estimated exposure instances that are expected. In other words, stressors from the Environmental Baseline and these acoustic stressors that by themselves do not result in significant behavioral responses, are present in the environment (baseline) and may or may not elicit minor responses such as alerting animals to pending danger such as exposure to higher-level acoustic stressors that may result or simply adding to ocean noise levels.

Table 88. Summary of Predicted Exposure of Sperm Whales to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Submarine Mine Exercise	4.52	0%
Submarine Sonar Maintenance	462.20	7%
Surface Ship Sonar Maintenance	487.76	7%
TRACKEX - Helo	77.34	1%
TRACKEX - MPA	303.19	5%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	356.14	5%
TRACKEX - Sub	1389.96	21%
TRACKEX - Surface	3474.61	54%
Total Unprocessed Exposure Instances	6555.72	100%

Based on the Navy's exposure models, each year we would expect 81 instances, of the total 6,556 potential instances above 120 dB SPL, annually in which sperm whales might be exposed to active sonar associated with training exercises and would exhibit behavioral responses to the exposures. We would not expect any exposures leading to TTS or injury including permanent threshold shift.

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 (as opposed to the shock waves from underwater detonations) the Navy conducts at the Naval Surface Warfare Center 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 1975a), 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 1975a). 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 1975a) 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 testing. 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.

6.7.1.2.4 Southern Resident Killer Whale

The NAEMO output estimates that Southern Resident killer whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 89. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The vast majority of exposure instances are at received levels that are not likely to result in significant behavioral responses or higher level effects. In other words, the number of exposures that may result in a take is a small subset of the total estimated exposure instances that are expected. In other words, stressors from the Environmental Baseline and these acoustic stressors that by themselves do not result in significant behavioral responses, are present in the environment (baseline) and may or may not elicit minor responses such as alerting animals to pending danger such as exposure to higher-level acoustic stressors that may result or simply adding to ocean noise levels.

Table 89. Summary of Predicted Exposure of Southern Resident Killer Whales to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Maritime Homeland Defense/Security Mine Countermeasures Integrated Exercise	5.00	100%
Submarine Sonar Maintenance	0.00	0%
Surface Ship Sonar Maintenance	0.00	0%
TRACKEX - Surface	0.00	0%
Total Unprocessed Exposure Instances	5.00	100%

Two of the five exposures are expected to result in behavioral responses and will occur every other year, (3 events in the five year period) from biennial civilian port defense training exercises. Therefore, no more than two behavioral responses rising to the level of take are expected to occur in any year of training and in years without civilian port defense training no behavioral responses are anticipated. No instances of TTS, PTS, injury, or mortality are expected as a result of training activities.

The response of Southern Resident killer whales to non-impulsive acoustic stressors associated with training exercises is expected to be similar to that of the sperm whale as discussed in section 6.7.1.2.3.

6.7.1.3 *Guadalupe Fur Seal*

Guadalupe fur seals are present within the coastal margins of the offshore portion of the Action Area during the warm season (summer and early autumn). Guadalupe fur seals are considered "seasonal" migrants since they return to rookeries in Mexican waters in the cold season. The Guadalupe fur seal is considered a single Mexico stock and is listed as threatened under the ESA (Carretta et al. 2013a).

The acoustic analysis predicts that Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in seven behavioral reactions annually. No instances of TTS are expected, and no instances of PTS or other injury or mortality are anticipated.

Research has demonstrated that for pinnipeds as for other mammals, recovery from a hearing threshold shift (i.e., TTS; temporary partial hearing loss) can take a few minutes to a few days depending on the severity of the initial shift. More severe shifts may not fully recover and thus would be considered PTS. Threshold shifts do not necessarily affect all hearing frequencies equally, so threshold shifts may not necessarily interfere with an animal's ability to hear biologically relevant sounds. As discussed previously, it is uncertain whether some permanent hearing loss over a part of a marine mammal's hearing range would have long-term consequences for that individual given that natural hearing loss occurs in marine mammals as a result of disease, parasitic infestations, and age-related impairment (Ketten 2012).

Pinnipeds in the water are tolerant of anthropogenic noise and activity. If seals 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. Seals 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.

6.7.2 Summary of Predicted Effects to Marine Mammals

Table 90 provides a summary of predicted effects on marine mammals from non-impulsive acoustic stressors during training activities.

Table 90. Summary of Effects on Marine Mammals from Non-impulsive Acoustic Stressors During Training

Species	Stock	Behavioral (Non-TTS)	TTS	PTS
Blue whale	Eastern North Pacific	3	2	0
Fin whale	Northeast Pacific	0	0	0
	CA/OR/WA	14	11	0
Humpback whale	CA/OR/WA	7	5	0
	Central North Pacific	0	0	0
Sei whale	Eastern North Pacific	0	0	0
Sperm whale	North Pacific	0	0	0
	CA/OR/WA	81	0	0
Killer whale	Southern Resident	2*	0	0
Guadalupe fur seal	Mexico	7	0	0

***Behavioral harassment of Southern Resident killer whales results from biennial civilian port defense training which will not occur more often than every other year starting with the first year under this BiOp and continuing into to reasonably foreseeable future.**

6.7.3 Exposure and Response of Leatherback Sea Turtles to Non-Impulsive Acoustic Stressors – Training Activities

For this consultation, we considered exposure estimates from the Navy Acoustic Effects Model (NAEMO) at two output points. First, the total number of ESA-listed species (animats) that would be exposed to acoustic sources greater than 120 dB 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 exercises and testing 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).

The second set of predicted exposures ("processed") 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. Neither sets of exposure estimates, the unprocessed or processed, consider standard mitigation actions to avoid marine mammals, nor did the estimates consider any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft or the stressor.

Most Navy sonar and other active acoustic sources used during training and testing in the NWTT Action Area use frequency ranges that are higher than the estimated hearing range of sea turtles (10 Hz to 2 kHz). Therefore, most of these sources have no impact on sea turtle hearing. It is likely that only sonar with source levels greater than 160 dB re 1 μPa at 1 m using frequencies within the hearing range of sea turtles have potential for acoustic impacts on sea turtles that may lead to take pursuant to the ESA. Other active acoustic sources with low source level, narrow beam width, downward-directed transmission, short pulse lengths, frequencies above known hearing ranges, or some combination of these factors are not anticipated to result in impacts on sea turtles. These sources generally have frequencies greater than 200 kHz and source levels less than 160 dB re 1 μPa. The types of sources with source levels less than 160 dB are primarily hand-held sonar, range pingers, transponders, and acoustic communication devices.

Sea turtles that reside in the Action Area may be exposed several times throughout the year to sound from sonar and other active acoustic sources. Exposures to sonar and other active acoustic sources in open water areas would be intermittent and geographically variable. The potential for leatherback sea turtles to experience some form of hearing loss was predicted by the Navy using the NAEMO. To quantify the impacts of acoustic exposures to sea turtles, training and testing activities that employ acoustic sources using frequencies in the hearing range of sea turtles were analyzed. Most Navy sonar and active acoustic sources used during testing and training activities use frequencies that are outside of the estimated hearing range of turtles.

The model output estimates that leatherback sea turtles will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year.

Table 91. Summary of Predicted Exposure of Leatherback Sea Turtles to Non-impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Submarine Mine Exercise	0.00	0%
Submarine Sonar Maintenance	0.00	0%
Surface Ship Sonar Maintenance	0.00	0%
TRACKEX - Helo	0.00	0%
TRACKEX - MPA	0.00	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	321.28	100%
TRACKEX - Sub	0.00	0%
TRACKEX - Surface	0.00	0%
Total Unprocessed Exposure Instances	321.28	100%

Our analysis based on Navy modeling indicates that leatherback sea turtles may be exposed approximately 321 times annually to sonar and other active acoustic sources associated with training activities in the Offshore Area of the NWTT Action Area. Of the total predicted exposures above 120 dB SPL, these exposures are anticipated to result in one TTS exposure annually. Zero sea turtles are predicted to experience PTS due to training with sonar and other active acoustic sources, which would permanently reduce perception of sound within a limited frequency range. We do not anticipate any instances of behavioral harassment above the predicted instance of TTS.

Cues preceding the commencement of the event (e.g., vessel presence and movement, aircraft overflight) may result in some animals departing the immediate area, even before active sound sources begin transmitting. Avoidance behavior could reduce the sound exposure level experienced by a sea turtle and therefore reduce the likelihood and degree of TTS predicted near sound sources. In addition, PTS and TTS threshold criteria for sea turtles are conservatively based on criteria developed for mid-frequency marine mammals. Therefore, actual TTS impacts are expected to be substantially less than the predicted quantities.

Sea turtles may exhibit short-term behavioral reactions, such as swimming away or diving to avoid the immediate area around a source, although studies examining sea turtle behavioral responses to sound have used impulsive sources, not non-impulsive sources. In most cases, acoustic exposures are intermittent, allowing time to recover from an incurred energetic cost, resulting in no long-term consequence. These short-term intermittent behavioral reactions are not expected to rise to the level of take.

6.7.4 Exposure and Response of Fish to Non-Impulsive Acoustic Stressors – Training Activities

For the purposes of this consultation, the Navy and NMFS developed sound exposure criteria for low and mid frequency sonar for fish (see section 6.5.8.1). 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. (2014), 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 (with the exception of eulachon which do not have a swim bladder and are even less susceptible to injury or mortality from underwater sound) so our discussion and analysis focuses on these criteria. The criteria developed are discussed in Section 3.1.3 and shown in Section 6.5.8.1.

Threshold criteria were not developed for high frequency sonar sources. 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 (with the exception of eulachon which do not have swim bladders and are even less susceptible to injury or mortality from underwater sound). 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 6.5.8.1, all fish within a two meter radius of sources in the LF4 source bin (low-frequency sources equal to 180 dB and up to 200 dB) and within 1 meter of sources in the ASW2 source bin (mid-frequency multi-static active coherent sonobuoy) could experience TTS. Sources in the LF4 source bin are proposed to be used for 110 hours and sources in the ASW2 source bin are proposed to be used for 64 hours annually. Sonar use within the LF4 source bin will only occur during testing in inland waters. Fish within two meters of the source could experience TTS. Sonar use with the ASW2 source bin will occur during testing in the offshore environment, >12 nautical miles from shore. Fish within one meter of this source could experience TTS.

However, 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). Additionally, the ESA-listed 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. (2014) 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. 2014). Additionally, hearing is not thought to play a role in salmon migration (e.g., Putnam et al. 2013). Because these species are able to rely on alternative mechanisms for these essential life functions, instances of TTS would not increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take. 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. (2014) 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. 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.

6.8 Exposure and Response – Non-Impulsive Acoustic Stressors - Testing

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 NWTT Action Area, it is discussed further in this Section.

6.8.1 Exposure and Response of Marine Mammals to Non-Impulsive Acoustic Stressors – Testing Activities

For this consultation, we considered exposure estimates from the Phase II NAEMO model at several output points for marine mammals and sea turtles. 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 testing exercises, 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. Neither sets of exposure estimates, the unprocessed or processed, were adjusted to account for standard mitigation actions that NMFS' Permits Division requires under the MMPA rule and LOA to avoid marine mammals, nor were the estimates adjusted to account for any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft. Since the processed exposure estimates represent incidental take for purposes of the ESA, we base our jeopardy analyses and determinations on these estimates. As stated above, these estimates were not adjusted to account for the standard mitigation measures required under the MMPA rule or conducted by the Navy.

The following sections describe the predicted exposures of marine mammals to non-impulsive acoustic stressors during testing including the unprocessed estimate of total exposures above 120 dB SPL and the processed exposure estimates that we use to enumerate take. Responses of species to these stressors are expected to be the same as described in Section 6.7.1.

6.8.1.1 *Blue Whale*

The NAEMO output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 92. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 92. Summary of Predicted Exposure of Blue Whales to Non-impulsive Sources during Testing

Activity	**Number of Exposures > 120dB SPL**	**Percent Contribution by Activity**
Acoustic Component Testing	0.85	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	1.18	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	143.40	57%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	0.04	0%
Countermeasure Testing	21.62	8%
Fleet Training / Support	2.96	1%
LCS Mission Package Testing – ASW	80.28	32%
Torpedo (Explosive) Testing	1.52	1%
Torpedo Non-Explosive Testing	3.81	1%
Total Unprocessed Exposure Instances	255.64	100%

Blue whales will be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic analysis predicts that blue whales of the Eastern

North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that the 256 potential exposures above 120 dB SPL each year may result in six TTS exposures annually. We do not anticipate any instances of injury including PTS.

As we discussed in 6.7.1.1, Blue whales exposed to mid-frequency are less likely to produce low frequency calls usually associated with feeding behavior (Melcon et al. 2012). 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. In contrast, blue whales increased their likelihood of calling when ship noise was present, and decreased their likelihood of calling in the presence of explosive noise, although this result was not statistically significant (Melcon et al. 2012).

Additionally, 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 (Melcon et al. 2012). Preliminary results from the 2010 to 2011 field season 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 but that those responses were mild and there was a quick return to their baseline activity (Southall et al. 2011a). Blue whales responded to a mid-frequency sound source, with a source level between 160 and 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 during CEEs (Goldbogen et al. 2013). However, reactions were not consistent across individuals based on received sound levels alone, and likely were 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 CEEs, but deep feeding and non-feeding whales showed temporary reactions that quickly abated after sound exposure. Whales were sometimes less than a mile from the sound source during CEEs.

6.8.1.2 ***Fin Whale***

The NAEMO output estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 93. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales

(Au et al. 2000b) 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 that distance less likely.

Table 93. Summary of Predicted Exposure of Fin Whales to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Acoustic Component Testing	5.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	8.16	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	971.74	48%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	0.35	0%
Cold Water Training	8.39	0%
Component System Testing	24.02	1%
Countermeasure Testing	101.21	5%
Electromagnetic Measurement	15.00	1%
Fleet Training / Support	18.41	1%
LCS Mission Package Testing - ASW	466.97	23%
Measurement System Repair/Replacement	24.22	1%
POPS	135.07	6%
Surface Vessel Acoustic Measurement - SAS	12.01	1%
Target Strength Testing	4.67	0%
Torpedo (Explosive) Testing	2.56	0%
Torpedo Non-Explosive Testing	24.12	1%
Underwater Vessel Acoustic Measurement - HLF/AASP	5.07	0%
Underwater Vessel Acoustic Measurement - Life Cycle	162.00	8%
Underwater Vessel Acoustic Measurement - Post Construction	90.00	4%
Underwater Vessel Acoustic Measurement - WSQ-9	6.00	0%
Total Unprocessed Exposure Instances	2,084.96	100%

The acoustic analysis predicts that fin whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area and that potential 2,085 exposures above 120 dB SPL each year may result in 30 TTS, and 4 behavioral reactions. Fin whales found in Southeast Alaska (Western Behm Canal) are recognized as part of the Northeast Pacific stock. Furthermore, the acoustic analysis predicts fin whales of the Northeast Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities that may result in two behavioral reactions annually. We do not anticipate any instances of injury including PTS.

6.8.1.3 ***Humpback Whale***

The NAEMO output estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 94. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 94. Summary of Predicted Exposure of Humpback Whales to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Acoustic Component Testing	6.26	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	3.74	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	1372.01	49%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	0.17	0%
Cold Water Training	8.44	0%
Component System Testing	24.01	1%
Countermeasure Testing	136.62	5%
Electromagnetic Measurement	15.00	1%
Fleet Training / Support	18.20	1%
LCS Mission Package Testing - ASW	734.89	27%
Measurement System Repair/Replacement	25.03	1%
POPS	135.07	5%
Surface Vessel Acoustic Measurement - SAS	12.01	0%
Target Strength Testing	3.20	0%
Torpedo (Explosive) Testing	0.73	0%
Torpedo Non-Explosive Testing	27.14	1%
Underwater Vessel Acoustic Measurement - HLF/AASP	5.71	0%
Underwater Vessel Acoustic Measurement - Life Cycle	162.00	6%
Underwater Vessel Acoustic Measurement - Post Construction	90.00	3%
Underwater Vessel Acoustic Measurement - WSQ-9	6.00	0%
Total Unprocessed Exposure Instances	2,786.23	100%

The acoustic analysis predicts that humpback whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area. Of the total 2,786 potential exposures above 120 dB SPL, we expect 39 instances of TTS and 5 behavioral reactions in addition to the behavioral reactions associated with the instances of TTS. Humpback whales found in Southeast Alaska (Western Behm Canal) are recognized as part of the Central North Pacific stock and the acoustic analysis predicts that the Central North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities that may result in 1 behavioral reaction annually. We do not anticipate any instances of injury including PTS.

6.8.1.4 *Sei Whale*

The NAEMO output estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 95. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 95. Summary of Predicted Exposure of Sei Whales to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Acoustic Component Testing	1.66	2%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	0.47	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	55.31	51%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	0.02	0%
Countermeasure Testing	10.54	10%
Fleet Training / Support	3.58	3%
LCS Mission Package Testing – ASW	28.17	27%
Torpedo (Explosive) Testing	1.26	1%
Torpedo Non-Explosive Testing	6.39	6%
Total Unprocessed Exposure Instances	107.40	100%

The acoustic analysis predicts that sei whales of the Eastern North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area.

Of the potential 107 instances of exposure above 120dB SPL, we expect two instances of TTS exposure annually. We do not anticipate any instances of injury including PTS.

6.8.1.5 ***Sperm Whale***

The NAEMO output estimates that sperm whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 96. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 96. Summary of Predicted Exposure of Sperm Whales to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Acoustic Component Testing	12.92	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	58.32	1%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	2242.28	53%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	3.20	0%
Countermeasure Testing	209.94	5%
Fleet Training / Support	65.63	2%
LCS Mission Package Testing - ASW	1598.87	39%
Torpedo (Explosive) Testing	2.36	0%
Torpedo Non-Explosive Testing	12.08	0%
Total Unprocessed Exposure Instances	4,205.59	100%

The acoustic analysis predicts that sperm whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area. Of the 4,206 potential instances of exposure above 120 dB SPL, we anticipate 67 instances of TTS and 11 behavioral reactions in addition to the behavioral reactions associated with TTS. There are no model predicted exposures for the North Pacific stock of sperm whales. We do not anticipate any instances of injury including PTS.

Recovery from TTS 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.

Sperm whales that are exposed to sonar and other active acoustic sources may react by alerting, ignoring the stimulus, changing their behaviors or vocalizations, or avoiding the area by swimming away or diving. Occasional behavioral reactions and temporary threshold shifts are unlikely to cause long-term consequences for individual animals or populations.

6.8.1.6 ***Southern Resident Killer Whale***

The NAEMO output estimates that Southern Resident killer whales will not be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 97. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources. The low received level (approximately 120 dB SPL) from the sonar at a distance exceeding approximately 54 mi. (100 km) is modeled as having some minor behavioral effects although masking by other ambient sounds, such as other vocalizing whales (Au et al. 2000b) 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 that distance less likely.

