NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 BIOLOGICAL OPINION

Action Agencies:	The United States Navy (U.S. Navy) and NOAA's National Marine Fisheries Service, Office of Protected Resources' Permits and Conservation Division
Activity Considered:	(1) The U.S. Navy's Training Exercises and Testing Activities in the Hawaii-Southern California Training and Testing Study Area (2) National Marine Fisheries Services' promulgation of regulations pursuant to the Marine Mammal Protection Act for the U.S. Navy to "take" marine mammals incidental to training exercises and testing activities in the Hawaii-Southern California Training and Testing Study Area from December 2013 through December 2018 (3) National Marine Fisheries Services' issuance of two Letters of Authorization pursuant to regulations under the Marine Mammal Protection Act to "take" marine mammals incidental to training exercises and testing activities in the Hawaii-Southern California Training and Testing Study Area from December 2013 through December 2018
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service
Public Consultation Tracking System Number:	FPR-2015-9111
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ACRONYMS AND ABBREVIATIONS

A-A	Air-to-Air
A-G	Air-to-Ground
A-S	Air-to-Surface
AAW	Anti-Air Warfare
ACM	Air Combat Maneuvering
AMW	Amphibious Warfare
ASUW	Anti-Surface Warfare
ASW	Anti-Submarine Warfare
BOMBEX	Bombing Exercise
COMPTUEX	Composite Training Unit Exercise
CPF	Commander, Pacific Fleet
CV	Coefficient of Variation
dB	Decibel
DoN	Department of Navy
DPS	Distinct Population Segment
EEZ	Exclusive Economic Zone
EIS	Environmental Impact Statement
EOD	Explosive Ordnance Disposal
ESA	Endangered Species Act
EXTORP	Running Torpedo Exercise
°F	Degrees Fahrenheit
ft	Feet
ft^2	Square Feet
GUNEX	Gun Exercise
HE	High Explosive
HSTT	Hawaii-Southern California Training and Testing
IEER	Improved Extended Echo Ranging
JTFEX	Joint Task Force Exercise
kHz	Kilohertz
Km	Kilometer
kts	Knots
lb	Pound
LOA	Letter of Authorization
m	Meter
m^2	Square Meter
MFA	Mid-Frequency Active
min	Minute
MIW	Mine Warfare
MMPA	Marine Mammal Protection Act
NAVSEA	Naval Sea Systems Command
NEPA	National Environmental Policy Act
NEW	Net Explosive Weight
nm	nautical miles
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration

NUWC	Naval Undersea Warfare Center
OEIS	Overseas Environmental Impact Statement
PTS	Permanent Threshold Shift
RHIB	Rigid Hull Inflatable Boats
RIMPAC	Rim of the Pacific Exercise
S-A	Surface-to-Air
S-S	Surface-to-Surface
SAR	Search and Rescue
SINKEX	Sinking Exercise
SONAR	Sound Navigation And Ranging
STW	Strike Warfare
SURTASS	Surveillance Towed Array Sensor System
TORPEX	Torpedo Exercise
TRACKEX	Tracking Exercise
TTS	Temporary Threshold Shift
USW	Undersea Warfare

1 INTRODUCTION

The Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7 (a)(2) of the ESA requires Federal agencies to consult with the United States Fish and Wildlife Service (USFWS), NMFS, or both (collectively the Services), to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy designated critical habitat. Section 7 (b)(3) requires that at the conclusion of consultation, the Services provide an opinion stating how the agencies' actions will affect listed species and their critical habitat. If incidental take is expected, section 7 (b)(4) requires the consulting agency to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts.

When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with NMFS or the 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)).

For the actions described in this document, the action agencies are the United States Navy (U.S. Navy), which proposes to continue military training exercises and testing activities, and NMFS Office of Protected Resources - Permits and Conservation Division (Permits Division), which has promulgated regulations pursuant to the Marine Mammal Protection Act of 1972, as amended (MMPA 16 U.S.C. 1361 et seq.) related to the U.S. Navy's activities in the Hawaii-Southern California Training and Testing (HSTT) Action Area that may affect several ESA-listed species. The regulations authorize the issuance of two Letters of Authorization (LOAs) that allow the U.S. 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.