Table 97. Summary of Predicted Exposure of Southern Resident Killer Whales to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Fleet Training / Support	0.00	0%
LCS Mission Package Testing - ASW	0.00	0%
Total Unprocessed Exposure Instances	0.00	100%

The majority of testing events occur in areas such as Hood Canal, where Southern Resident killer whales are not believed to be present; the remaining testing activities occur offshore, where they are only present briefly during their annual migration period. Acoustic modeling predicts that Southern Resident killer whales may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area. However, there are no predicted effects from these testing activities because there are no potential exposures above 120 dB SPL and subsequently there are no potential exposures that would exceed the current effects thresholds.

6.8.1.7 ***Guadalupe Fur Seal***

Guadalupe fur seals are present within the coastal margins of the offshore portion of the Action Area during the warm season (summer and early autumn). Guadalupe fur seals are considered "seasonal" migrants since they return to rookeries in Mexican waters in the cold season. The Guadalupe fur seal is considered a single Mexico stock and is listed as threatened under the ESA (Carretta et al. 2013a).

The acoustic analysis predicts that Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in three behavioral reactions annually. We do not anticipate any instances of injury including PTS.

Research and observations show that pinnipeds in the water are tolerant of anthropogenic noise and activity. If seals 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. Seals may not react at all until the sound source is approaching within a few hundred meters and then may alert, ignore the stimulus, change their behaviors, or avoid the immediate area by swimming away or diving. Significant behavioral reactions would not be expected in most cases.

6.8.2 Summary of Predicted Effects to Marine Mammals

Table 98 provides a summary of predicted effects on marine mammals from non-impulsive acoustic stressors during testing activities.

Table 98. Summary of Effects on Marine Mammals from Non-impulsive Acoustic Stressors During Testing

Species	Stock	Behavioral (Non-TTS)	TTS	PTS
Blue whale	Eastern North Pacific	0	6	0
Fin whale	Northeast Pacific	2	0	0
	CA/OR/WA	4	30	0
Humpback whale	CA/OR/WA	5	39	0
	Central North Pacific	1	0	0
Sei whale	Eastern North Pacific	0	2	0
Sperm whale	North Pacific	0	0	0
	CA/OR/WA	11	67	0
Killer whale	Southern Resident	0	0	0
Guadalupe fur seal	Mexico	3	0	0

6.8.3 Exposure and Response of Leatherback Sea Turtles to Non-Impulsive Acoustic Stressors – Testing Activities

This section describe the predicted exposures of leatherback sea turtles to non-impulsive acoustic stressors during training including the unprocessed estimate of total exposures above 120 dB SPL and the processed exposure estimates that we use to enumerate take (

Table 99).

Table 99. Summary of Predicted Exposure of Leatherback Sea Turtles to Non-impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Acoustic Component Testing	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (DICASS)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-84)	0.00	0%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (Multi-Static Active Coherent-MAC)	2087.89	62%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (NAVAIR HDC)	1.14	0%
Countermeasure Testing	0.00	0%
Fleet Training / Support	0.54	0%
LCS Mission Package Testing - ASW	1267.25	38%
Torpedo (Explosive) Testing	0.00	0%
Torpedo Non-Explosive Testing	1.57	0%
Total Unprocessed Exposure Instances	3,358.39	100%

Our analysis indicates that leatherback sea turtles may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area of the NWTT Action Area approximately 3,358 times annually. Of those exposure instances, five will likely result in TTS annually. We do not anticipated any instances of behavioral harassment above the five instances of TTS. We do not anticipate any instances of injury including PTS.

For an analysis of the likely responses of leatherback sea turtles to exposure to non-impulsive acoustic stressors, see our discussion in section 6.7.3.

6.8.4 Exposure and Response of Fish to Non-Impulsive Acoustic Stressors – Testing Activities

See discussion in Section 6.7.4 for an analysis of the potential effects of non-impulsive acoustic stressors on fish.

6.9 Exposure and Response – Impulsive Stressors - Training

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 NWTT Action Area, it is discussed further in this Section.

6.9.1 Exposure and Response of Marine Mammals to Impulsive Acoustic Stressors – Training Activities

As with non-impulsive sources, we considered exposure estimates from the Navy Acoustic Effects Model for marine mammals and sea turtles. 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 exercises, 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. As above, these are the estimates we use in our jeopardy analysis and determination.

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. In this case there are no Level A exposures and thus the estimated amounts are not changed. In other words, neither sets of exposure estimates, the unprocessed or processed, were adjusted to account for standard mitigation actions that NMFS' Permits Division requires under the MMPA rule and LOA to avoid marine mammals, nor were the estimates adjusted to account for any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft.

Estimated unprocessed exposures of marine mammals and sea turtles to impulsive sound from underwater detonations during training exercises at levels greater than 120 dB SPL are summarized in the following sections. As with these estimates for the non-impulsive sources, we do not rely on these estimates in our jeopardy analysis.

6.9.1.1 *Blue Whale*

The NAEMO output estimates that blue whales will be exposed to explosions associated with training activities throughout the year. The unprocessed estimate of exposure events above

120dB SPL and the contribution of activities to these exposures are provided in Table 100. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 100. Summary of Predicted Exposure of Blue Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	3.88127	49%
GUNEX [S-S] – Ship	0.2962	4%
MISSILEX [A-S]	2.402	30%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	1.301	17%
Total Unprocessed Exposure Instances	7.88047	100%

Acoustic modeling predicts that blue whales may be exposed approximately eight times per year to impulses from explosive sources at levels above 120 dB SPL during training activities in the Offshore Area. However, there are no predicted effects (processed exposure instances) from these eight instances, because exposures would not exceed the current effects thresholds.

6.9.1.2 *Fin Whale*

The NAEMO output estimates that fin whales will be exposed to explosions and other impulsive stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 101. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 101. Summary of Predicted Exposure of Fin Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	22.34309	48%
GUNEX [S-S] - Ship	1.1686	2%
MISSILEX [A-S]	12.162	26%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	11.07675	24%
Total Unprocessed Exposure Instances	46.75044	100%

Acoustic modeling predicts that fin whales may be exposed approximately 47 times per year to impulses from explosive sources at levels above 120 dB SPL during training activities in the Offshore Area. However, there are no predicted effects (processed exposure instances) from these 47 instances, because exposures would not exceed the current effects thresholds.

6.9.1.3 *Humpback Whale*

The NAEMO output estimates that humpback whales will be exposed to explosions and other impulsive stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 102. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 102. Summary of Predicted Exposure of Humpback Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	8.09876	48%
GUNEX [S-S] - Ship	1.338	8%
MISSILEX [A-S]	3.898	23%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	3.41825	20%
Total Unprocessed Exposure Instances	16.75301	100%

Acoustic modeling predicts that Humpback whales may be exposed approximately 17 times per year to impulses from explosive sources at levels above 120 dB SPL during training activities in the Offshore Area. However, there are no predicted effects (processed exposure instances) from these 17 instances, because exposures would not exceed the current effects thresholds.

6.9.1.4 *Sei Whale*

The NAEMO output estimates that sei whales will be exposed to explosions and other impulsive stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 103. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 103. Summary of Predicted Exposure of Sei Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	3.19126	43%
GUNEX [S-S] - Ship	0.4294	6%
MISSILEX [A-S]	3.128	42%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	0.692	9%
Total Unprocessed Exposure Instances	7.44066	100%

Acoustic modeling predicts that sei whales may be exposed approximately seven times per year to impulses from explosive sources at levels above 120 dB SPL during training activities in the Offshore Area. However, there are no predicted effects (processed exposure instances) from these seven instances, because exposures would not exceed the current effects thresholds.

6.9.1.5 *Sperm Whale*

The NAEMO output estimates that sperm whales will be exposed to explosions and other impulsive stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 104. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive sources.

Table 104. Summary of Predicted Exposure of Sperm Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	29.83622	57%
GUNEX [S-S] - Ship	4.3074	8%
MISSILEX [A-S]	7.514	15%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	10.59225	20%
Total Unprocessed Exposure Instances	52.24987	100%

Acoustic modeling predicts that sperm whales may be exposed approximately 52 times per year to impulses from explosive sources at levels above 120 dB SPL during training activities in the Offshore Area. However, there are no predicted effects (processed exposure instances) from these 52 instances, because exposures would not exceed the current effects thresholds.

6.9.1.6 *Southern Resident Killer Whale*

The model output estimates that no southern resident killer whales will be exposed to impulsive stressors associated with training activities throughout the year. The Navy modeled exposure from Mine Neutralization – Explosive Ordnance Disposal activities resulted in zero exposures (Table 105).

Table 105. Summary of Predicted Exposure of Southern Resident Killer Whales to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Mine Neutralization - EOD	0.066	100%
Total Unprocessed Exposure Instances	0.066	100%

6.9.1.7 *Guadalupe Fur Seal*

Acoustic modeling predicts that Guadalupe fur seal may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds.

6.9.2 Exposure and Response of Leatherback Sea Turtles to Impulsive Stressors – Training Activities

Impulsive stressors such as explosions in the water or near the water's surface can introduce loud, impulse, broadband sounds into the marine environment. These sounds are likely to be within the audible range of the leatherback sea turtles, but the duration of individual sounds is very short. Energy from explosions is capable of causing mortalities, injuries to the lungs or gastrointestinal tract, TTS or PTS, or behavioral responses. The level of effect on leatherback sea turtles from the use of explosives depends on the net explosive weight (NEW) of the charge, the depth of the charge, the properties of detonations underwater, the animal's distance from the charge, the animal's location in the water column, and environmental factors such as water depth, water temperature, and bottom type. The NEW accounts for the weight and the type of explosive material.

The NAEMO output estimates that leatherback sea turtles will be exposed to explosions and other impulsive stressors associated with training activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 106. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive sources. The vast majority of exposure instances are at received levels that are not likely to result in significant behavioral responses or higher level effects. In other words, the number of exposures that may result in a take is a small subset of the total estimated exposure instances that are expected. In other words, stressors from the Environmental Baseline and these acoustic stressors that by themselves do not result in significant behavioral responses, are present in the environment (baseline) and may or may not elicit minor responses such as alerting animals to pending danger such as exposure to higher-level acoustic stressors that may result or simply adding to ocean noise levels.

Table 106. Summary of Predicted Exposure of Leatherback Sea Turtles to Impulsive Sources during Training

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
BOMBEX [A-S]	41.56747	50%
GUNEX [S-S] - Ship	4.2301	5%
MISSILEX [A-S]	19.19	23%
SINKEX	0	0%
TRACKEX - MPA (Multi-Static Active Coherent - MAC)	18.35175	22%
Total Unprocessed Exposure Instances	83.33932	100%

As the table above illustrates, up to 142 training activities using explosives at or beneath the water surface would be conducted annually and potentially expose leatherback sea turtles to underwater impulse sound. The largest source class used during training would be E12 (> 650 to 1,000 lb. NEW), which would be used 10 times annually in the NWTT Action Area. Leatherback sea turtles may be exposed to impulse sound from explosive sources associated with training activities in the Offshore Area; however, there are no predicted mortality, onset slight lung injury, gastrointestinal tract injury, PTS, or TTS effects from these activities, because exposures would not exceed the current effects thresholds.

In addition to Navy-modeled estimates for TTS, PTS, Slight Lung Injury, and Mortality from impulsive acoustic stressors, we calculated the number of modeled (unprocessed) instances of behavioral harassment of sea turtle species. To do this, we summed the number of modeled exposures greater than 175 dB and subtracted instances of estimated TTS, PTS, slight lung injury, and mortality (zero in this case) to provide an estimate (balance of exposures greater than 175dB) of exposures that may result in more than minor behavioral responses potentially rising to the level of "take" pursuant to the ESA. These unprocessed estimates do not take into account Navy standard operating procedures and mitigation activities that might reduce actual instances of harassment. From this we concluded that 14 leatherback sea turtles would be behaviorally harassed.

Activities consisting of single explosions, such as bombing and missile exercise, are expected to only elicit short-term startle reactions. If a sea turtle hears multiple explosions in a short period, such as during gunnery, firing, or sonobuoy exercises, it may react by avoiding the area. Most activities would consist of a limited number of explosions and exposures would not occur over long durations. Similar to training activities, behavioral effects are expected to be minor and temporary.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication.

Further, although the information on the hearing capabilities of sea turtles is limited, the information available suggests that the auditory capabilities of sea turtles are centered in the low-frequency range (<2 kHz) (Bartol et al. 1999b; Dow Piniak et al. 2012; Lenhardt et al. 1983;

Lenhardt et al. 1994a; Martin et al. 2012; O'Hara and Wilcox 1990; Ridgway et al. 1969), with greatest sensitivity below 1 kHz. A more recent review on sea turtle hearing and sound exposure indicated that sea turtles detect sounds at less than 1000Hz and therefore would not be affected by mid or high-frequency active sonar (Popper et al. 2014).

Similarly, a study on the effects of airguns on sea turtle behavior also suggests that sea turtles are most likely to respond to low-frequency sounds. McCauley et al. (2000) reported that green and loggerhead turtles will avoid air-gun arrays at 2 km and at 1 km with received levels of 166 dB re 1 μPa and 175 db re 1 μPa, respectively. The sea turtles responded consistently: above a level of approximately 166 dB re 1 μPa the turtles noticeably increased their swimming activity compared to non-airgun operation periods. Above 175 dB re 1 μPa mean squared pressure their behavior became more erratic possibly indicating the turtles were in an agitated state. A study conducted in the Mediterranean Sea found that of 164 loggerhead turtles observed, 57 percent responded to the firing of an air gun array (source level 252 dB re 1 μPa [peak]) by diving at or before their closest point of approach to the airguns, with dive probability decreasing with increasing distance from the airgun array (DeRuiter and Larbi Doukara 2012).

Acoustic stressors associated with the Navy's activities in the NWTT Action Area have the ability to cause behavioral responses in sea turtles. The response of a sea turtle 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). Distance from the sound source and whether it is perceived as approaching or moving away could also affect the way a sea turtle responds. Potential behavioral responses to anthropogenic sound could include startle reactions, disruption of feeding, disruption of migration, changes in respiration, alteration of swim speed, alteration of swim direction, and area avoidance. Any disruptions are expected to be temporary in nature, with the animal resuming normal behaviors shortly after the exposure. To result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for the energy expended during evasion and any potential lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Similarly, to result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for the energy expended during evasion rather than would have been incurred during normal migration, respiration, swimming speed, and swimming direction. There is no indication this is the case, particularly since foraging habitat would still be available in the environment following the cessation of acoustic exposure. Therefore, behavioral responses of leatherback sea turtles to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population.

6.10 Exposure and Response – Impulsive Stressors - Testing

The following sections discuss our analysis of impulsive 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 NWTT Action Area, it is discussed further in this Section.

6.10.1 Exposure and Response of Marine Mammals to Impulsive Acoustic Stressors – Testing Activities

As with impulsive sources during training activities, we considered exposure estimates from the Navy Acoustic Effects Model for marine mammals. First, the total number of ESA-listed species (animats) that would be exposed to acoustic sources prior to the application of 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 exercises, 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. As above, these are the estimates we use in our jeopardy analysis and determination.

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. In this case there are no Level A exposures and thus the estimated amounts are not changed. In other words, neither sets of exposure estimates, the unprocessed or processed, were adjusted to account for standard mitigation actions that NMFS' Permits Division requires under the MMPA rule and LOA to avoid marine mammals, nor were the estimates adjusted to account for any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft.

Estimated unprocessed exposures of marine mammals to impulsive sound from underwater detonations during training exercises at levels greater than 120 dB SPL are summarized in the following sections. As with these estimates for the non-impulsive sources, we do not rely on these estimates in its jeopardy analysis.

6.10.1.1 *Blue Whale*

The NAEMO output estimates that blue whales will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 107. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive sources.

Table 107. Summary of Predicted Exposure of Blue Whales to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	0.59416	10%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	0.719	12%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	0.17978	3%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	0.17978	3%
Torpedo (Explosive) Testing	4.398	72%
Total Unprocessed Exposure Instances	6.07072	100%

Acoustic modeling predicts (processed exposure estimates) that blue whales may be exposed to approximately six impulses from explosive sources associated with testing activities in the Offshore Area; however, there are no predicted effects (processed exposure instances) from these activities that rise to the level of take, because exposures would not exceed the current effects thresholds.

6.10.1.2 *Fin Whale*

The NAEMO output estimates that fin whales will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 108. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive acoustic sources.

Table 108. Summary of Predicted Exposure of Fin Whales to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	5.01468	22%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	4.674	20%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	1.16847	5%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	1.16847	5%
Torpedo (Explosive) Testing	11.133	48%
Total Unprocessed Exposure Instances	23.15862	100%

Acoustic modeling predicts (processed exposure estimates) that fin whales may be exposed to approximately 23 impulses from explosive sources associated with testing activities in the Offshore Area; however, there are no predicted effects (processed exposure instances) from these activities that rise to the level of take, because exposures would not exceed the current effects thresholds.

6.10.1.3 *Humpback Whale*

The NAEMO output estimates that humpback whales will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 109. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 109. Summary of Predicted Exposure of Humpback Whales to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	1.51028	17
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	1.4555	6
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	0.36388	44
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	0.36388	44
Torpedo (Explosive) Testing	4.848	56
Total Unprocessed Exposure Instances	8.54154	100%

Acoustic modeling predicts (processed exposure estimates) that humpback whales may be exposed to approximately nine impulses from explosive sources associated with testing activities in the Offshore Area; however, there are no predicted effects (processed exposure instances) from these activities that rise to the level of take, because exposures would not exceed the current effects thresholds.

6.10.1.4 *Sei Whale*

The NAEMO output estimates that sei whales will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 110. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to non-impulsive acoustic sources.

Table 110. Summary of Predicted Exposure of Sei Whales to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	0.31587	7%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	0.3	6%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	0.07502	2%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	0.07502	2%
Torpedo (Explosive) Testing	4.044	84%
Total Unprocessed Exposure Instances	4.80991	100%

Acoustic modeling predicts (processed exposure estimates) that sei whales may be exposed to approximately five impulses from explosive sources associated with testing activities in the Offshore Area; however, there are no predicted effects (processed exposure instances) from these activities that rise to the level of take, because exposures would not exceed the current effects thresholds.

6.10.1.5 *Southern Resident Killer Whale*

The NAEMO and post-model analysis predicts no exposures of Southern Resident killer whales from impulse sources associated with testing activities above 120 dB SPL. Therefore, we do not anticipate any effects to Southern Resident killer whales from these activities.

6.10.1.6 *Sperm Whale*

The NAEMO output estimates that sperm whales will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 111. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive sources.

Table 111. Summary of Predicted Exposure of Sperm Whales to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	5.0138	21%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	6.4355	27%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	1.60888	7%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	1.60888	7%
Torpedo (Explosive) Testing	9.15	38%
Total Unprocessed Exposure Instances	23.81706	100%

Acoustic modeling predicts (processed exposure estimates) that sperm whales may be exposed to approximately 24 impulses from explosive sources associated with testing activities in the Offshore Area; however, there are no predicted effects (processed exposure instances) from these activities that rise to the level of take, because exposures would not exceed the current effects thresholds.

6.10.1.7 *Guadalupe Fur Seal*

As with training activities, the NAEMO and post-model analysis predicts no exposures of Guadalupe fur seals from impulse sources associated with testing activities.

6.10.2 Summary of Predicted Effects to Marine Mammals

Acoustic modeling predicts that ESA-listed mysticetes (humpback whales, blue whales, fin whales, sei whales), odontocetes (sperm whales and Southern Resident killer whales), and Guadalupe fur seals may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects (Table 112) from these activities, because exposures would not exceed the current effects thresholds. There is potential for exposure above 120 dB SPL that would not rise to the level of take.

Table 112. Summary of Effects on Marine Mammals from Impulsive Acoustic Stressors During Testing

Species	Stock	Behavioral (Non-TTS)	TTS	PTS
Blue whale	Eastern North Pacific	0	0	0
Fin whale	Northeast Pacific	0	0	0
	CA/OR/WA	0	0	0
Humpback whale	CA/OR/WA	0	0	0
	Central North Pacific	0	0	0
Sei whale	Eastern North Pacific	0	0	0
Sperm whale	North Pacific	0	0	0
	CA/OR/WA	0	0	0
Killer whale	Southern Resident	0	0	0
Guadalupe fur seal	Mexico	0	0	0

6.10.3 Exposure and Response of Leatherback Sea Turtles to Impulsive Acoustic Stressors – Testing Activities

The NAEMO output estimates that leatherback sea turtles will be exposed to explosions and other impulsive stressors associated with testing activities throughout the year. The unprocessed estimate of exposure events above 120 dB SPL and the contribution of activities to these exposures are provided in Table 113. We do not rely on these potential exposures for our jeopardy analysis, but include this information to illustrate the context of exposures of species to impulsive sources. The vast majority of exposure instances are at received levels that are not likely to result in significant behavioral responses or higher level effects. In other words, the number of exposures that may result in a take is a small subset of the total estimated exposure instances that are expected. In other words, stressors from the Environmental Baseline and these acoustic stressors that by themselves do not result in significant behavioral responses, are present in the environment (baseline) and may or may not elicit minor responses such as alerting animals to pending danger such as exposure to higher-level acoustic stressors that may result or simply adding to ocean noise levels.

Table 113. Summary of Predicted Exposure of Leatherback Sea Turtles to Impulsive Sources during Testing

Activity	Number of Exposures > 120dB SPL	Percent Contribution by Activity
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (IEER)	8.46216	21%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-61)	10.066	25%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-64)	2.51651	6%
Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft (MK-82)	2.51651	6%
Torpedo (Explosive) Testing	16.59	42%
Total Unprocessed Exposure Instances	40.15118	100%

Explosives would be used only in the Offshore Area of the Action Area and for a total of 148 events per year. Explosive ordnance used during testing activities include explosive torpedoes (12), sound underwater signal sonobuoys (72), and Improved Extended Echo Ranging (IEER) sonobuoys (70). While the number of leatherback turtles anticipated in the NWTT Action Area is low, if a leatherback turtle was within the ranges listed in Table 79, the respective impact would be anticipated. However, results from modeling indicate no leatherback sea turtles are predicted to be exposed to impulse levels associated with the onset of mortality, onset slight lung injury, gastrointestinal tract injury, PTS, or TTS over any testing year for explosives use in open ocean.

In addition to Navy-modeled estimates for TTS, PTS, Slight Lung Injury, and Mortality from impulsive acoustic stressors, we calculated the number of modeled (unprocessed) instances of behavioral harassment of sea turtle species. To do this, we summed the number of modeled exposures greater than 175 dB and subtracted instances of estimated TTS, PTS, slight lung injury, and mortality (zero in this case) to provide an estimate of exposures that may result in more than minor behavioral responses potentially rising to the level of "take" pursuant to the ESA (balance of exposures greater than 175 dB). These unprocessed estimates do not take into account Navy standard operating procedures and mitigation activities that might reduce actual instances of harassment. From this we concluded that 14 leatherback sea turtles would be behaviorally harassed.

For an analysis of the likely responses of leatherback sea turtles to exposure to impulsive acoustic stressors, see our discussion in section 6.9.2.

6.11 Exposure and Response of Fish to Impulsive Acoustic Stressors – Training and Testing Activities

Training and testing activities using explosive ordnances that could affect ESA-listed fish will occur in the offshore portion of the Action Area and in inland waters. Table 114 lists the number of explosive ordnances used during training and testing 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 EIS (DoN 2015b).

Table 114. Explosive ordnances used during Navy training and testing activities in the NWTT Action Area that are likely to adversely affect fish.

Action (Representative Item)	Training /Testing	Location	Source Class: Net Explosive Weight (pounds, unless otherwise noted)	Ordnance per year (annual)
Gunnery Exercise Surface-to-Surface -- Ship	Training	Offshore area (>12nm[5])	E1: 0.1 – 0.25	48
Sound Underwater Signal buoy	Testing	Offshore area (45% at >12-50nm; 55% at >50nm)	E3: 0.5 – 2.5	72
Improved Extended Echo Ranging sonobuoy	Testing	Offshore area (20% at >12-50nm; 80% at >50nm)	E4: 2.5 – 5	70
Gunnery Exercise	Training	Offshore area (5% at >20-50nm; 95% at >50nm)	E5: 5 – 10	80
Torpedo (MK-46/54)	Testing	Offshore area (>50nm)	E8: 60 – 100	3
Missile Exercise	Training	Offshore area (>50nm)	E10: 250 – 500	4
Torpedo (MK-48)	Testing	Offshore area (>50nm)	E11: 500 – 650	3
Bombing Exercise	Training	Offshore area (outside the OCNMS[1]; all HE bombs >50nm)	E12: 650 – 1000	110 NEPM[2] (max of 10 HE[3] bombs)
Mine Neutralization – Explosive Ordnance Disposal (EOD)	Training	Inland waters (Crescent Harbor, WA)	E3: 0.5 – 2.5	3
Mine Neutralization – EOD	Training	Inland waters (Crescent Harbor, WA)	SWAG[4]: 15 grams	18
Mine Neutralization – EOD	Training	Inland waters (Hood Canal EOD Training Range)	E3: 0.5 – 2.5	3
Mine Neutralization – EOD	Training	Inland waters (Hood Canal EOD Training Range)	SWAG: 15 grams	18

[1] Olympic Coast National Marine Sanctuary

[2] Non-explosive practice munition

[3] High explosive

[4] Shock Wave Action Generator

[5] Nautical miles from shore

Below we analyze the effects of these explosions on ESA-listed fish species in the offshore and inland portions of the Action Area.

6.11.1.1 *Offshore area*

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 offshore environment because fish distribution is influenced by a number of environmental factors. Further, salmon and steelhead from different ESUs/DPSs (some ESA-listed and some not) are mixed in the open ocean environment (Bellinger et al. 2015), making it even more difficult to identify how many individuals of certain ESU/DPS may be affected by an activity at a specified location and time. Ocean salmon fisheries off the west coast of the United States are managed using a tool that predicts cohort-based stock abundance and time and area stock compositions (Bellinger et al. 2015). This allows managers to predict harvest levels from specific stocks of fish in order to maximize harvest, while maintaining conservation goals. However, a similar analysis was not possible for this Opinion because the Navy is not able to identify when and where specific activities will occur beyond the general terms described in Section 2 (e.g., >50nm offshore of Washington, Oregon, or California).

We estimated the number of salmon, steelhead, and eulachon (ESA-listed rockfish do not occur in the offshore portion of the Action Area) that may be killed or injured from detonations in the offshore environment using the acoustic threshold criteria established for this Opinion and available information on the offshore densities and distributions of the ESA-listed fish species considered in this Opinion. Below we summarize the information available regarding ESA-listed fish habitat use in offshore marine waters. 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.

6.11.1.1.1 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). Additionally, some Chinook salmon originating from Puget Sound remain within the Salish Sea throughout their lives, before returning to their natal systems as adults to spawn (Chamberlin et al. 2011). 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). Juvenile Chinook salmon are generally found within 55 km of the Washington, Oregon, and California coast, with the vast majority found less than 28 km offshore (Fisher and

Pearcy 1995; Pearcy and Fisher 1990; PFMC 2014). Bi et al. (2011) similarly found that while juvenile Chinook can occur in more offshore, deeper waters, they are most abundant on the inner continental shelf at a median depth of 55 meters. Bi et al. (2007) found that subyearling, yearling, and subadult Chinook salmon abundances were negatively correlated with depth. Peterson et al. (2010) found that during sampling of continental shelf and oceanic waters of the Pacific Northwest, juvenile salmonids were found almost exclusively in continental shelf waters. Coded wire tag recoveries from high seas fisheries and tagging programs suggest Chinook salmon do utilize areas outside the continental shelf. Commercial fisheries catch data suggest that most maturing Chinook salmon off the West Coast of the continental United States are found within 60 km of the coast (PFMC 2014). 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 off the Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids.

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 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. However, most recoveries occurred off the Washington coast and west of Vancouver Island. 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). Bellinger et al. (2015) also observed Central Valley spring-run Chinook caught in Central Oregon Coast fisheries. The California coastal ESU of Chinook salmon appears to be widely distributed along the mainland west coast of the U.S., with Bellinger et al. (2015) observed individuals from this ESU from Northern Oregon to south of Monterey Bay. This encompassed the author's entire study area, indicating the species' distribution could range even further.

A number of factors can drive variation in migratory pathways for Chinook salmon. Bi et al. (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 in waters off the Washington and Oregon coast. Sampling in 2003 and 2000 indicated large areas of habitat usage off the coast of Washington and Oregon, whereas the total area of habitat usage was lower in 2001, 2002, and 2004. This study also highlighted the variability in habitat selection by life stage with subyearling and subadult Chinook found closer to shore than yearling Chinook.

6.11.1.1.2 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). The degree to which juveniles migrate offshore appears to depend on the strength of upwelling, with strong upwelling years leading to wider dispersal, farther from shore (Pearcy 1992). Juvenile coho appear to be distributed further offshore than juvenile Chinook. Bi et al. (2011) observed juvenile coho to be most abundant at a median depth of 88 m and Chinook to be most abundant at a median depth of

58 m. While coho can be found further offshore, juvenile and maturing coho salmon are most abundant within 60 km off the coasts of Washington, Oregon, and California (PFMC 2014). The majority of juveniles are found within 37 km of the coast (Pearcy 1992; Pearcy and Fisher 1990). Similarly, Bi et al. (2007) found that abundance of yearling and subadult coho was negatively correlated with depth.

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.

Similar to Chinook, 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 inhabit 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). 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 in waters off the Washington and Oregon coast. Sampling in 2003 and 2000 indicated large areas of habitat usage off the coast of Washington and Oregon, whereas the area of habitat usage was lower in 2001, 2002, and 2004. This study also highlighted the variability in habitat selection by life stage with subadult coho found closer to shore than yearling coho.

6.11.1.1.3 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). Pearcy and Fisher (1990) observed the highest CPUE of juvenile chum inshore of 37 km, though some were caught over 55 km offshore. 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. Though 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), 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 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). Pearcy and Fisher (1990) noted that juvenile chum salmon were less abundant than either coho or chinook salmon off the Oregon and Washington coast. Chum salmon are known to be surface oriented, using the upper 20 meters 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 Oregon and Washington coast, Pearcy and Fisher (1990) did not find any evidence of large schools of juvenile salmonids. Data from Neave et al. (1976) indicated that catches of chum salmon in the NWTT Action Area off the coast of the continental US were lower than areas further to the north and further offshore (e.g., the Gulf of Alaska, the Bering Sea, and areas far offshore in the North Pacific).

6.11.1.1.4 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. (1976c) 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^{o}W and north of 48^{o}N (north of the NWTT Action Area). 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. 1976c). Pearcy and Fisher (1990) observed the highest CPUE of juvenile sockeye inshore of 37 km, though some were caught over 55 km offshore. They noted that, similar to juvenile chum salmon, juvenile sockeye salmon were less abundant than either coho or salmon off the Oregon and Washington coast. 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.

ESA-listed sockeye salmon from Redfish Lake, Idaho and Ozette Lake, Washington may occur in the offshore Action Area prior to and during their migration north of their natal watersheds. Tucker et al. (2009) caught juvenile sockeye salmon originating from the Columbia River (inclusive of Redfish Lake sockeye) and the Washington coast (inclusive of Lake Ozette sockeye) from May through August. By October, all juvenile sockeye from these areas had moved north to waters off Vancouver Island, southeast Alaska, and the Gulf of Alaska. Northward migration is thought to be rapid for juvenile sockeye. For example, Tucker et al.

(2009) recovered three coded wire tagged sockeye and determined that each had traveled between 1,800 and 2,500 km in under two months at an estimated travel rate of 40-48 km/day.

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.

6.11.1.1.5 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 1-3 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). Daly et al. (2014) analyzed NOAA Fisheries pelagic trawl survey data from off the coast of Oregon and Washington that targeted early marine phase juvenile salmonids to learn more about the distribution of steelhead in marine waters. Juvenile steelhead were consistently caught at the westernmost stations (>55km from shore) indicating a more offshore distribution for the species. Further, some of the steelhead that were caught in these far offshore waters had only been in saltwater for 1-3 days, indicating a rapid offshore migration (Daly et al. 2014). The results of Daly et al. (2014) were consistent with those presented by Pearcy and Fisher (1990), who found that catches of juvenile steelhead were generally highest at stations located more than 28 km from shore. 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.

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.

6.11.1.1.6 Eulachon

Eulachon appear to live near the ocean bottom, or on the continental shelf at depths most commonly of 20 to 200 m, though they may occur as deep as 500 m (NMFS 2006a). The species typically spends 3 to 5 years in saltwater before returning to freshwater to spawn.

6.11.1.1.7 Estimation of Take

As described in the preceding sections, juvenile and adult salmon are generally expected to occur on the continental shelf in the Action Area, but are less likely to occur in portions of the Action Area past the shelf break (generally defined as 200m depth contour along the west coast of Washington, Oregon, and California). Eulachon are also expected to occur in portions of the Action Area on the continental shelf, with less abundance at more offshore areas of the Action Area. Steelhead are expected to occur throughout the Action Area, as they will occur in the continental shelf portion of the Action Area during migration and in high seas portions of the Action Area while foraging. This information was used to determine the likelihood of co-occurrence between Navy explosive activities and each of these species. Explosions that may occur on the continental shelf are likely to adversely affect salmon, steelhead, and eulachon whereas explosions that occur off the continental shelf (<50 nm from shore) are only likely to adversely affect steelhead (Table 115).

Table 115. Expected co-occurrence of ESA-listed fish species with Navy explosive ordnance activity in the offshore Action Area. Unless otherwise noted, species is expected to co-occur with all ordnances from the specified explosive bin.

Species	Ordnance co-occurrence
Chinook – all ESUs	Testing: E3 (45% of ordnances), E4 (20% of ordnances) Training: E1, E5 (5% of ordnances)
Coho – all ESUs	Testing: E3 (45% of ordnances), E4 (20% of ordnances) Training: E1, E5 (5% of ordnances)
Columbia River chum ESU[1]	Testing: E3 (45% of ordnances), E4 (20% of ordnances) Training: E1, E5 (5% of ordnances)
Snake River sockeye ESU[2]	Testing: E3 (45% of ordnances), E4 (20% of ordnances) Training: E1, E5 (5% of ordnances)
Southern DPS eulachon	Testing: E3[4] (45% of ordnances), E4[4] (20% of ordnances) Training: E1[5], E5[5] (5% of ordnances)
Steelhead – Puget Sound, Middle Columbia River, Upper Willamette River, Lower Columbia River, Snake River Basin, Upper Columbia River, Northern California DPSs	Testing: E3, E4, E8, E11 Training: E1, E5, E10, E12
Steelhead – California Central Valley, Central California, South-Central California DPSs[3]	Testing: E3 (55% of ordnances), E4 (80% of ordnances), E8, E11 Training: E1, E5 (95% of ordnances), E10, E12

[1] Hood Canal summer-run chum would not be expected to co-occur with offshore explosive activity as the species migrates north and west following their entrance into saltwater (Byron and Burke 2014; Quinn 2005). Following this migratory path, they would be north of the Action Area before they would be far enough offshore to co-occur with Navy explosive activities.

[2] Similar to Hood Canal summer-run chum, Lake Ozette sockeye are expected to migrate north and west following their entrance into saltwater (Byron and Burke 2014; Quinn 2005). Since the Navy will not conduct explosive activities in the Olympic Coast National

Marine Sanctuary and considering this migratory path, Lake Ozette sockeye would be expected to be north of the Action Area before they would be far enough offshore to co-occur with Navy explosive activities.

[3] Steelhead from these DPSs originate from rivers south of the Action Area. Because, steelhead migrate rapidly through estuaries, bypassing coastal migration routes of other salmonids, moving into oceanic offshore feeding grounds (Daly et al. 2014; Quinn and Myers 2004), we would not expect individuals from these DPSs to co-occur with explosions that may occur on the continental shelf. However, based on their life history strategy of maturing in oceanic waters throughout the North Pacific (Light et al. 1989), we would expect individuals from these DPSs to be present in portions of the Action Area off the continental shelf (> 50nm from shore).

[4] Detonations in these explosive bins are expected to occur at > 30 meter water depths and therefore, would be expected to co-occur with eulachon.

[5] Detonations in these explosive bins will occur at the surface. Based on the small range to effects for eulachon in these explosive bins and the depth at which this species would be expected to occur, we do not expect eulachon would experience injurious underwater sound levels from detonations in this explosive bin.

For each ESA-listed salmon and eulachon ESU and steelhead DPS, we estimated a density of animals in the offshore 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 contained the latest information on salmon and steelhead abundance (i.e., outmigrants and adults)(NMFS 2015). This information is presented in Table 116.

Table 116. Summary of estimated annual abundance of salmonids (NMFS 2015). Abundance estimates for each ESU and DPS are divided into natural, listed hatchery intact adipose, and listed hatchery adipose clip[19].

Species	Life Stage	Natural	Listed Hatchery Intact Adipose[1]	Listed Hatchery Adipose Clip[1]
Sacramento River winter-run Chinook	Adult	2,023	-	83
	Smolt	161,840	-	193,900
Central Valley spring-run Chinook	Adult	7,464	-	6,414
	Smolt	1,552,885	-	2,178,601
California Coastal Chinook	Adult	7,144	-	-
	Smolt	1,298,065	-	-
Snake River fall Chinook	Adult[1]	14,438	30,475	
	Smolt	570,821	3,780,129	3,076,642
Snake River spring/summer Chinook	Adult[1]	20,422	60,058	-
	Smolt	1,454,727	1,164,078	4,381,302
Lower Columbia River Chinook	Adult[1]	13,594	22,868	-
	Smolt	13,271,270	1,070,253	35,337,495
Upper Willamette River Chinook	Adult[1]	11,061	38,135	-
	Smolt	1,813,726	42,420	6,006,713
Upper Columbia River spring Chinook	Adult[1]	3,170	5,887	-
	Smolt	570,965	931,815	504,620
Puget Sound Chinook	Adult	18,127	11,089	-
	Smolt	2,337,280	5,992,150	36,617,500
Hood Canal summer run	Adult	17,556	3,452	-

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[19] ESA take prohibitions do not apply to hatchery fish with clipped adipose fins from threatened ESUs/DPSs. Additionally, hatchery fish are generally considered to be of less conservation value than individuals from the natural population NMFS. 2015. Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion: Continued Prosecution of Fisheries Research Conducted and Funded by the Southwest Fisheries Science Center; Issuance of a Letter of Authorization under the Marine Mammal Protect Act for the Incidental Take of Marine Mammals Pursuant to those Research Activities; and Issuance of a Scientific Research Permit under the Endangered Species Act for Directed Take of ESA-Listed Salmonids. W. C. Region, editor..