This Opinion is based on information provided in the U.S. Navy's September 24, 2012, request for ESA consultation package, which included the *Hawaii-Southern California Training and Testing Draft Environmental Impact Statement/Overseas Environmental Impact Statement* (DEIS/OEIS dated November 2011) and the *Hawaii-Southern California Training and Testing Endangered Species Act Section 7 Consultation Supplemental Information* (Navy 2012a); and NMFS Permits Division's February 26, 2013, request for Section 7 consultation under the ESA, which included the proposed Federal regulations under the MMPA specific to the proposed activities (78 FR 6977). The DEIS/OEIS was subsequently updated and replaced with the *Hawaii-Southern California Training and Testing Final Environmental Impact Statement/Overseas Environmental Impact Statement* (FEIS/OEIS dated August 2013) (Navy 2013d). The proposed MMPA regulations were also finalized and published on December 24, 2013 (78 FR 78106). The information contained in the FEIS/OEIS and the final MMPA regulations were considered in the development of this Opinion. The Opinion is also based on information provided in the Navy's September 9, 2014 request to reinitiate formal consultation on the actions described above to assess potential effects on the Eastern Pacific Distinct Population Segment of scalloped hammerhead shark (*Sphyrna lewini*), recently listed as threatened under the ESA. We also considered issues regarding the prior biological opinion recently raised in litigation. 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 prior biological opinion.

The Navy proposes to continue to conduct training exercises and testing activities within the HSTT Study Area. Navy training exercises and testing activities have been ongoing in the same general geographic area since the 1940s. Most of the activities occurring in recent years (2008 to present) were analyzed in three separate environmental impact statements (EISs) completed between 2008 and 2011: the *Hawaii Range Complex (Hawaii Range Complex) EIS/OEIS* (Navy 2008a), the *Southern California (SOCAL) Range Complex EIS/OEIS* (Navy 2008d), and the *Silver Strand Training Complex (SSTC) EIS* (Navy 2011b).

This Opinion was 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' final Opinion on the effects of these actions on endangered and threatened species and critical habitat that has been designated for those species. This Opinion supersedes and replaces, in its entirety, the prior biological opinion dated April 23, 2014.

The NMFS completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The document will be available at http://www.nmfs.noaa.gov/pr/consultation/opinions.htm or via NMFS' Public Consultation Tracking System . A complete record of this consultation is on file at National Marine Fisheries Service, Office of Protected Resources, 1315 East-West Highway, Silver Spring, MD 20910.

1.1 Consultation History

On September 24, 2012, NMFS received a request for consultation pursuant to section 7 of the ESA on proposed training exercises and testing activities (i.e., training exercises and testing activities) to be conducted in the HSTT Study Area over five years.

Between September 2012 and September 2013 the ESA Interagency Cooperation Division of NMFS Office of Protected Resources had numerous meetings, phone calls, and exchanges of information that comprise the ESA section 7 consultation process. This included the pre-decision review of information and draft versions of this final biological opinion. Comments were received from the U.S. Navy on two draft biological opinions. NMFS Permits Division did not provide comments on the draft biological opinions. Comments were considered and addressed in this final document.

On November 5, 2012, we responded to the Navy's request for consultation indicating that we had received sufficient information to initiate formal consultation with the Navy. At this time we also determined that NMFS Permits Division's proposed action of promulgating a rule under the MMPA regulating the take of marine mammals and subsequent letters of authorization (LOAs) pursuant to those MMPA regulations authorizing "take" of marine mammals incidental to Navy proposed action in the HSTT Action Area was inter-dependent and interrelated and therefore should be included in the consultation. However, at that time we did not have sufficient information on the MMPA regulations or LOAs to initiate consultation on those actions. Further, due to the complexity of the proposed action and extent of species potentially affected, we proposed an extended consultation timeline with a final opinion issued no later than October 24, 2013.

On January 31, 2013, NMFS' Permits Division published a notice of proposed rulemaking and request for comments for Takes of Marine Mammals Incidental to Specified Activities; U.S. Navy Training and Testing Study Activities in Hawaii-Southern California Training and Testing Study Area.

On February 26, 2013, we received a request for consultation from NMFS Permits Division on its proposed issuance of regulations under the MMPA to take marine mammals during training exercises and testing activities in the HSTT Study Area. On May 29, 2013, NMFS' ESA Interagency Cooperation Division provided a copy of the preliminary draft biological opinion to the U.S. Navy per agreed upon milestones. The Navy provided comments on the preliminary draft on June 4, 2013. On August 16, 2013, NMFS' ESA Interagency Cooperation Division provided a copy of the revised draft biological opinion to the U.S. Navy per agreed upon milestones. The Navy provided comments on the revised draft biological opinion draft on August 24, 2013.

On July 7, 2013, NMFS' Permits Division provided a draft Final Rule for Takes of Marine Mammals Incidental to Specified Activities; U.S. Navy Training and Testing Activities in the Hawaii-Southern California Training and Testing Study Area. The Navy provided comments on the draft final rule on July 16, 2013.

On August 14, 2013, NMFS' Permits Division provided a revised Final Rule for Takes of Marine Mammals Incidental to Specified Activities; U.S. Navy Training and Testing Activities in Hawaii-Southern California Training and Testing Study Area. The Navy provided comments on the revised Final Rule on August 21, 2013.