Species	Life Stage	Natural	Listed Hatchery Intact Adipose[1]	Listed Hatchery Adipose Clip[1]
chum	Smolt	3,072,420	275,000	-
Columbia River chum	Adult	12,239	428	-
	Smolt	2,978,550	391,973	-
Central California Coast coho	Adult	1,294	-	327
	Smolt	90,580	-	225,825
Southern Oregon/Northern California Coast coho	Adult	10,193	6,368	1,526
	Parr	1,026,707	575,000	200,000
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
Snake River sockeye	Adult	-	-	1,373*
	Smolt	15,560	-	124,767
South-Central California steelhead	Adult	695	-	-
	Smolt	79,057	-	-
Central California Coast steelhead	Adult	1,429	-	3,866
	Smolt	162,549	-	648,891
California Central Valley steelhead	Adult	1,374	-	3,359
	Smolt	156,293	-	1,600,653
Northern California steelhead	Adult	4,286	-	-
	Smolt	487,533	-	-
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	-	-

Species	Life Stage	Natural	Listed Hatchery Intact Adipose[1]	Listed Hatchery Adipose Clip[1]
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, and eulachon typically spend three to five years at sea before returning to freshwater to spawn. Information is not available for all ESA-listed salmon and eulachon ESUs 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; four years for eulachon). 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, steelhead, and eulachon 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 north-south oceanic distribution for Chinook was based on the results presented in Weitkamp (2010), which used coded-wire-tags to estimate the distribution of Chinook salmon from various regions along the west coast of North America. For coho, the north-south oceanic distribution was based on Weitkamp and Neely (2002) which used similar methodology. Since Chinook and coho primarily reside on the continental shelf, we used the shelf break as the westward boundary of these species' distribution (the shelf break was defined as the 200 meter depth contour; (Landry and Hickey 1989)). Similar studies were not available for chum, sockeye, and steelhead. Chum geographic distribution was based on the ocean migration of the species from British Columbia, Washington, and Oregon, as determined from tagging data and presented in Neave et al. (1976). The migration pattern described in Neave et al. (1976) did not include information on individuals found immediately offshore of their river of origin in Oregon and Washington. 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). Therefore, we added the area of the continental shelf from each ESU's river of origin north to the mouth of Puget Sound (the area southernmost point where Neave et al. (1976) presented tagging data). 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, moving north and west along the coast, and having moved offshore by the end of their first ocean year (Byron and Burke 2014; Quinn 2005). For 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 winter, where they were most geographically condensed. 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, summer, and fall. Since we do not know when Navy detonations will occur, 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., all detonations occurring in winter when the species is most densely distributed). However, it is more likely that Navy explosive activities will be spread out more evenly throughout the seasons during times when steelhead are also more widely dispersed. Based on the distribution information presented above, we used ArcMap version 10.2.1 (ESRI, Redlands, CA) to determine the area (km2) of habitat in the offshore portion of the Action Area that each species ESU or DPS is known to occupy (Table 117).

Table 117. Habitat area (distribution) used for each ESU/DPS (km^2) in the offshore marine environment.

Species	Marine Habitat Area (km^2)
Sacramento River winter-run Chinook	123,717
Central Valley spring-run Chinook	123,717
California Coastal Chinook	64,316
Snake River fall Chinook	657,628
Snake River spring/summer Chinook	657,628
Lower Columbia River Chinook	562,179
Upper Willamette River Chinook	634,343
Upper Columbia River spring Chinook	657,628
Puget Sound Chinook	241,626
Columbia River chum	4,414,073
Central California Coast coho	49,908
Southern Oregon/Northern California Coast coho	181,607
Oregon Coast coho	131,699
Lower Columbia River coho	106,339
Snake River sockeye	4,414,073
Steelhead (all DPSs)	6,083,400

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 for the offshore environment are presented in Table 118, Table 119, and Table 120. Table 118 gives estimates for hatchery fish with the adipose fin intact, Table 119 gives estimates for hatchery fish with an adipose clip, and Table 120 gives estimates for the number of wild fish. All estimates are the annual number of injured or killed fish.

Table 118. Estimated annual number of ESA-listed salmonids (hatchery fish w/adipose fin intact) that would die or be injured by explosive activities in the offshore environment.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook	Adult	Sac River winter run - E	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Central valley spring run - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	California coastal - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Snake River fall - T	0.25	0.98	0.06	0.27	0.32	1.25
	Juvenile		40.03	94.57	9.40	30.86	49.43	125.43
	Adult	Snake River spring/summer - T	0.50	1.93	0.12	0.53	0.62	2.46
	Juvenile		12.33	29.12	2.89	9.50	15.22	38.63
	Adult	Lower Columbia River - T	0.22	0.86	0.05	0.24	0.28	1.10
	Juvenile		13.26	31.32	3.11	10.22	16.37	41.54
	Adult	Upper Willamette River - T	0.33	1.27	0.08	0.35	0.41	1.62
	Juvenile		0.47	1.10	0.11	0.36	0.58	1.46
	Adult	Upper Columbia River spring - E	0.05	0.19	0.01	0.05	0.06	0.24
	Juvenile		9.87	23.31	2.32	7.61	12.19	30.92
	Adult	Puget Sound - T	0.25	0.97	0.06	0.27	0.31	1.24
	Juvenile		172.71	408.00	40.55	133.15	213.26	541.15
Coho	Adult	Central Calif coast - E	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	S. Oregon/N. Calif coast - T	0.15	0.53	0.04	0.15	0.19	0.68
	Juvenile		20.87	49.66	4.86	16.10	25.73	65.76
	Adult	Oregon coast - T	0.06	0.20	0.01	0.06	0.07	0.26
	Juvenile		3.00	7.15	0.70	2.32	3.70	9.46
	Adult	Lower Columbia River - T	8.51	29.34	2.00	8.44	10.50	37.79
	Juvenile		18.59	44.24	4.33	14.34	22.92	58.58
Chum	Adult	Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.81	1.75	0.19	0.50	0.99	2.25
Sockeye	Adult	Snake River -	0.00	0.00	0.00	0.00	0.00	0.00

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Juvenile	E	0.00	0.00	0.00	0.00	0.00	0.00
Steelhead	Adult	South-Central California - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Central Calif - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	California Central Valley - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Northern Calif - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Upper Columbia River - E	0.06	0.15	0.03	0.05	0.08	0.20
	Juvenile		0.78	1.69	0.36	0.67	1.14	2.36
	Adult	Snake River basin - T	1.03	2.67	0.46	0.90	1.49	3.58
	Juvenile		4.30	9.32	2.00	3.71	6.30	13.03
	Adult	Lower Columbia River - T	0.17	0.44	0.08	0.15	0.25	0.59
	Juvenile		0.01	0.02	0.00	0.01	0.02	0.03
	Adult	Upper Willamette River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Middle Columbia River - T	0.02	0.05	0.01	0.02	0.03	0.07
	Juvenile		1.89	4.10	0.88	1.63	2.77	5.72
	Adult	Puget Sound - T	0.01	0.02	0.00	0.01	0.01	0.03
	Juvenile		0.28	0.61	0.13	0.24	0.42	0.86

Table 119. Estimated annual number of ESA-listed salmonids (hatchery fish w/adipose clip) that would die or be injured by explosive activities in the offshore environment.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook	Adult	Sac River winter run - E	0.00	0.01	0.00	0.00	0.00	0.02
	Juvenile		10.92	25.79	2.56	8.41	13.48	34.20
	Adult	Central valley spring run - T	0.28	1.10	0.07	0.30	0.35	1.40
	Juvenile		122.64	289.71	28.80	94.55	151.44	384.26
	Adult	California coastal - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Snake River fall - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		32.58	76.97	7.65	25.12	40.23	102.09
	Adult	Snake River spring/summer - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		46.40	109.61	10.89	35.77	57.29	145.38
	Adult	Lower Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		437.77	1034.15	102.79	337.49	540.56	1371.63
	Adult	Upper Willamette River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		65.95	155.79	15.48	50.84	81.43	206.63
	Adult	Upper Columbia River spring - E	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		5.34	12.62	1.25	4.12	6.60	16.74
	Adult	Puget Sound - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		1055.43	2493.25	247.81	813.65	1303.24	3306.90
Coho	Adult	Central Calif coast - E	0.03	0.10	0.01	0.03	0.04	0.13
	Juvenile		29.82	70.97	6.95	23.01	36.77	93.98
	Adult	S. Oregon/N. Calif coast - T	0.04	0.13	0.01	0.04	0.05	0.16
	Juvenile		7.26	17.27	1.69	5.60	8.95	22.87
	Adult	Oregon coast - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Lower Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		535.34	1273.93	124.75	413.08	660.09	1687.00
Chum	Adult	Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
Sockeye	Adult	Snake River - E	0.00	0.01	0.00	0.00	0.00	0.01
	Juvenile		0.18	0.44	0.04	0.14	0.23	0.58

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Steelhead	Adult	South-Central California - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Central Calif - T	0.02	0.05	0.01	0.02	0.04	0.08
	Juvenile		2.28	4.69	1.31	2.44	3.59	7.13
	Adult	California Central Valley - T	0.02	0.05	0.01	0.02	0.03	0.07
	Juvenile		5.61	11.57	3.24	6.02	8.85	17.59
	Adult	Northern Calif - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Upper Columbia River - E	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		2.92	6.32	1.35	2.52	4.27	8.84
	Adult	Snake River basin - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		13.62	29.53	6.33	11.74	19.94	41.27
	Adult	Lower Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		4.54	9.85	2.11	3.92	6.65	13.77
	Adult	Upper Willamette River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.00	0.00	0.00	0.00	0.00	0.00
	Adult	Middle Columbia River - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		1.54	3.33	0.71	1.33	2.25	4.66
	Adult	Puget Sound - T	0.00	0.00	0.00	0.00	0.00	0.00
	Juvenile		0.69	1.50	0.32	0.60	1.01	2.09

Table 120. Estimated annual number of ESA-listed salmonids (natural fish) that would die or be injured by explosive activities in the offshore environment.

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
Chinook	Adult	Sac River winter run - E	0.09	0.35	0.02	0.10	0.11	0.44
	Juvenile		9.11	21.52	2.14	7.02	11.25	28.55
	Adult	Central valley spring run - T	0.33	1.27	0.08	0.35	0.41	1.63
	Juvenile		87.42	206.50	20.53	67.39	107.94	273.90
	Adult	California coastal - T	0.61	2.34	0.15	0.65	0.75	2.98
	Juvenile		140.56	332.05	33.00	108.36	173.56	440.41
	Adult	Snake River fall - T	0.12	0.46	0.03	0.13	0.15	0.59
	Juvenile		6.05	14.28	1.42	4.66	7.46	18.94
	Adult	Snake River spring/summer - T	0.17	0.66	0.04	0.18	0.21	0.84
	Juvenile		15.41	36.39	3.62	11.88	19.02	48.27
	Adult	Lower Columbia River - T	0.13	0.51	0.03	0.14	0.16	0.65
	Juvenile		164.41	388.38	38.60	126.75	203.01	515.13
	Adult	Upper Willamette River - T	0.10	0.37	0.02	0.10	0.12	0.47
	Juvenile		19.91	47.04	4.68	15.35	24.59	62.39
	Adult	Upper Columbia River spring - E	0.03	0.10	0.01	0.03	0.03	0.13
	Juvenile		6.05	14.28	1.42	4.66	7.47	18.95
	Adult	Puget Sound - T	0.41	1.59	0.10	0.44	0.51	2.02
	Juvenile		67.37	159.14	15.82	51.94	83.19	211.08
Coho	Adult	Central Calif coast - E	0.11	0.39	0.03	0.11	0.14	0.50
	Juvenile		11.96	28.47	2.79	9.23	14.75	37.70
	Adult	S. Oregon/N. Calif coast - T	0.24	0.84	0.06	0.24	0.30	1.08
	Juvenile		37.26	88.67	8.68	28.75	45.94	117.42
	Adult	Oregon coast - T	6.35	21.90	1.49	6.30	7.84	28.20
	Juvenile		674.13	1604.19	157.09	520.16	831.22	2124.35
	Adult	Lower Columbia River - T	0.45	1.54	0.11	0.44	0.55	1.99
	Juvenile		52.01	123.76	12.12	40.13	64.13	163.90
Chum	Adult	Columbia River - T	0.02	0.08	0.01	0.02	0.03	0.10
	Juvenile		6.12	13.28	1.42	3.79	7.53	17.07
Sockeye	Adult	Snake River -	0.00	0.00	0.00	0.00	0.00	0.00

			Testing		Training		Total	
Species	**Life stage**	**ESU/DPS**	**Mortality**	**Injury**	**Mortality**	**Injury**	**Mortality**	**Injury**
	Juvenile	E	0.02	0.05	0.01	0.02	0.03	0.07
Steelhead	Adult	South-Central California - T	0.00	0.01	0.00	0.00	0.01	0.01
	Juvenile		0.28	0.57	0.16	0.30	0.44	0.87
	Adult	Central Calif - T	0.01	0.02	0.00	0.01	0.01	0.03
	Juvenile		0.57	1.18	0.33	0.61	0.90	1.79
	Adult	California Central Valley - T	0.01	0.02	0.00	0.01	0.01	0.03
	Juvenile		0.55	1.13	0.32	0.59	0.86	1.72
	Adult	Northern Calif - T	0.03	0.08	0.01	0.03	0.05	0.11
	Juvenile		2.16	4.68	1.00	1.86	3.16	6.54
	Adult	Upper Columbia River - E	0.02	0.05	0.01	0.02	0.03	0.07
	Juvenile		1.27	2.75	0.59	1.09	1.86	3.84
	Adult	Snake River basin - T	0.34	0.89	0.15	0.30	0.49	1.19
	Juvenile		6.20	13.44	2.88	5.34	9.08	18.78
	Adult	Lower Columbia River - T	0.08	0.21	0.04	0.07	0.12	0.29
	Juvenile		1.98	4.30	0.92	1.71	2.90	6.01
	Adult	Upper Willamette River - T	0.04	0.12	0.02	0.04	0.06	0.15
	Juvenile		0.96	2.07	0.44	0.82	1.40	2.90
	Adult	Middle Columbia River - T	0.18	0.46	0.08	0.16	0.26	0.62
	Juvenile		2.39	5.19	1.11	2.07	3.51	7.26
	Adult	Puget Sound - T	0.10	0.26	0.04	0.09	0.15	0.35
	Juvenile		7.39	16.02	3.43	6.37	10.82	22.39

6.11.1.1.8 Estimation of Take for Eulachon

Eulachon appear to live near the ocean bottom, or on the continental shelf at depths most commonly of 20 to 200 m (NMFS 2006a). Because most eulachon are expected to occur on the continental shelf, we assessed impacts to eulachon for the detonations that may occur over the continental shelf and in water deep enough where they may be affected by explosions as determined by the range to injury and mortality values (explosions in bins E3, E4). We do not expect the eulachon to be affected by explosions off the shelf break due to the depth at which the animals will occur in these areas (near the ocean bottom) and sound propagation loss.

To estimate the number of eulachon that may be killed or injured from Navy explosive activities in the offshore environment, we used a similar methodology as described for estimating salmonid take in the offshore environment. First, we determined the average density of eulachon in the offshore environment. We determined that the southern DPS of eulachon has a marine distribution area of 1,183,304 km^2. When determining density, we assumed equal distribution throughout their area of habitat since information does not suggest we should expect higher densities of these species in some areas versus others. We then multiplied this density by the area of mortality and injury for each explosive bin that would be expected to co-occur with the species (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). The latest estimate of the population abundance of the southern DPS of eulachon was 185,000,000 spawners estimated in the Columbia River in 2014. Because we do not have estimates of eulachon abundance in marine waters, the number of spawners in the Columbia River was used as a proxy for abundance in the oceanic environment. We multiplied the number of returning adults 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., four years for eulachon). 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 (generally how eulachon abundance and trends are tracked). Results of these calculations are presented in Table 121.

Table 121. Estimated annual number of ESA-listed eulachon from the Southern DPS that would die or be injured by explosive activities in the offshore environment.

Species	Testing		Training		Total	
	Mortality	Injury	Mortality	Injury	Mortality	Injury
Eulachon - Southern DPS	0.33	0.87	0	0	0.33	0.87

6.11.1.2 *Inland waters*

Detonations in inland waters (Puget Sound) will occur in Crescent Harbor and Hood Canal. In Hood Canal, charges are generally placed a few feet (1-3) above the seafloor, in approximately 27 feet of water (Andrea Balla-Holden, personal communication, August 14, 2015). However, surface detonations are possible. In Crescent Harbor, E3 charges are generally placed in water depths greater than 40 feet, in areas with no bottom vegetation. The Crescent Harbor EOD range is located 1000 meters offshore in order to minimize impacts to nearshore salmonids (Andrea Balla-Holden, personal communication, October 2015). This charge placement (offshore and at depth) was determined, through discussions with NMFS and USFWS prior to this consultation, to be least impactful to listed resources (John Mosher, personal communication, August 14, 2015). Prior to any detonation in inland waters there is a significant amount of human in-water activity in the area that would likely deter fish from being in the immediate vicinity of the explosion (John Mosher, personal communication, August 14, 2015).

Below we present information on the ESA-listed fish species considered in this Opinion that are likely to occur in Puget Sound, their likelihood of co-occurring with detonations in Crescent Harbor and Hood Canal, and the methods we used to estimate the number of fish that may be killed or injured by Navy activities. Several fish species considered in this Opinion are not discussed in this section because they are not expected to occur in Puget Sound (e.g., Lower Columbia River Chinook salmon ESU) or because they occur in extremely low densities in Puget Sound where the likelihood of an individual of these species co-occurring with Navy EOD activities in Puget Sound is so low as to be discountable (e.g., green sturgeon).

6.11.1.2.1 Chinook

Juvenile and adult Puget Sound Chinook would likely be affected by Navy EOD activities at both Crescent Harbor and Hood Canal.

6.11.1.2.1.1 Crescent Harbor

Crescent Harbor is located at the north end of Saratoga Passage, just west of Skagit Bay. There is no known Chinook salmon spawning habitat in streams flowing into Crescent Harbor (http://apps.wdfw.wa.gov/salmonscape/map.html), but Puget Sound Chinook from the Skagit, Stillaguamish, and Snohomish Rivers (and to a lesser degree, rivers from southern Puget Sound) would be expected to occur in the Harbor (NMFS 2012a). Additionally, an unknown number of Puget Sound Chinook which mature solely in the Salish Sea and do not migrate into the North Pacific Ocean (i.e., blackmouths), may spend at least a portion of their time foraging in Crescent Harbor. Survey and research sampling within Crescent Harbor have indicated juvenile Puget Sound Chinook use both the intertidal and offshore areas of the Harbor with smaller individuals generally found in intertidal habitat and larger individuals found in the offshore (E. Beamer, Skagit Coop, personal communication, D. Heatwole, Skagit Coop, personal communication, cited in (NMFS 2012a)). This is consistent with Puget Sound Chinook life history information which suggests that most juveniles enter estuaries and occupy nearshore habitats primarily during the spring and early summer, but that by mid-summer, most Chinook have migrated to more offshore habitats (Duffy et al. 2010b). Beamish et al. (1998) studied intra-annual changes in the abundance of Chinook salmon in Puget Sound and collected the majority of juveniles in rope trawl tows at depths ranging from 0 to 15 meters. Beamer et al. (2003) studied wild Chinook salmon pocket estuary, nearshore, and offshore habitat use in Skagit Bay. The authors found juvenile Chinook are over 100 times more abundant in pocket estuary habitat (0.5 meter max depth) and 10 times more abundant in nearshore habitat (2.8 meter max depth) than in more offshore areas adjacent to nearshore habitat. Based on the range to injury and mortality for the charges proposed for use in Crescent Harbor and the distance of the site from these nearshore habitats (EOD range is in waters greater than 40 feet deep and 1,000 meters offshore), we do not expect injurious sound levels to be able to reach nearshore and pocket estuary habitats (as described in (Beamer et al. 2003)) in Crescent Harbor that would be expected to hold the highest concentrations of fish.

Rice et al. (2011) studied the use of the Puget Sound neritic zone (surface waters overlaying the sublittoral zone; similar habitat to the location of the Crescent Harbor EOD range) in greater Puget Sound. They sampled at various sites in the Whidbey Basin, including Crescent Harbor. In the Whidbey Basin, 73 percent of the fish captured were unmarked, which is consistent with there being more wild fish in this portion of the Sound than in areas further south. Juvenile Chinook density in the Whidbey Basin was highest in July and August with approximately 15 unmarked and 6 marked fish/hectare.

To estimate the number of juvenile Chinook salmon that we would expect to be killed or injured from EOD activity at Crescent Harbor, we used a similar methodology as described for estimating take of ESA-listed salmonids in the offshore portion of the Action Area. We used the juvenile density information presented above for Whidbey Basin from Rice et al. (2011). In order to be conservative and because we do not know when the Navy will be conducting EOD activities, we assumed all detonations would occur during July and August, when we would expect the highest densities of juvenile Chinook in the EOD range. Once we derived our density estimates, we then multiplied these densities by the area of mortality and injury as estimated by the Navy (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). Results of these calculation are presented in Table 122.

For adults, we do not have open water density information for the EOD site in Crescent Harbor or any similar areas adjacent to Crescent Harbor. However, information suggests that adult Chinook density in Crescent Harbor is likely much lower than would be expected in some other areas and habitat types in the Sound (e.g., river mouths, deltas, and estuaries of the Skagit and Snohomish Rivers) because there are no documented spawning rivers in the immediate vicinity of the EOD range. Therefore, adult salmon are unlikely to stage in large numbers in the Harbor, but available information suggests that very low numbers of migrating adults could occur within the range to effects for underwater detonations for very short periods of time (though Crescent Harbor is not a direct migratory route from the ocean to any Chinook spawning rivers). Given the very low densities of adults that may occur in the area, the short amount of time any individuals would be likely to spend in Crescent Harbor, and the low number of explosion events that will occur throughout the year, we do not expect more than one adult Puget Sound Chinook salmon may be injured or killed each year from Navy explosive activities at Crescent Harbor.

6.11.1.2.1.2 Hood Canal

Hood Canal has extant populations of Puget Sound Chinook in the Skokomish River watershed and Mid Hood Canal region (including spawning populations in the Hamma Hamma, Duckabush, and Dosewallips watersheds) (Ford 2011c). Hood Canal Chinook typically spawn mid-September to late October (Commission 2003). All juvenile Chinook emigrating from or adults immigrating to these watersheds must pass by the Hood Canal EOD site. Additionally, an unknown number of Puget Sound Chinook which mature solely in the Salish Sea and do not migrate into the North Pacific Ocean, are likely to spend at least a portion of their time foraging in proximity to the Hood Canal EOD range (e.g., (Tribe and Wildlife 2007)). We do not have information on the marine residence times of individual Hood Canal chinook, but based on life history information, we assume juvenile Chinook spend significantly more time rearing in Hood Canal nearshore areas than steelhead or chum. Generally, Puget Sound Chinook juveniles emigrate from freshwater natal areas to estuarine and nearshore habitats from January through April as fry, and from April through early July as larger subyearlings (NMFS 2011h).

As indicated in the *Status of Listed Resources* section of this Opinion, the number of natural origin spawners returning to Hood Canal Rivers averaged 311 individuals from 2008-2012. Over the same time period, hatchery origin spawners averaged 1,226 individuals per year. NMFS (2015) calculated that this number of returning spawners would result in a total of 122,960 outmigrants per year.

In order to minimize impacts to adult Puget Sound Chinook from Hood Canal, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (>0.5-2.5 lb), and only conduct training with one ounce or less charges during August, September, and October. Because the Navy could not rule out conducting explosions using these larger size charges during the adult migration and spawning period and in order to be conservative (i.e., assess the maximum effect to the species which could be considered reasonable), we assumed E3 detonations could occur during the adult migration period.

To estimate the number of Chinook salmon that we would expect to be killed or injured from EOD activity at Hood Canal, we first estimated the average density of Chinook salmon of each life stage in Hood Canal. We do not have EOD range-specific density information for Chinook salmon at Hood Canal. Since there are no documented Chinook spawning rivers in the immediate vicinity of the Hood Canal EOD range, adult and juvenile Chinook salmon are unlikely to stage in large concentrations near the range. This information suggests Chinook density at the EOD ranges is likely lower than would be expected in some other areas and habitat types in Hood Canal (e.g., Hood Canal salmon spawning river mouths/estuaries). Most individuals that will occur in close proximity to the range would be expected to be migrating through the range, though some juveniles could reside in higher numbers in nearshore areas within the injury/mortality range of the explosions. In order to be conservative, we assumed equal distribution of each life stage throughout the Canal since information does not suggest we should expect higher densities of these species at the EOD ranges than in other areas of the Canal. In order to derive our density estimates, we divided the number of adult or juvenile fish expected in the Hood Canal[20] by the area of the Canal (Hood Canal has a surface area of 358.4 km^2). We then multiplied this density by the area of mortality and injury as estimated by the Navy (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). Results of these calculation are presented in Table 122.

We recognize that this methodology likely overestimates the number of adult Puget Sound Chinook from Hood Canal that will die or be injured from Navy EOD activities. Most notably, our analysis and resulting take estimates assume all detonations (E3 and SWAG charges) will occur when adult Puget Sound Chinook are in the Canal. While we did this to be conservative and to analyze the highest potential effect of the activity on ESA-listed species that could be considered reasonable, it is also possible that EOD activity may be spread more evenly throughout the year and occur during times when the species is not in Hood Canal or near the EOD range, particularly since the Navy will only conduct detonations with the larger charges during their migration period if necessitated by readiness requirement. If the Navy were to conduct some or all EOD activities during times of the year when adult Puget Sound Chinook are not present, the number of individuals that would die or be injured as a result of Navy EOD activities would be expected to be much lower. Also, this methodology does not consider habitat selection within the Hood Canal by the species. Adult fish in close proximity to the EOD range that may be affected would only be migrating through the area since there are no spawning rivers in immediate proximity to the range and would not be staging to enter freshwater spawning rivers at this location. Further, juveniles are expected at lower densities in the EOD range than in other areas of the Canal, where individuals of this life stage would be more likely to congregate (e.g., tidal channels, mudflats, marshes, and eelgrass meadows). These habitat types are not present in the Hood Canal EOD range. Additionally, it is worth noting that Navy monitoring of

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[20] Without better information on residence time of outmigrating juvenile or returning adult Chinook salmon in Hood Canal and since Chinook are thought to spend significantly more time rearing in estuarine and nearshore waters before migrating into open water habitats (months instead of days), we assumed all Chinook salmon originating from and returning to Hood Canal rivers could reside in the Canal concurrently. This likely overestimates Chinook salmon abundance in the Canal since some proportion of juveniles will spend less time in the Canal than others and some will die from other causes (e.g., predation) while migrating through the Canal. Additionally, adult migration is staggered such that some returning fish will have likely already entered freshwater before others have entered Hood Canal. This methodology results in a conservative estimate of take.

previous similar UNDETs at Hood Canal have never documented fish kills of the magnitude estimated to occur (Chip Johnson, personal communication, October 2015). However, Navy monitoring may not detect all fish that would die from an explosive event as some may not float to the surface of the water in order to be detected by visual observation and some may be captured by foraging birds prior to detection. Additionally, injured fish that may later die would not be expected to be detected as they also may not immediately float to the surface of the water to be detected by visual observation.

6.11.1.2.2 Steelhead

Juvenile and adult Puget Sound steelhead would likely be affected by Navy training activities using explosives at both Crescent Harbor and Hood Canal.

6.11.1.2.2.1 Crescent Harbor

Puget Sound steelhead abundance in Crescent Harbor is likely very low. Steelhead are seldom observed in Puget Sound at any life stage (NMFS 2012a). Steelhead do not spawn on Whidbey Island, including streams flowing into Crescent Harbor (http://apps.wdfw.wa.gov/salmonscape/map.html). As mentioned previously, steelhead do not rear in estuarine or nearshore areas for long periods of time, but migrate quickly through these areas into oceanic waters of the North Pacific Ocean. Puget Sound steelhead generally migrate to the ocean from mid-April through late May when they are typically 2 to 3 years old. At this age and size, they are not dependent on shallow nearshore areas, and they are well dispersed throughout Puget Sound. Therefore, any outmigrating steelhead occurring in Crescent Harbor, would likely only be present for a short period of time as they migrate towards open water. Similarly, Crescent Harbor is not in direct route to steelhead spawning rivers on the eastern side of Skagit Bay from oceanic waters, so any adults that may migrate through Crescent Harbor would likely only occur in the harbor for a brief period of time. This information leads us to conclude that steelhead density in Crescent Harbor is expected to be much lower than in other locations in Puget Sound (e.g., in marine waters directly offshore from their natal rivers). However, a few individuals could occur in the Harbor while migrating.

We do not have information available to estimate the density of juvenile or adult steelhead that may migrate through the EOD site at Crescent Harbor. However, the information presented above suggests that very low numbers of migrating juveniles and adults could occur within the range to effects for underwater detonations for very short periods of time. Given the very low densities of juveniles and adults that may occur in the area, the short amount of time any individuals would be likely to spend in Crescent Harbor, and the low number of explosion events that will occur throughout the year, we do not expect more than one juvenile and one adult Puget Sound may be injured or killed each year from Navy explosive activities at Crescent Harbor.

6.11.1.2.2.2 Hood Canal

Steelhead spawn in rivers flowing into the Hood Canal, though not in rivers adjacent to the Hood Canal EOD range. Immigrating and emigrating steelhead must pass by the Hood Canal EOD range on their way to their natal rivers to spawn or on their way out to oceanic waters in the North Pacific to mature. Typically, Puget Sound steelhead juveniles emigrate from their natal rivers as two-year olds from March through June (NMFS 2011h). Moore et al. (2010b) found that acoustically tagged steelhead smolts from Hood Canal rivers migrated from their natal river mouths to areas outside of the Hood Canal over an average of 15 to 17 days. Therefore, based on emigration timing and residence time in Hood Canal, most steelhead smolts will have left Hood Canal by mid-July.

As indicated in the *Status of Listed Resources* section of this Opinion, the number of natural origin spawners returning to Hood Canal Rivers averaged 967 individuals from 2009 to 2013. NMFS (2015) calculated that this number of returning spawners would result in a total of 109,997 outmigrants per year.

We estimated the number of juvenile Puget Sound steelhead that may be injured or killed from detonations at the Hood Canal EOD site using similar methodology as described for estimating offshore take. To estimate steelhead juvenile density at the EOD site in Hood Canal, we first determined the average number of fish leaving freshwater from Hood Canal rivers per day by dividing the annual number of outmigrating juveniles by the number of days in the migration period (March through June; 122 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (17 days, as described above). This results in the average number of juvenile Puget Sound steelhead we would expect in the Canal on a given day during the middle portion of the out-migration period (during first and last 17 days of the out-migration period, fewer individual steelhead would be expected in the Canal). We then divide this number by the area of Hood Canal (358.4 km^2) to determine juvenile steelhead density during the outmigration period. To be conservative, we assumed all detonations will occur during the outmigration period as the Navy is unable to predict when detonations will occur. For adults, we did not have information to estimate the amount of time each fish would be expected to reside in Hood Canal before entering freshwater to spawn. Therefore, we assumed the entire annual run could occur in the Canal on a given day. We then divided this number by the area of Hood Canal to estimate adult density in the Canal on a given day during the run. Once we had juvenile and adult densities, we multiplied these numbers by the area of injury and mortality for each detonation, then multiplied this number by the number of detonations. (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).

6.11.1.2.3 Chum

Hood Canal summer-run chum juveniles migrate from freshwater into estuary habitat generally from the first week in February through the second week in April. Estimated peak emergence occurs around March 22 (Wildlife and Tribes 2000). Upon initially arriving in the estuary, chum inhabit nearshore waters, at depths ranging between 1.5 and 5 meters (Wildlife and Tribes 2000). At this initial stage, they are thought to concentrate in the top few meters of the water column, forming loose aggregations during daylight hours. Once fry reach a size of approximately 45 to 50 mm, they tend to move rapidly into deeper offshore areas, migrating approximately 7 to 14 km/day (Wildlife and Tribes 2000). At this speed and growth rate, Hood Canal chum exit the Canal approximately 14 days after entering seawater, with approximately 90 percent of the annual population exiting by late April, on average (Wildlife and Tribes 2000).

Returning adults begin to arrive in Hood Canal in early August, and are thought to stage in front of their stream of origin for approximately ten to twelve days. During this time, they are generally found in subestuary habitats (e.g., tidal channels, mudflats, marshes, and eelgrass meadows) of the Hood Canal. Migration into freshwater spawning grounds generally occurs from late August to late October (Wildlife and Tribes 2000). The number of natural origin spawners in Hood Canal averaged 11,946 individuals from 2009 to 2013, and the number of hatchery origin spawners averaged 1,189 individuals over the same period (NMFS 2015). NMFS (2015) estimated this number of spawners would produce 1,920,944 outmigrants annually.

Based on the timing of returning adults and out-migrating juveniles, we do not expect Hood Canal summer chum of any life stage to be present in the Hood Canal portion of the Action Area from May through July, or November through January. From February through April we would expect juveniles to be present in the Action Area as they rear in Hood Canal, and then migrate towards the main basin of Puget Sound and into the North Pacific Ocean. Additionally, we expect adults to be present in the Hood Canal portion of the Action Area from August through October. If Navy EOD activities were to occur during either of these time periods, Hood Canal summer-chum could be affected by underwater explosions.

In order to minimize impacts to juvenile Hood Canal summer-run chum, the Navy will not conduct any of the planned larger size (E3; >0.5 to 2.5 lb) detonations at the Hood Canal site during February, March, or April (the juvenile outmigration period of juvenile chum (Wildlife and Tribes 2000)). During this time period, the EOD unit will only conduct training with SWAG charges, which are one ounce or less in net explosive weight (Andrea Balla-Holden, personal communication, October 21, 2015). Therefore, when estimating the number of juvenile Hood Canal summer-run chum that may be killed or injured, we assumed there would be no co-occurrence of juvenile chum with the use of E3 detonations.

In order to minimize impacts to adult Hood Canal summer-run chum, the Navy will avoid to the maximum extent practicable (unless necessitated by readiness requirements) the use of the larger size charges (>0.5 to 2.5 lb), and only conduct training with one ounce or less charges during August, September, and October, the adult migration period for Hood Canal summer-run Chum. Because the Navy could not rule out conducting explosions using these larger size charges during the adult migration period and in order to be conservative (i.e., assess the maximum effect to the species which could be considered reasonable), we assumed E3 detonations could occur during the adult migration period.

We estimated the number of Hood Canal summer-run Chum (juvenile and adult) that may be injured or killed from detonations in inland waters using similar methodology as described for estimating offshore take. To estimate chum juvenile density at the EOD site in Hood Canal, we first determined the average number of fish leaving freshwater from Hood Canal rivers per day by dividing the annual number of outmigrating juveniles by the number of days in the migration period (74 days). We then multiplied this number by the average time an individual fish is expected to spend in Hood Canal once it leaves freshwater (14 days, as described above). This results in the average number of juvenile Hood Canal summer-run chum we would expect in the Canal on a given day during the middle portion of the out-migration period (during first and last 14 days of the out-migration period, fewer individual salmon would be expected in the Canal). We then divide this number by the area of Hood Canal (358.4 km^2) to determine summer-run chum density during the outmigration period. To be conservative, we assumed all SWAG detonations will occur during the outmigration period as the Navy is unable to predict when these detonations will occur (as noted above, E3 charges will not be used during the outmigration period). For adults, we did not have information to estimate the amount of time each fish would be expected to reside in Hood Canal before entering freshwater to spawn. Therefore, we assumed the entire annual summer-run could occur in the Canal on a given day. We then divided this number by the area of Hood Canal to estimate adult density in the Canal on a given day during the run. Once we had juvenile and adult densities, we multiplied these numbers by the area of injury and mortality for each detonation, then multiplied this number by the number of detonations. (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 recognize that this methodology likely overestimates the number of Hood Canal summer-run chum that will die or be injured from Navy EOD activities. Most notably, our analysis and resulting take estimates assume all SWAG charge detonations occur when juvenile Hood Canal summer-run chum are in the Canal. Additionally, our analysis and resulting take estimates assume all detonations (E3 and SWAG charges) will occur when adult Hood Canal summer-run chum are in the Canal. While we did this to be conservative and to analyze the highest potential effect of the activity on ESA-listed species that could be considered reasonable, it is also possible that EOD activity may be spread more evenly throughout the year and occur during times when the species is not in Hood Canal or near the EOD range. If the Navy were to conduct some or all EOD activities during times of the year when Hood Canal summer-run chum are not present, the number of individuals that would die or be injured as a result of Navy EOD activities would be expected to be much lower or zero. Also, this methodology does not consider habitat selection within the Hood Canal by summer-run chum adults or juveniles. Adult fish in close proximity to the EOD range that may be affected would only be migrating through the area since there are no spawning rivers in immediate proximity to the range and would not be staging to enter freshwater spawning rivers at this location. Further, juveniles are expected at lower densities in the EOD range than in other areas of the Canal, where individuals of this life stage would be more likely to congregate (e.g., tidal channels, mudflats, marshes, and eelgrass meadows). These habitat types are not present in the Hood Canal EOD range. Additionally, it is worth noting that Navy monitoring of previous similar UNDETs at Hood Canal have never documented fish kills of the magnitude estimated to occur (Chip Johnson, personal communication, October 2015). However, Navy monitoring may not detect all fish that would die from an explosive event as some may not float to the surface of the water in order to be detected by visual observation and some may be captured by foraging birds prior to detection. Additionally, injured fish that may later die would not be expected to be detected as they also may not immediately float to the surface of the water to be detected by visual observation.