On September 26, 2013, NMFS’ Permits Division provided a revised Final Rule for Takes of Marine Mammals Incidental to Specified Activities; U.S. Navy Training and Testing Activities in the Hawaii-Southern California Training and Testing Study Area.

On November 6, 2013, NMFS’ ESA Interagency Cooperation Division provided a copy of the draft final biological opinion to the U.S. Navy, upon their request. The Navy provided comments on the draft biological opinion on November 14, 2013.

On December 13, 2013, NMFS issued a final biological opinion to the Navy. After issuance of the biological opinion, NMFS identified several inadvertent factual errors and omissions regarding amount of incidental take of species from vessel strike and omission of potential sea turtle injury or mortality from the incidental take statement. These errors and omissions were the result of oversights during the drafting process, and the biological opinion (primarily the incidental take statement) was corrected. Because the corrections were for drafting and computational errors, and were not the result of any new analyses, new information, or new facts not previouslt considered, the corrections did not require reinitiation of consultation under 50 C.F.R. 402.16.

On April 23, 2014, NMFS issued a corrected final biological opinion and incidental take statement that superceded the December 13, 2013 biological opinion.

On July 3, 2014, NMFS issued a final determination to list the Central and Southwest (SW) Atlantic Distinct Population Segment (DPS) and the Indo-West Pacific DPS of scalloped hammerhead shark (*Sphyrna lewini*) as threatened species, and to list the Eastern Atlantic DPS and Eastern Pacific DPS of scalloped hammerhead sharks as endangered species under the ESA. Critical habitat was not designated for any of the DPSs. NMFS intends to consider critical habitat for the Central & SW Atlantic, Indo-West Pacific, and Eastern Pacific DPSs in a separate rulemaking.

On September 9, 2014, NMFS received the Navy's request to reinitiate formal consultation pursuant to the ESA on the Navy’s HSTT activities and effects on the Eastern Pacific Distinct Population Segment (DPS) of scalloped hammerhead shark (*Sphyrna lewini*). In the Biological Evaluation accompanying the Navy's request for reinitiation, the Navy determined the proposed activities are likely to adversely affect the newly listed Eastern Pacific DPS of scalloped hammerhead shark due to stressors resulting from surface and bottom detonation from Mine Neutralization training events in the near shore waters < 24 m (80 ft) of the Silver Strand Training Complex and surface and sub-surface detonations associated with at-sea training ordnance use in the offshore pelagic waters > 16 to 32 km offshore in the Southern California (SOCAL) Range Complex.

On November 21, 2014, NMFS determined that there was sufficient information to reinitiate formal consultation as the Navy requested. NMFS also expanded the scope of the reinitiated consultation to include a re-examination of NMFS's analysis of effects to listed cetaceans, pinnipeds, and sea turtles.

On December 2, 2014, Navy and NMFS held a teleconference to discuss initial steps necessary for both agencies to proceed forward with this reinitiated consultation. We agreed that Navy and NMFS would review training and testing activities planned for the expected duration of this reinitiated consultation to assess whether our analysis of effects (including potential for vessel strikes) remains consistent with what was evaluated under the initial HSTT consultation. Additionally, Navy and NMFS agreed to conduct a joint review of pertinent scientific literature published since th issuance of the prior biological opinion on April 23, 2014.

On December 5, 2014, the U.S. Pacific Fleet released a naval message to all subordinate commands, requiring mitigation for all hammerhead sharks prior to underwater detonations in the same manner employed for marine mammals and sea turtles. This message served to implement protective measures for ESA-listed scalloped hammerhead sharks until the reinitiation is completed and the Navy's Protective Measures Assessment Protocol can be updated with these mitigation requirements.

On December 9, 2014, the Navy provided NMFS with a letter documenting the Navy's compliance with Section 7(a) (2) and Section 7(d) of the ESA during the reinited consultation period. The Navy reviewed projected training and testing activities anticipated in the Hawaii and SOCAL study area between December 2014 and April 2015, when the consultation is anticipated to conclude.

On December 11, 2014, NMFS provided a letter to Navy reflecting its review of information provided by the Navy on December 9th and NMFS’ determination that the Navy's planned activities during the reinitiated consultation period are consistent with the actions analyzed in the April 23, 2014 biological opinion, which concluded that HSTT activities would not result in jeopardy of ESA-listed species or destruction or adverse modification of designated critical habitat.

On January 14, 2015, NMFS and the Navy met to refine the approach to vessel strike analysis for large whales in the reinitiated Opinion.

On January 16, 2015, NMFS' West Coast Stranding Coordinator provided an update of large whale ship strikes for California through 2014.