Table 122. Estimated annual number of ESA-listed salmonids that would die or be injured by explosive activities in Puget Sound.

Species	Location	Life Stage	Natural		Hatchery: adipose intact		Hatchery: adipose clip	
			Mortality	Injury	Mortality	Injury	Mortality	Injury
Chinook - T	Crescent Harbor	Juveniles	498.05	1278.74	0	0	199.22	511.50
		Adults*	1	0	0	0	0	0
	Hood Canal	Juveniles	23.05	59.18	90.86	233.29	0	0
		Adults	0.07	0.30	0.29	1.20	0	0
Steelhead - T	Crescent Harbor	Juveniles	1	0	0	0	0	0
		Adults	1	0	0	0	0	0
	Hood Canal	Juveniles	10.48	27.84	0	0	0	0
		Adults	0.37	1.19	0	0	0	0
Chum - T	Hood Canal	Juveniles	192.32	703.65	19.14	70.04	0	0
		Adults	4.45	14.58	0.44	1.45	0	0

***Estimated mortality or injury of one adult Chinook salmon at Crescent Harbor may be of natural or hatchery origin.**

6.11.1.2.4 Eulachon

Eulachon occur within Puget Sound, but are at very low abundance relative to coastal waters, and typically occupy very deep waters (Donnelly and Burr 1995) (74 FR 10857). Because of their relative low abundance and occurrence in very deep waters where EOD activities will not occur, we do not expect underwater explosions to adversely affect the southern DPS of eulachon.

6.11.1.2.5 Rockfish

The life histories, and therefore potential for exposure to underwater detonations, of the three rockfish species considered in this Opinion are similar, with some slight differences discussed below. All three have pelagic larval and juvenile stages, followed by a juvenile stage in shallow water, and a sub-adult/adult stage occurring in deeper waters. Larvae can make small movements on their own, but predominantly their movement is determined by ocean currents (Drake et al. 2010; Tagal et al. 2002). Larvae and pelagic juveniles are expected to occur throughout the water column in Puget Sound (Love et al. 2002; Weis 2004). Once canary rockfish and bocaccio reach 1 to 3.5 inches, they move into shallow nearshore waters, with rocky or cobble substrates, preferably with kelp (Love et al. 1991; Love et al. 2002). Alternatively, juvenile yellow rockfish do not occupy intertidal habitat, but are observed in deeper, offshore waters greater than 30 meters (Love et al. 1991; Studebaker et al. 2009; Yamanaka et al. 2006). Adults of each species are most commonly found between 40 and 250 meters (Love et al. 2002; Orr et al. 2000).

Based on rockfish life histories, as described above, larvae and pelagic juvenile ESA-listed rockfish would be expected to be adversely affected by underwater explosions in inland waters. Greene and Godersky (2012) conducted larval rockfish surveys throughout Puget Sound. In Hood Canal, the highest density of rockfish larvae sampled was 26.5 rockfish larvae/1000 m^3, observed near the Skokomish River in April. With the exception of May, when lower densities were observed, the authors did not catch larval rockfish in Hood Canal during any other months sampled (April through October). The highest density of rockfish larvae sampled in the Whidbey Basin (the region where Crescent Harbor is located) was 16.3 rockfish larvae/1000 m^3, observed in April. Rockfish larvae were not observed in any months in the Whidbey Basin except for April.

Additionally, juvenile canary rockfish and bocaccio may be affected when they reside in nearshore waters. We do not expect adult rockfish of any species or juvenile yelloweye rockfish to be affected by underwater explosions because they would occur in waters deeper than underwater explosions are expected to occur.

For juvenile (canary rockfish and bocaccio) and larval (all species) rockfish, we estimated the impact of EOD activity on these populations by assessing the percentage of the population that may be affected because we did not have information to estimate densities of these species at these life stages. We divided the area of injury and mortality by the area of Puget Sound (the species range). This estimates the proportion of larval and juvenile rockfish that may be impacted by Navy EOD activity at Hood Canal or Crescent Harbor, assuming equal distribution of the species. Using this methodology, we determined the proportion of Puget Sound larval (all species) and juvenile rockfish (Canary and bocaccio) that would be injured and killed is 0.003 and 0.002, respectively.

6.11.1.3 *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.

6.11.1.4 *Potential for TTS and Behavioral Reactions*

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. To our knowledge, no studies have examined the fitness implications when a fish, without notable hearing specialization, experiences TTS. Popper et al. (2014) 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. 2014). Additionally, hearing is not thought to play a role in salmon, steelhead, or eulachon 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 increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take. 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, 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 increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering, and therefore 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.

6.11.1.5 *Assessing the Effect of Take on ESA-listed Fish Populations*

It is important to note injury was defined as the distance from the detonation where no injury would occur; we consider this the point of onset injury. 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 conservative.

Many of the 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 10 year period (e.g., if 0.1 fish were estimated to be killed in one year, 1 fish was estimated to be killed in 10 years). If take estimates over this 10 year period were still <1 fish killed or injured, but ≥0.1, this estimate was rounded up to one. If over the 10 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 10 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.

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. If the status of the species and Navy activity levels remain the same, we would expect the estimated number of fish killed or injured by Navy explosive activities in a 10-year period to continue into the reasonably foreseeable future. However, we believe that because our estimates of the number of ESA-listed fish that will be injured or killed is dependent on fish abundance, attempting to predict the number of fish affected by Navy activities beyond ten years into the future is too speculative to allow for a scientifically meaningful assessment of the impacts on ESA-listed species beyond this timeframe. This is because fish abundance may change significantly over even a limited period of time (e.g., escapement of Upper Willamette

River spring Chinook, Table 25; escapement of Central Valley spring Chinook, Table 28) 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. That being said, there is no evidence that expressing take over a ten-year period risks 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. 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, assess any changes in ESA-listed fish species status, and to adjust our estimates of injury and mortality accordingly. 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 order to evaluate the potential for this level of take from Navy activities (training and testing) 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 the most current 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 10 year timeframe, assuming activity levels are maintained and status of the species does not change. Results for the offshore portion of the Action Area are in Table 123, Table 124, and Table 125. Results from inland waters are presented in Table 126.

Table 123. Percent of each ESA-listed fish ESU/DPS (hatchery produced with adipose fin-clip) that would be killed or injured in the offshore portion of the NWTT Action Area over a 10-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	Testing		Training		Percent of ESU/DPS (adipose fin-clip) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	0	1	0	0	0.040
	Juvenile		110	258	26	85	0.025
	Adult	Central valley spring run - T	3	11	1	3	0.009
	Juvenile		1,227	2,898	288	946	0.025
	Adult	California coastal - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Snake River fall - T	0	0	0	0	--
	Juvenile		326	770	77	252	0.005
	Adult	Snake River spring/summer - T	0	0	0	0	--
	Juvenile		464	1,097	109	358	0.005
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		4,378	10,342	1,030	3,375	0.005
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		660	1,558	155	509	0.005
	Adult	Upper Columbia River spring - E	0	0	0	0	--
	Juvenile		54	127	13	42	0.005
	Adult	Puget Sound - T	0	0	0	0	--
	Juvenile		10,555	24,933	2,479	8,137	0.013
Coho	Adult	Central Calif coast - E	1	1	1	1	0.061
	Juvenile		299	710	70	231	0.058
	Adult	S. Oregon/N. Calif coast - T	1	2	1	1	0.016
	Juvenile		73	173	17	56	0.016
	Adult	Oregon coast - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		5,354	12,740	1,248	4,131	0.027
Chum	Adult	Columbia River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
Sockeye	Adult	Snake River - E	0	1	0	0	0.002

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (adipose fin-clip) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
	Juvenile		2	5	1	2	0.001
Steelhead	Adult	South-Central California - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central Calif - T	1	1	1	1	0.003
	Juvenile		23	47	14	25	0.002
	Adult	California Central Valley - T	1	1	1	1	0.004
	Juvenile		57	116	33	61	0.002
	Adult	Northern Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Upper Columbia River - E	0	0	0	0	--
	Juvenile		30	64	14	26	0.002
	Adult	Snake River basin - T	0	0	0	0	--
	Juvenile		137	296	64	118	0.002
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		46	99	22	40	0.002
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Middle Columbia River - T	0	0	0	0	--
	Juvenile		16	34	8	14	0.002
	Adult	Puget Sound - T	0	0	0	0	--
	Juvenile		7	15	4	6	0.002

Table 124. Percent of each ESA-listed fish ESU/DPS (hatchery produced with intact adipose) that would be killed or injured in the offshore portion of the NWTT Action Area over a 10-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	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central valley spring run - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	California coastal - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Snake River fall - T	3	10	1	3	0.002
	Juvenile		401	946	94	309	0.005
	Adult	Snake River spring/summer - T	5	20	2	6	0.002
	Juvenile		124	292	29	95	0.005
	Adult	Lower Columbia River - T	3	9	1	3	0.002
	Juvenile		133	314	32	103	0.005
	Adult	Upper Willamette River - T	4	13	1	4	0.002
	Juvenile		5	11	2	4	0.005
	Adult	Upper Columbia River spring - E	1	2	1	1	0.003
	Juvenile		99	234	24	77	0.005
	Adult	Puget Sound - T	3	10	1	3	0.005
	Juvenile		1,728	4,080	406	1,332	0.013
Coho	Adult	Central Calif coast - E	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	S. Oregon/N. Calif coast - T	2	6	1	2	0.009
	Juvenile		209	497	49	161	0.016
	Adult	Oregon coast - T	1	2	1	1	0.014
	Juvenile		30	72	7	24	0.022
	Adult	Lower Columbia River - T	86	294	20	85	0.012
	Juvenile		186	443	44	144	0.027
Chum	Adult	Columbia River - T	0	0	0	0	<0.001
	Juvenile		9	18	2	5	0.001
Sockeye	Adult	Snake River - E	0	0	0	0	--

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
	Juvenile		0	0	0	0	--
Steelhead	Adult	South-Central California - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	California Central Valley - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Northern Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Upper Columbia River - E	1	2	1	1	0.002
	Juvenile		8	17	4	7	0.002
	Adult	Snake River basin - T	11	27	5	9	0.001
	Juvenile		43	94	20	38	0.002
	Adult	Lower Columbia River - T	2	5	1	2	0.001
	Juvenile		1	1	0	1	0.012
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Middle Columbia River - T	1	1	1	1	0.005
	Juvenile		19	41	9	17	0.002
	Adult	Puget Sound - T	1	1	0	1	0.010
	Juvenile		3	7	2	3	0.002

Table 125. Percent of each ESA-listed fish ESU/DPS (naturally produced) that would be killed or injured in the offshore portion of the NWTT Action Area over a 10-year period. --Indicates there are no naturally produced fish from this ESU/DPS that would be affected.

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	1	4	1	1	0.012
	Juvenile		92	216	22	71	0.025
	Adult	Central valley spring run - T	4	13	1	4	0.010
	Juvenile		875	2,065	206	674	0.025
	Adult	California coastal - T	7	24	2	7	0.019
	Juvenile		1,406	3,321	330	1,084	0.047
	Adult	Snake River fall - T	2	5	1	2	0.002
	Juvenile		61	143	15	47	0.005
	Adult	Snake River spring/summer - T	2	7	1	2	0.002
	Juvenile		155	364	37	119	0.005
	Adult	Lower Columbia River - T	2	6	1	2	0.003
	Juvenile		1,645	3,884	386	1,268	0.005
	Adult	Upper Willamette River - T	1	4	1	1	0.002
	Juvenile		200	471	47	154	0.005
	Adult	Upper Columbia River spring - E	1	1	1	1	0.004
	Juvenile		61	143	15	47	0.005
	Adult	Puget Sound - T	5	16	1	5	0.005
	Juvenile		674	1,592	159	520	0.013
Coho	Adult	Central Calif coast - E	2	4	1	2	0.035
	Juvenile		120	285	28	93	0.058
	Adult	S. Oregon/N. Calif coast - T	3	9	1	3	0.008
	Juvenile		373	887	87	288	0.016
	Adult	Oregon coast - T	64	219	15	63	0.009
	Juvenile		6,742	16,042	1,571	5,202	0.022
	Adult	Lower Columbia River - T	5	16	2	5	0.013
	Juvenile		521	1,238	122	402	0.027
Chum	Adult	Columbia River - T	1	1	1	1	0.001
	Juvenile		62	133	15	38	0.001
Sockeye	Adult	Snake River - E	0	0	0	0	--

Species	**Life stage**	**ESU/DPS**	**Testing**		**Training**		**Percent of ESU/DPS (intact adipose) Killed or Injured**
			Mortality (10 years)	**Injury (10 years)**	**Mortality (10 years)**	**Injury (10 years)**	
	Juvenile		1	1	1	1	0.003
Steelhead	Adult	South-Central California - T	0	1	0	0	0.005
	Juvenile		3	6	2	3	0.002
	Adult	Central Calif - T	1	1	0	1	0.007
	Juvenile		6	12	4	7	0.002
	Adult	California Central Valley - T	1	1	0	1	0.007
	Juvenile		6	12	4	6	0.002
	Adult	Northern Calif - T	1	1	1	1	0.003
	Juvenile		0	47	10	19	0.002
	Adult	Upper Columbia River - E	1	1	1	1	0.005
	Juvenile		13	28	6	11	0.002
	Adult	Snake River basin - T	4	9	2	3	0.001
	Juvenile		62	135	29	54	0.002
	Adult	Lower Columbia River - T	1	3	1	1	0.002
	Juvenile		20	43	10	18	0.002
	Adult	Upper Willamette River - T	1	2	1	1	0.003
	Juvenile		10	21	5	9	0.002
	Adult	Middle Columbia River - T	2	5	1	2	0.001
	Juvenile		24	52	12	21	0.002
	Adult	Puget Sound - T	1	3	1	1	0.001
	Juvenile		74	161	35	64	0.002

Table 126. Percent of each ESA-listed Puget Sound salmonid ESU/DPS that would be killed or injured during Navy training activities in the inland portion of the NWTT Action Area over a 10-year period. --Indicates there are no fish from this ESU/DPS that are either naturally producing, from a hatchery with an intact adipose fin, or from a hatchery with an adipose clip that would be affected.

Species	Life Stage	Natural (10 year)			Hatchery: adipose intact (10 year)			Hatchery: adipose clip (10 year)		
		Mortality	Injury	Percent affected	Mortality	Injury	Percent affected	Mortality	Injury	Percent affected
Chinook - T	Juveniles	5,212	13,380	0.080	909	2,333	0.005	1,993	5,115	0.002
	Adults	11	3	0.008	3	12	0.014	0	0	--
Steelhead - T	Juveniles	115	279	0.002	0	0	--	0	0	--
	Adults	14	12	0.019	0	0	--	0	0	--
Chum - T	Juveniles	1,924	7,037	0.029	192	701	0.032	0	0	--
	Adults	45	146	0.109	5	15	0.058	0	0	--

We determined that no greater than 0.109 percent (adult Hood Canal summer-run chum) of any salmonid ESU or DPS would be injured or killed from Navy activities[21]. Most salmon ESUs and steelhead DPSs would be affected at much lower levels. It is also worth noting that the injury and mortality estimates for Hood Canal summer-run Chum and Puget Sound Chinook are likely overestimates because the Navy will avoid, to the maximum extent practicable, the use of larger charges at the Hood Canal site when adult salmon are migrating to spawn[22]. 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. It is also worth noting that some of the relatively higher take numbers (in terms of individual fish) would be of hatchery stock fish (e.g., juvenile Puget Sound Chinook hatchery fish with adipose clip) which is considered to be of less conservation value for the ESU compared to the natural origin population (NMFS 2015).

As discussed above, we determined the proportion of Puget Sound larval (all species) and juvenile rockfish (Canary and bocaccio) populations that would be injured and killed by Navy training activities is 0.003 and 0.002, respectively. Additionally, we determined that <0.001 percent of the southern DPS of eulachon may die or be injured from Navy detonations during

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[21] We added the effects of offshore and inland water detonations to Puget Sound ESUs/DPSs. The percent of the population injured or killed did not exceed 0.093 percent for Puget Sound Chinook or steelhead.
[22] For more detail on this methodology and caveats, please see section 6.11.1.2.3.

training (0 percent) and testing (<0.001 percent). These levels of injury and mortality are miniscule from a population-level perspective and, assuming no changes in the status of the species, would not be expected to reduce appreciably the likelihood of survival and recovery of these populations over a ten-year period or continuing into the reasonably foreseeable future.

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 also 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 Action Area where they may be affected by Navy activities. Additionally, 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 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. Additionally, because of the variability in life history exhibited within the ESA-listed entities considered in this Opinion and the infrequent nature of Navy training and testing (spread out across space and time), we would not expect the proposed action to affect species structure and diversity.

In summary, we conclude that the level of incidental take of ESA-listed fish species during Navy training and testing 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. We also conclude that the diversity of ESA-listed fish populations will be not be affected by this limited amount of take that should be distributed across populations across their ranges and through time during outmigration and emigration periods. 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 population structure or diversity 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.

6.12 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 6.11) 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 and testing 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 NWTT Action Area will continue to introduce a suite of stressors into the marine and coastal ecosystems. 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, 6.4.7, and 6.5.7. 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 NWTT 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 NWTT Action Area and associated impacts will continue into the reasonably foreseeable future at the levels set forth in the Final EIS/OEIS and MMPA rule. 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 and hearing loss. We expect ESA-listed species will not receive repeated exposures at a rate at which recovery between exposures would not occur because of the intermittent nature and duration of Navy acoustic sources. Navy activities in the NWTT

Action Area involving active sonar or underwater detonations are infrequent, short-term, and generally unit level. Unit level events occur over a small spatial scale (one to a few 10s of square miles) and with few participants (usually one or two). Single-unit unit level training would typically involve a few hours of sonar use, with a typical nominal ping of every 50 seconds (duty cycle). Even though an animal's exposure to active sonar may be more than one time, the intermittent nature of the sonar signal, its low duty cycle, and the fact that both the vessel and animal are moving provide a very small chance that exposure to active sonar for individual animals and stocks would be repeated over extended periods of time. Consequently, the Navy's NWTT 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 exercises), 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 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 decades of training in the NWTT Action Area using many of the same systems. Most of the training activities the Navy conducts in the NWTT Action Area are similar, if not identical, to activities that have been occurring in the same locations for decades.

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 and testing 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 only present in the offshore portion of the NWTT Action Area. Blue whales may be exposed to acoustic stressors associated with training and testing activities throughout the year. Blue whales found in offshore portion of the NWTT 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 active acoustic sources associated with training activities in the Offshore Area that may result in two TTS and three behavioral reactions annually. Blue whales may also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic analysis predicts that blue whales of the Eastern North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in six TTS exposures annually.

Acoustic modeling predicts that ESA-listed mysticetes including blue whales may be exposed to impulses from explosive sources associated with training and testing activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds.

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 and testing 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. Additionally, in instances of TTS, individuals would likely fully recover within 24 hours of exposure and resume normal behavioral activities. There is no indication, 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.

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. 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.

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 exercises), 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.

Most of the blue whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including TTS, most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously, including in this Opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training and testing 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 and testing activities, including the use of MFA sonar, have been ongoing in this area for decades. Therefore, any potential temporary behavioral impacts on blue whales from the use of MFA during Navy training and testing activities do not appear to have inhibited population growth of the Eastern North Pacific blue whale population in that area. We would expect similar effects to exist in the NWTT Action Area.

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 NWTT 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 NWTT activities is not expected to impact the fitness of any individuals of this species. No mortality of blue whales is expected to occur from NWTT activities.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the Navy will conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA Rule from November 2015 through November 2020, 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 and testing 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. NWTT 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, 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. 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 Fin Whale

Fin whales could be present in all portions of the NWTT Action Area and may be exposed to acoustic sources associated with training and testing activities throughout the year. Fin whales found in the offshore portions of the Action Area and the Inland Waters of Puget Sound area are recognized as part of the California, Oregon, Washington stock. The acoustic analysis predicts that fin whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in 11 TTS and 14 behavioral reactions annually. Fin whales may also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic analysis predicts that fin whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in 30 TTS, and 4 behavioral reactions. Fin whales found in Southeast Alaska (Western Behm Canal) are recognized as part of the Northeast Pacific stock. The acoustic analysis predicts fin whales of the Northeast Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities that may result in two behavioral reactions annually.

Acoustic modeling predicts that ESA-listed mysticetes including fin whales may be exposed to impulses from explosive sources associated with training and testing activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds.

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 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 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.

Most of the fin whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, most often 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 and testing 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 fin 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 fisheries, climate change, direct harvest, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. As discussed previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. 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 and testing activities the Navy conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA Rule from November 2015 through November 2020, 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 and testing 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. NWTT 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.

7.3 Humpback Whale

Humpback whales could be in all portions of the NWTT Action Area and may be exposed to sonar and other active acoustic sources associated with training activities throughout the year. Humpback whales found in the offshore portions of the Action Area and the Inland Waters of Puget Sound area are recognized as part of the California, Oregon, Washington stock. The humpback whales in the Action Area potentially belong to one of three proposed DPSs: the non-ESA-listed Hawaii DPS, the non-ESA-listed Mexico DPS, or the threatened Central America DPS. All three of these DPSs may feed seasonally in the Action Area.

The acoustic analysis predicts that humpback whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in five TTS and seven behavioral reactions annually. The acoustic analysis also predicts that humpback whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in 39 TTS and 5 behavioral reactions.

Humpback whales found in Southeast Alaska (Western Behm Canal) are recognized as part of the Central North Pacific stock and the acoustic analysis predicts that the Central North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities that may result in 1 behavioral reaction annually.

Acoustic modeling predicts that ESA-listed mysticetes including humpback whales may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds. The NAEMO and post-model analysis also predicts no exposures of humpback whales from impulse sources associated with testing activities.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of humpback 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 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.

Most of the humpback whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously in this Opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training and testing 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 general increase in the number of humpback whales range-wide suggests that the stress regime these whales are exposed to including activities in the NWTT Action Area have not prevented these whales from increasing their numbers. Humpback whales have been exposed to Navy training exercises in the NWTT Action Area, including vessel traffic, aircraft traffic, active sonar, and underwater detonations, for more than a generation. Although the Navy proposes to increase the frequency of some of these activities, we do not believe those increases are likely to affect the rate at which humpback whale counts are increasing.

The 1991 humpback whale recovery plan does not outline specific downlisting and delisting criteria. The recovery plan does list several threats known or suspected of impacting humpback whale recovery including subsistence 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 NWTT activities. As described previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the Navy will conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA rule from November 2015 through November 2020, 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 humpback 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 and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of humpback 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. NWTT 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 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. Due to a lack of fitness consequences to individuals and the populations they represent, we also do not anticipate any reductions in survival rate or trajectory of recovery of the threatened Central America DPS as currently proposed.

7.4 Sei Whale

Sei whales may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds. Sei whales may also be exposed to sonar and other active acoustic sources associated with testing activities throughout the year. The acoustic analysis predicts that sei whales of the Eastern North Pacific stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in two TTS exposures annually.

Acoustic modeling predicts that ESA-listed mysticetes including sei whales may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds. The NAEMO and post-model analysis also predicts no exposures of sei whales from impulse sources associated with testing activities.

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.

Most of the sei whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, most often 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 and testing 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 fisheries, climate change, direct harvest, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. As described previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. Downlisting criteria for fin 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 sei whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects,* stressors resulting from training and testing activities the Navy will conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA regulations from November 2015 through November 2020, 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 and testing 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. NWTT 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.

7.5 Sperm Whale

Sperm whales are likely to be found only in the offshore portion of the Action Area. Sperm whales found in the Offshore Area are recognized as part of the California, Oregon, Washington stock. The acoustic analysis predicts that sperm whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in 81 behavioral reactions annually. The acoustic analysis predicts that sperm whales of the California, Oregon, Washington stock may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in 67 TTS and 11 behavioral reactions. There are no model predicted exposures for the North Pacific stock of sperm whales.

Acoustic modeling predicts that including sperm whales may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds. The NAEMO and post-model analysis also predicts no exposures of sperm whales from impulse sources associated with testing activities.

Most of the sperm whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, most often 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 and testing 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 fisheries, climate change, direct harvest, oil spills, anthropogenic noise, and ship collision. Of these, anthropogenic noise and ship collision are relevant to NWTT activities. As discussed previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. 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 and testing activities the Navy plans to conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA regulations from November 2015 through November 2020, 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 and testing 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. NWTT 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.

7.6 Southern Resident Killer Whale

Southern Resident killer whales may be exposed to sonar and other active acoustic sources associated with training activities while they reside in the NWTT Action Area. The majority of training activities occur in the offshore portion of the NWTT Action Area. The predicted effects would occur in the Inland Waters area of Puget Sound, where Southern Resident killer whales could be exposed to sonar and other active acoustic sources that may result in two behavioral reactions biennially from civilian port defense training. In years without civilian port defense training no Southern Resident killer whale takes are anticipated. We do not expect any instances of TTS, PTS, injury, or mortality from training or testing activities. Southern Resident killer whales may also be exposed to sonar and other active acoustic sources associated with testing activities while they reside in the NWTT Action Area. The majority of testing events occur in areas such as Hood Canal, where Southern Resident killer whales are not believed to be present; the remaining testing activities occur offshore, where they are only present briefly during their annual migration period. Acoustic modeling predicts that Southern Resident killer whales may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area; however, there are no predicted effects from these testing activities, because exposures would not exceed the current effects thresholds.

Acoustic modeling predicts Southern Resident killer whales may be exposed to impulses from explosive sources associated with training activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds. The NAEMO and post-model analysis also predicts no exposures of Southern Resident killer whales from impulse sources associated with testing activities.

The small numbers of Southern Resident killer whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the NWTT Action Area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously in this Opinion, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training and testing 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 2008 southern resident killer whale recovery plan sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats such as those associated with prey availability, water contamination, ship collision, and anthropogenic noise. Of these, anthropogenic noise is relevant to NWTT activities. As discussed previously, anthropogenic noise associated with NWTT activities will not impact the fitness of any individuals of this species. Downlisting criteria for southern resident killer whales include an average growth of 2.3 percent per year for 14 years and progress toward addressing threats represent sustained growth to indicate that the population could be downlisted to threatened. To qualify for delisting, the population of southern resident killer whales must include a sustained average growth of 2.3 percent per year for 28 years and population parameters consistent with a healthy growing population.

Based on the evidence available, including the environmental baseline and cumulative effects, we conclude that impulsive and non-impulsive stressors resulting from training and testing activities the Navy plans to conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA regulations from November 2015 through November 2020, 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 Southern Resident killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Southern Resident killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. NWTT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual Southern Resident killer 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 Guadalupe Fur Seal

Guadalupe fur seals are present within the coastal margins of the offshore portion of the Action Area during the warm season (summer and early autumn). Guadalupe fur seals are considered “seasonal” migrants since they return to rookeries in Mexican waters in the cold season. The acoustic analysis predicts that Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in seven behavioral reactions annually. Additionally, Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with testing activities. The acoustic analysis predicts that Guadalupe fur seals may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in three behavioral reactions annually.

Acoustic modeling predicts that Guadalupe fur seals may be exposed to impulses from explosive sources associated with training and testing activities in the Offshore Area; however, there are no predicted effects from these activities, because exposures would not exceed the current effects thresholds.

We do not anticipate any instances of PTS, injury or mortality from any stressors associated with training or testing activities. Pinnipeds in the water are tolerant of anthropogenic noise and activity. The available scientific information does not provide evidence that behavioral responses of pinnipeds to acoustic stressors from Navy training and testing 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 Guadalupe fur seal does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of individuals would occur and that effects from acoustic stressors would be temporary and not impact the fitness of individuals or the population. In the absence of fitness consequences on individuals or the population to which those individuals belong, we do not expect an appreciable reduction in the ability of this entity to recover.

Therefore, based on the evidence available, including the environmental baseline and cumulative effects, we conclude that impulsive and non-impulsive stressors resulting from training and testing activities the Navy plans to conduct in the NWTT Action Area on an annual basis, cumulatively over the five year period of the MMPA regulations from November 2015 through November 2020, 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 Guadalupe fur seals in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Guadalupe fur seals in the wild by reducing the reproduction, numbers, or distribution of that species. NWTT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual seals 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.

7.8 Leatherback Sea Turtles

The Navy's acoustic analysis predicts that leatherback sea turtles may be exposed to sonar and other active acoustic sources associated with training activities in the Offshore Area that may result in one TTS (from non-impulsive sources) and 14 behavioral (from impulsive sources) exposures annually. The acoustic analysis also predicts that leatherback sea turtles may be exposed to sonar and other active acoustic sources associated with testing activities in the Offshore Area that may result in five TTS (from non-impulsive sources) and 14 behavioral (from impulsive sources) exposures annually.

Up to 142 training activities using explosives at or beneath the water surface would be conducted annually and potentially expose leatherback sea turtles to underwater impulse sound. The largest source class used during training would be E12 (> 650 to 1,000 lb. NEW), which would be used 10 times annually 50 nm or further from shore in the NWTT Action Area. Leatherback sea turtles may be exposed to impulse sound from explosive sources associated with training activities in the Offshore Area; however, there are no predicted mortality, onset slight lung injury, gastrointestinal tract injury, PTS, or TTS effects from these activities, because exposures would not exceed the current effects thresholds.

Testing activities using explosives at or beneath the water surface would potentially expose leatherback sea turtles to underwater impulse sound. Explosives would be used only in the Offshore Area of the Action Area and for a total of 148 events per year. Explosive ordnance used during testing activities include explosive torpedoes (6), sound underwater signal sonobuoys (72), and Improved Extended Echo Ranging (IEER) sonobuoys (70). Results from modeling indicate no leatherback sea turtles are predicted to be exposed to impulse levels associated with the onset of mortality, onset slight lung injury, gastrointestinal tract injury, PTS, or TTS over any testing year for explosives use in open ocean.

Some sea turtles beyond the ranges of the above impacts may behaviorally react if they hear a detonation. Activities consisting of single detonations, such as bombing and missile exercise, are expected to only elicit short-term startle reactions. If a sea turtle hears multiple detonations in a short period, such as during gunnery, firing, or sonobuoy exercises, it may react by avoiding the area. Any significant behavioral reactions could lead to a sea turtle expending energy and missing opportunities to secure resources. However, because most activities would consist of a limited number of detonations and exposures would not occur over long durations, there would be an opportunity to recover from an incurred energetic cost.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication.

Further, although the information on the hearing capabilities of sea turtles is limited, the information available suggests that the auditory capabilities of sea turtles are centered in the low-frequency range (<2 kHz) (Bartol et al. 1999b; Dow Piniak et al. 2012; Lenhardt et al. 1983; Lenhardt et al. 1994a; Martin et al. 2012; O'Hara and Wilcox 1990; Ridgway et al. 1969), with greatest sensitivity below 1 kHz.

As described previously in this Opinion, potential behavioral responses of sea turtles to anthropogenic sound could include startle reactions, disruption of feeding, disruption of migration, changes in respiration, alteration of swim speed, alteration of swim direction, and area avoidance. Any disruptions are expected to be temporary in nature, with the animal resuming normal behaviors shortly after the exposure. To result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for lost feeding opportunities 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 foraging habitat would still be available in the environment following the cessation of acoustic exposure. Similarly, we expect temporary disruptions of migration and swim speed or direction to be inconsequential because they can resume these behaviors almost immediately following the cessation of the sound exposure. Further, these sorts of behavioral disruptions may be similar to natural disruptions such those resulting from predator avoidance, or fluctuations in oceanographic conditions. Therefore, behavioral responses of sea turtles to acoustic stressors are unlikely to lead to fitness consequences and long-term implications for the population.

We assume that acoustic stressors from Navy training and testing activities conducted during the five-year MMPA rule and into the reasonably foreseeable future are likely to cause TTS in ESA-listed sea turtles in the NWTT Action Area at the levels described above. However, as described previously in this Opinion, there is no evidence that TTS results in energetic effects to individual sea turtles or would be likely to significantly reduce the viability of the population these individuals represent. Given that sea turtles do not rely on acoustic cues for most important life functions, it is anticipated that TTS will not result in fitness consequences to individuals or the populations to which they belong.

Most of the sea turtles that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training and testing in the Action

Area. These periodic or episodic exposure and response scenarios most often allow sufficient time for the affected individuals to return to baseline conditions and resumption of normal behavioral activities. As described previously in this Opinion, available scientific information does not provide evidence that exposure of sea turtles to acoustic stressors leading to sub-lethal effects from Navy training and testing activities will reduce the fitness of any individuals of this species.

The 1998 recovery plan for the U.S. Pacific populations of leatherback sea turtles does not identify any major threats occurring in the Action Area, but sets criteria for the delisting the species. Delisting requires identifying regional stocks to source beaches, stability in the number of nesting females over 25 years, maintenance of at least 5,000 females in each stock over 6 years, maintenance of healthy foraging habitat, increases of foraging populations, completion of all priority one tasks, and the finalization of management plans. Leatherbacks affected by the proposed testing and training activities are not expected to experience fitness consequences because this species does not rely heavily on auditory cues from their environment for breeding, feeding, or sheltering. Because of a lack of fitness consequences and that acoustic stressors are not identified as a threat to leatherback recovery, we do not expect sub-lethal leatherback sea turtle take to impede recovery of this species.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects,* stressors resulting from training and testing activities the Navy will conduct in the NWTT Action Area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of leatherback sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of leatherback sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of sub-populations in the North Pacific or to these species as listed pursuant to the ESA.

7.9 Threatened and Endangered Fishes

As described in Section 6, the only impacts to ESA-listed fish species that are expected would be injury and/or death of some individuals due to explosive activities. As described in Section 6.11.1.5, the highest percentage of any ESU or DPS that is expected to be injured or killed from these activities in the offshore portion of the Action Area is 0.061 percent. The highest percentage of any ESU or DPS that is expected to be injured or killed from these activities in the inland portion of the Action Area is 0.109 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, steelhead, rockfish, or southern eulachon. It is also worth noting that, as described in Section 6.11, 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 NWTT Action Area would not appreciably reduce the likelihood of ESA-listed Pacific salmon, steelhead, rockfish, or southern eulachon 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. 2014). Additionally, hearing is not thought to play a role in salmon, steelhead, or eulachon 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 increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including 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 increase the likelihood of injury by annoying a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects,* stressors resulting from training and testing activities the Navy will conduct in the NWTT Action Area on an annual basis, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Pacific salmon, steelhead, eulachon, or rockfish in the wild by reducing the reproduction, numbers, or distribution of those species, ESUs, or DPSs. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Pacific salmon, steelhead, eulachon, or rockfish in the wild by reducing the reproduction, numbers, or distribution of those species, 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, fin whales, humpback whales, sei whales, Southern resident killer whales, sperm whales, Guadalupe fur seals, leatherback sea turtles, Puget Sound/Georgia Basin bocaccio, Puget Sound/Georgia Basin yelloweye and canary rockfish, southern DPS of Pacific eulachon, Chinook salmon (Sacramento River winter-run, Central valley spring-run, California coastal, Snake River fall, Snake River spring/summer, Lower Columbia River, Upper Willamette River, Upper Columbia River, and Puget Sound ESUs), chum salmon (Hood Canal summer-run and Columbia River ESUs), coho salmon (Central California coast, Southern Oregon Northern California coast, Lower Columbia River, and Oregon Coast ESUs), Snake River sockeye salmon ESU, and steelhead (South Central California Coast, Central California Coast, California Central Valley, Northern California, Upper Columbia River, Snake River Basin, Lower Columbia River, Upper Willamette River, Middle Columbia River, and Puget Sound DPSs). We also assessed the status of the proposed threatened Central America DPS of humpback whale as part of our conference report. Additionally, we assessed the *Environmental Baseline* for the NWTT Action Area including ongoing Navy training and testing in the NWTT Action Area along with the potential effects of Navy proposed NWTT from November 2015 through November 2020 (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 and testing activities and proposed letter of authorization for the five-year period.

We conclude that Navy training and testing activities in the NWTT Action Area 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. 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 and testing 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 is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Under the Marine Mammal Protection Act, 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 this consultation, we interpreted "harass" for marine mammals using the MMPA definition.[23] For sea turtles and fish species, we applied "harass" to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behavior to a point where such behaviors are abandoned or significantly altered.

Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 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.

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[23] As discussed in the Opinion, the Navy uses a 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. In the Opinion, we therefore considered the available scientific evidence to determine the likely nature of the modeled marine mammals behavioral responses and the potential fitness consequences for affected individuals.

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 while the extent of take or "the extent of land or marine area that may be affected by an action" if we cannot assign numerical limits for animals that could be incidentally taken during the course of 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, fin whales, humpback whales, sei whales, sperm whales, Southern Resident killer whales, leatherback sea turtles, and listed fish species have small probabilities of being exposed to the active sonar, sound fields associated with underwater detonations, or noise and other environmental cues associated with the movement of surface vessels. In some instances, we concluded that this exposure was 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 identified in Table 127 (Training Activities) and Table 128 (Testing Activities) 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 minor disruptions of the normal behavioral patterns of the marine mammals and sea turtles that have been exposed from which individuals would recover from quickly. As discussed throughout this Opinion, these disruptions are expected to be temporary and the animals are expected to fully recover. 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 whales or sea turtles are likely to die or be wounded or injured as a result of their exposure to the Navy NWTT activities. Therefore, for the purposes of this Biological Opinion and Incidental Take Statement, we assume that the Navy NWTT training and testing activities are likely to result in incidental "take" shown in Table 127 and Table 128, respectively.

Table 127. The number of threatened or endangered marine mammals and sea turtles that are likely to be "taken" as a result of their exposure to Navy Training activities conducted on in the NWTT Action Area on an Annual Basis.

Species	Annual Estimated Take – Training Activities			
	Non-Impulsive Sound (Sonar)		Impulsive Sound (Detonations)	
	Behavioral Harassment (including TTS) The subset of TTS is in parentheses	Harm (PTS, Injury, Mortality)	Harass (Behavioral & TTS)	Harm (PTS, Injury, Mortality)
Blue Whale	5 (2)	0	0	0
Fin Whale	25 (11)	0	0	0
Humpback Whale*	12 (5)	0	0	0
Sei Whale	0	0	0	0
Killer Whale – Southern Resident DPS	2** (0)	0	0	0
Sperm Whale	81 (0)	0	0	0
Guadalupe Fur Seal	7 (0)			
Leatherback Sea Turtle	1 (1)	0	14	0

***The humpback whale take estimates apply to the rangewide listing of the species. In the event the humpback whale DPSes are finalized as currently proposed, these take estimates will apply to the threatened Central America DPS. Although affected humpback whales could belong to non-listed DPSes, these whales would not be identifiable as not belonging to the threatened Central America DPS without genetic analysis.**

****Behavioral harassment of Southern Resident killer whales results from biennial civilian port defense training which will not occur more often than every other year starting with the first year under this BiOp and continuing into to reasonably foreseeable future.**

Table 128. The number of threatened or endangered marine mammals and sea turtles that are likely to be "taken" as a result of their exposure to Navy Testing Activities conducted on in the NWTT Action Area on an Annual Basis.

Species	Annual Estimated Take – Testing Activities			
	Non-Impulsive Sound (Sonar)		Impulsive Sound (Detonations)	
	Behavioral Harassment (including TTS) The subset of TTS is in parentheses	Harm (PTS, Injury, Mortality)	Harass (Behavioral & TTS)	Harm (PTS, Injury, Mortality)
Blue Whale	6 (6)	0	0	0
Fin Whale	36 (30)	0	0	0
Humpback Whale*	45 (39)	0	0	0
Sei Whale	2 (2)	0	0	0
Killer Whale – Southern Resident DPS	0	0	0	0
Sperm Whale	78 (67)	0	0	0
Guadalupe Fur Seal	3	0	0	0
Leatherback Sea Turtle	5 (5)	0	14	0

***The humpback whale take estimates apply to the rangewide listing of the species. In the event the humpback whale DPSes are finalized as currently proposed, these take estimates will apply to the threatened Central America DPS. Although affected humpback whales could belong to non-listed DPSes, these whales would not be identifiable as not belonging to the threatened Central America DPS without genetic analysis.**

In Section 6 of this Opinion, we estimated the number of ESA-listed fish that would be injured or die from Navy NWTT training and testing activities. As discussed in Section 6.11.1.5, we estimated the number of ESA-listed salmonids, rockfish, and eulachon that would be injured or die from Navy NWTT training and testing activities over a ten-year period. Table 129, Table 130, Table 131, and Table 132 show the number of ESA-listed fish that would be killed or injured by Navy NWTT training and testing activities over a 10 year period in the offshore portion of the Action Area. "Take" of these species will be exceeded if the number of detonations, the location of the detonations, or the Net Explosive Weight of the detonations are greater than we expected in our analyses or if the monitoring program associated with the Navy's activities detects a greater number of any fish species than are identified in the following tables.

Table 129. 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 and Testing Activities conducted in the offshore portion of the NWTT Action Area over a 10-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (adipose fin-clip) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	0	1	0	0	0.040
	Juvenile		110	258	26	85	0.025
	Adult	Central valley spring run - T	3	11	1	3	0.009
	Juvenile		1,227	2,898	288	946	0.025
	Adult	California coastal - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Snake River fall - T	0	0	0	0	--
	Juvenile		326	770	77	252	0.005
	Adult	Snake River spring/summer - T	0	0	0	0	--
	Juvenile		464	1,097	109	358	0.005
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		4,378	10,342	1,030	3,375	0.005
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		660	1,558	155	509	0.005
	Adult	Upper Columbia River spring - E	0	0	0	0	--
	Juvenile		54	127	13	42	0.005
	Adult	Puget Sound - T	0	0	0	0	--
	Juvenile		10,555	24,933	2,479	8,137	0.013
Coho	Adult	Central Calif coast - E	1	1	1	1	0.061
	Juvenile		299	710	70	231	0.058
	Adult	S. Oregon/N. Calif coast - T	1	2	1	1	0.016
	Juvenile		73	173	17	56	0.016
	Adult	Oregon coast - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		5,354	12,740	1,248	4,131	0.027
Chum	Adult	Columbia River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
Sockeye	Adult	Snake River - E	0	1	0	0	0.002

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (adipose fin-clip) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
	Juvenile		2	5	1	2	0.001
Steelhead	Adult	South-Central California - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central Calif - T	1	1	1	1	0.003
	Juvenile		23	47	14	25	0.002
	Adult	California Central Valley - T	1	1	1	1	0.004
	Juvenile		57	116	33	61	0.002
	Adult	Northern Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Upper Columbia River - E	0	0	0	0	--
	Juvenile		30	64	14	26	0.002
	Adult	Snake River basin - T	0	0	0	0	--
	Juvenile		137	296	64	118	0.002
	Adult	Lower Columbia River - T	0	0	0	0	--
	Juvenile		46	99	22	40	0.002
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Middle Columbia River - T	0	0	0	0	--
	Juvenile		16	34	8	14	0.002
	Adult	Puget Sound - T	0	0	0	0	--
	Juvenile		7	15	4	6	0.002

Table 130. 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 and Testing Activities conducted in the offshore portion of the NWTT Action Area over a 10-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central valley spring run - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	California coastal - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Snake River fall - T	3	10	1	3	0.002
	Juvenile		401	946	94	309	0.005
	Adult	Snake River spring/summer - T	5	20	2	6	0.002
	Juvenile		124	292	29	95	0.005
	Adult	Lower Columbia River - T	3	9	1	3	0.002
	Juvenile		133	314	32	103	0.005
	Adult	Upper Willamette River - T	4	13	1	4	0.002
	Juvenile		5	11	2	4	0.005
	Adult	Upper Columbia River spring - E	1	2	1	1	0.003
	Juvenile		99	234	24	77	0.005
	Adult	Puget Sound - T	3	10	1	3	0.005
	Juvenile		1,728	4,080	406	1,332	0.013
Coho	Adult	Central Calif coast - E	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	S. Oregon/N. Calif coast - T	2	6	1	2	0.009
	Juvenile		209	497	49	161	0.016
	Adult	Oregon coast - T	1	2	1	1	0.014
	Juvenile		30	72	7	24	0.022
	Adult	Lower Columbia River - T	86	294	20	85	0.012
	Juvenile		186	443	44	144	0.027
Chum	Adult	Columbia River - T	0	0	0	0	<0.001
	Juvenile		9	18	2	5	0.001
Sockeye	Adult	Snake River - E	0	0	0	0	--

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
	Juvenile		0	0	0	0	--
Steelhead	Adult	South-Central California - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Central Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	California Central Valley - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Northern Calif - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Upper Columbia River - E	1	2	1	1	0.002
	Juvenile		8	17	4	7	0.002
	Adult	Snake River basin - T	11	27	5	9	0.001
	Juvenile		43	94	20	38	0.002
	Adult	Lower Columbia River - T	2	5	1	2	0.001
	Juvenile		1	1	0	1	0.012
	Adult	Upper Willamette River - T	0	0	0	0	--
	Juvenile		0	0	0	0	--
	Adult	Middle Columbia River - T	1	1	1	1	0.005
	Juvenile		19	41	9	17	0.002
	Adult	Puget Sound - T	1	1	0	1	0.010
	Juvenile		3	7	2	3	0.002

Table 131. 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 and Testing Activities conducted in the offshore portion of the NWTT Action Area over a 10-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
Chinook	Adult	Sac River winter run - E	1	4	1	1	0.012
	Juvenile		92	216	22	71	0.025
	Adult	Central valley spring run - T	4	13	1	4	0.010
	Juvenile		875	2,065	206	674	0.025
	Adult	California coastal - T	7	24	2	7	0.019
	Juvenile		1,406	3,321	330	1,084	0.047
	Adult	Snake River fall - T	2	5	1	2	0.002
	Juvenile		61	143	15	47	0.005
	Adult	Snake River spring/summer - T	2	7	1	2	0.002
	Juvenile		155	364	37	119	0.005
	Adult	Lower Columbia River - T	2	6	1	2	0.003
	Juvenile		1,645	3,884	386	1,268	0.005
	Adult	Upper Willamette River - T	1	4	1	1	0.002
	Juvenile		200	471	47	154	0.005
	Adult	Upper Columbia River spring - E	1	1	1	1	0.004
	Juvenile		61	143	15	47	0.005
	Adult	Puget Sound - T	5	16	1	5	0.005
	Juvenile		674	1,592	159	520	0.013
Coho	Adult	Central Calif coast - E	2	4	1	2	0.035
	Juvenile		120	285	28	93	0.058
	Adult	S. Oregon/N. Calif coast - T	3	9	1	3	0.008
	Juvenile		373	887	87	288	0.016
	Adult	Oregon coast - T	64	219	15	63	0.009
	Juvenile		6,742	16,042	1,571	5,202	0.022
	Adult	Lower Columbia River - T	5	16	2	5	0.013
	Juvenile		521	1,238	122	402	0.027
Chum	Adult	Columbia River - T	1	1	1	1	0.001
	Juvenile		62	133	15	38	0.001
Sockeye	Adult	Snake River - E	0	0	0	0	--

Species	Life stage	ESU/DPS	Testing		Training		Percent of ESU/DPS (intact adipose) Killed or Injured
			Mortality (10 years)	Injury (10 years)	Mortality (10 years)	Injury (10 years)	
	Juvenile		1	1	1	1	0.003
Steelhead	Adult	South-Central California - T	0	1	0	0	0.005
	Juvenile		3	6	2	3	0.002
	Adult	Central Calif - T	1	1	0	1	0.007
	Juvenile		6	12	4	7	0.002
	Adult	California Central Valley - T	1	1	0	1	0.007
	Juvenile		6	12	4	6	0.002
	Adult	Northern Calif - T	1	1	1	1	0.003
	Juvenile		0	47	10	19	0.002
	Adult	Upper Columbia River - E	1	1	1	1	0.005
	Juvenile		13	28	6	11	0.002
	Adult	Snake River basin - T	4	9	2	3	0.001
	Juvenile		62	135	29	54	0.002
	Adult	Lower Columbia River - T	1	3	1	1	0.002
	Juvenile		20	43	10	18	0.002
	Adult	Upper Willamette River - T	1	2	1	1	0.003
	Juvenile		10	21	5	9	0.002
	Adult	Middle Columbia River - T	2	5	1	2	0.001
	Juvenile		24	52	12	21	0.002
	Adult	Puget Sound - T	1	3	1	1	0.001
	Juvenile		74	161	35	64	0.002
Eulachon	Adult	Southern - T	4	9	0	0	<0.001

Table 132. The number, or percent of ESU/DPS affected, of threatened or endangered salmonids that are likely to be killed or injured as a result of their exposure to Navy Training Activities conducted in the inland portion of the NWTT Action Area over a 10-year period. Also presented is the percent of the corresponding ESU/DPS that would be injured and/or killed.

Species	Life Stage	Natural (10 year)			Hatchery: adipose intact (10 year)			Hatchery: adipose clip (10 year)		
		Mortality	Injury	Percent affected	Mortality	Injury	Percent affected	Mortality	Injury	Percent affected
Chinook - T	Juveniles	5,212	13,380	0.080	909	2,333	0.005	1,993	5,115	0.002
	Adults	11	3	0.008	3	12	0.014	0	0	--
Steelhead - T	Juveniles	115	279	0.002	0	0	--	0	0	--
	Adults	14	12	0.019	0	0	--	0	0	--
Chum - T	Juveniles	1,924	7,037	0.029	192	701	0.032	0	0	--
	Adults	45	146	0.109	5	15	0.058	0	0	--

Table 133. The proportion of threatened or endangered rockfish that are likely to be killed or injured as a result of their exposure to Navy Training Activities conducted in the inland portion of the NWTT Action Area over a 10-year period.

Species	Life stage	Proportion of DPS injured/killed*
Puget Sound/Georgia Basin bocaccio - E	Larval	0.003
	Juvenile	0.002
Puget Sound/Georgia Basin canary rockfish - T	Larval	0.003
	Juvenile	0.002
Puget Sound/Georgia Basin yelloweye rockfish - T	Larval	0.003

***Take of ESA-listed rockfish species is not estimated by the number of individuals affected, but is expressed in terms of the proportion of the population affected. See Section 6.11.1.2.5 for further detail.**

9.1.1 Activity Levels as Indicators of Take

As discussed in this Opinion, the estimated take of ESA-listed sea turtles and 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 NWTT 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.

Detection of injury or mortality of larval, juvenile, or adult individuals of fish inshore or offshore during Navy training and testing activities will also be extremely difficult. Although we have established as a term and condition (number seven) a reasonable monitoring and reporting program for the take of fish during underwater detonations (EOD activities) in the inshore areas, monitoring techniques to calculate actual take of larval, juvenile, or adult fish including detection and collection of individuals and assessment of injuries or death is not feasible at the scale of NWTT activities. Therefore, in addition to the inshore monitoring and reporting program required by the terms and conditions, we must rely on predicted take associated with levels of activities and any opportunistic observations of potential injured or dead larval, juvenile, or adult fish during training and testing activities to determine when anticipated take levels have been exceeded. "Take" of these species will have been exceeded if the number of detonations, the location of the detonations, or the Net Explosive Weight of the detonations are greater than we expected in our analyses or if the monitoring program associated with the Navy's activities detects a greater number of any fish species than are identified in the preceding tables.

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 127 and Table 128 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, sea turtles, 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, sea turtles, fish) resulting from the proposed training and testing activities and any observations of stranded or dead ESA-listed marine mammals and sea turtles that are not attributable to Navy training and testing but are observed during the course of Navy training and testing activities and while implementing monitoring requirements required by this Opinion and the MMPA LOAs.

3. The Navy and NMFS' Permits Division shall report any activities not included in the Description of the Action in Section 2 prior to implementation (if operational security allows or as soon as operational security considerations allow after the relevant activity is conducted) and any exceedances of activity levels (Table 2, Table 5, and Table 6) immediately upon determining that a planned activity may exceed these levels or that these levels have been exceeded. Exceedance of an activity will require the Navy to reinitiate consultation.

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 exercises and testing activities, and any potential to exceed levels of training and testing analyzed in this Opinion they conduct in the NWTT Action Area over the five year period of the MMPA regulations and letters of authorization to help assess the actual amount or extent of take incidental to training and testing 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. If a dead or injured marine mammal or leatherback sea turtle is observed during the training and testing activities, the Navy shall immediately contact NMFS and appropriate stranding networks.

2. The Navy shall monitor and coordinate with marine mammal and sea turtle stranding networks to help determine any potential relationship of any stranding with Navy training and testing activities.
3. If the Navy plans to conduct a training or testing activity that was not assessed in this Opinion as part of the proposed action, the Navy shall contact NMFS prior to executing the activity (if operational security allows or as soon as operational security considerations allow after the relevant activity is conducted) and shall reinitiate consultation as appropriate.

4. The Navy must report to NMFS any exceedance of activity levels (e.g. sonar hours and the type and numbers of explosives used) of planned testing or training events specified in the preceding Opinion and in the final MMPA rule before the exceedance occurs and 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.

5. The Navy shall implement all mitigation and monitoring measures as proposed in the action described in the Final EIS/OEIS and consultation initiation package, as specified in the final MMPA rule and LOAs, and as described in this Opinion in Sections 0 and 2.3.

6. NMFS' Permits Division shall ensure that all mitigation and monitoring measures as proposed by the Navy and as proposed by NMFS in the final MMPA rule and in Sections 0 and 2.3 of this Opinion are implemented by the U.S. Navy through the issuance of a final rule and subsequent letters of authorization (LOA) pursuant to the MMPA.

7. The Navy shall accomplish the following monitoring for take of fish during underwater detonations (EOD activities) in the inshore areas:

   a. To the extent practicable, minimize potential effects of underwater detonations by scheduling training and testing events during periods of the year when salmonid abundance is lowest.

b. Survey underwater detonation areas prior to each event to inform "go", "no go" decisions and to minimize the potential for interactions with ESA-listed species. Surveys should attempt to confirm the absence of salmonids and/or indicators of the presence of salmonids in the mitigation zone. The Navy should employ visual observations from boats, and other available technology to detect fish and other indicators consistent with standard operating procedures and as deemed practicable.

c. After each explosion or at the conclusion of multiple explosion events, survey the impact area and areas immediately downstream to detect possible injured or killed salmonids. If injured or killed fish are detected, consistent with standard operating procedures and safety, try to determine the species affected and estimate the number of adult and juvenile fish.

d. Prepare a report of compliance after each EOD event.

e. Compile and provide hydrophone data from underwater detonation events to verify assumptions used in the derivation of ranges to effects to fish from explosives. If there is sufficient data to make conclusions on ranges to effect from all source classes (NEW) this condition will be met. If not, continue to collect measurements of received sound levels at various distances from the source using hydrophones and other appropriate devices as needed.

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. Monitor sighting, location, and stranding data for ESA-listed marine mammals and sea turtle species in the NWTT Action Area.
2. As practicable, develop procedures to aid any individuals of an ESA-listed marine mammals, sea turtles, and fish that have been impacted by NWTT activities and is in a condition requiring assistance to increase likelihood of survival.
3. 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.
4. Continue technical assistance/adaptive management efforts with NMFS to help inform future consultations on Navy training and testing in the NWTT Action Area.

5. The Navy should coordinate with NMFS to understand acoustic effects to fish and to develop criteria for future consultations.
6. The Navy should explore methods to better quantify behavioral take of sea turtles not rising to the level of TTS.

11 REINITIATION OF CONSULTATION

This concludes formal consultation on proposed Northwest Training and Testing activities the Navy will conduct and NMFS's promulgation of regulations and issuance of incidental take authorizations pursuant to the MMPA from November 2015 through November 2020. 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.

12 INFORMATION QUALITY ACT

Section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Public Law 106-554, AKA the Data Quality Act or Information Quality Act) directed the Office of Management and Budget (OMB) to issue government-wide guidelines that "provide policy and procedural guidance to federal agencies for ensuring and maximizing the quality, objectivity, utility, and integrity of information (including statistical information) disseminated by federal agencies." OMB complied by issuing guidelines which direct each federal agency to 1) issue its own guidelines; 2) establish administrative mechanisms allowing affected persons to seek and obtain correction of information that does not comply with the OMB 515 Guidelines or the agency guidelines; and 3) report periodically to OMB on the number and nature of complaints received by the agency and how the complaints were handled. The OMB Guidelines can be found at:

http://www.whitehouse.gov/omb/fedreg/reproducible2.pdf

The Department of Commerce Guidelines can be found at:
http://ocio.os.doc.gov/ITPolicyandPrograms/Information_Quality/index.htm

The NOAA Section 515 Information Quality Guidelines, created with input and reviews from each of the components of NOAA Fisheries, went into effect on October 1, 2002. The NOAA Information Quality Guidelines are posted on the NOAA Office of the Chief Information Officer Webpage. http://www.cio.noaa.gov/Policy_Programs/info_quality.html

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