On March 12, 2015, NMFS’ ESA Interagency Cooperation Division provided a copy of the draft final reinitiated biological opinion to the U.S. Navy, upon their request. The Navy provided comments on the draft biological opinion on March 20, 2015.

On March 24, 2015 NMFS and the Navy met to review new scientific information that became available since the issuance of the 2013 Opinion.

On March 31, 2015, the United States District Court for the District of Hawaii ruled that NMFS' biological opinion for the Navy's activities in the HSTT Action Area included an arbitrary and capricious "no jeopardy" finding for whales and sea turtles and an invalid incidental take statement for sea turtles (Conservation Council for Hawaii v. National Marine Fisheries Service; Natural Resources Defense Council v. National Marine Fisheries Service). The court identified three primary issues with the biological opinion and incidental take statement. First, it found that NMFS did not adequately support its conclusion that authorized mortalities of large whales will not appreciably reduce the likelihood of both the survival and recovery of affected large whale species in the wild. Second, the court determined that NMFS failed to support its no jeopardy finding for sea turtles with adequate analysis. Third, the court found the incidental take statement for sea turtles deficient because it failed to provide either a numerical cap on sea turtle take by vessel strike or a surrogate to trigger reinitiation of consultation. This reinitiated biological opinion addresses the issues identified by the court.

The court also found that NMFS' negligible impact and least adverse impact determinations under the Marine Mammal Protection Act (MMPA) were arbitrary and capricious, and found flaws with the Final Environmental Impact Statement that the Navy issued under the National Environmental Policy Act and that NMFS adopted. Under the court's scheduling order, appropriate relief is to be addressed in future proceedings. In the event the MMPA rule is vacated, the provisions of the Incidental Take Statement pertaining to listed marine mammals would become operative only upon issuance of new MMPA take authorization or if Navy testing and training activities in the HSTT Action Area are exempted from the requirements of the MMPA pursuant to 16 USC 1371(f).

2 DESCRIPTION OF THE 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 U.S. Navy's military training and testing (i.e., readiness activities) conducted in the HSTT Study Area; (2) the regulations promulgated by NMFS's Permits Division pursuant to the MMPA governing the U.S. Navy's "take" of marine mammals incidental to the Navy's military readiness activities from December 2013 through December 2018; and (3) NMFS's Permits Division's LOAs issued pursuant to the regulations that authorizes the U.S. Navy to "take" marine mammals incidental to military readiness activities in the HSTT Study Area through December 2018. This Opinion supersedes the April 23, 2014, biological opinion.

The purpose of the activities the U.S. Navy conducts in the HSTT Study Area is to meet the requirements of the U.S. 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 U.S. Navy to "take" marine mammals incidental to military readiness activities in the HSTT Study Area conducted through December 2018 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 activities proposed for the remainder of the five year period of the MMPA Rule would continue into the reasonably foreseeable future at levels similar to that assessed in this Opinion and described in the 2013 HSTT EIS/OEIS and MMPA rule.

The tempo of training within the HSTT Study Area is subject to variation within the scope of the activities described in the Navy's HSTT 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 HSTT; and
- Certification and training needs for a given ship, submarine, or aircraft crew (e.g., some units could require a certain amount of one kind of training versus another).

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

2.1 Navy Training Activities

The Navy has categorized training exercises into eight functional warfare areas (anti-air warfare; amphibious warfare; strike warfare; anti-surface warfare; anti-submarine warfare; electronic warfare; mine warfare; and naval special warfare), which are briefly described below. Details regarding each warfare area can be found in the *Hawaii-Southern California Training and Testing Final Environmental Impact Statement/Overseas Environmental Impact Statement*

(FEIS/OEIS) (Navy 2013e). Table 1 and the following sections provide an overview of typical training activities in the HSTT Study Area.

Table 1. Typical Training Activities in the Hawaii-SOCALTraining and Testing 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.
Air Defense Exercises (ADEX)	Aircrew and ship crews conduct defensive measures against threat aircraft or missiles.
Gunnery Exercise (Air-to-Air) (GUNEX [A-A])	Aircrews defend against threat aircraft with cannons (machine gun).
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 missiles and aircraft with guns.
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	Surface ship crews defend against threat missiles and aircraft with missiles.
Amphibious Warfare (AMW)	
Naval Surface Fire Support Exercise–Land-based target (FIREX [Land])	Surface ship crews use large-caliber guns to fire on land-based targets in support of forces ashore.
Naval Surface Fire Support Exercise-At Sea (FIREX [At Sea])	Surface ship crews use large-caliber guns to support forces ashore; however, the land target is simulated at sea. Rounds impact the water and are scored by passive acoustic hydrophones located at or near the target area.
Amphibious Assault	Forces move ashore from ships at sea for the immediate execution of inland objectives.
Amphibious Raid/Humanitarian Assistance Operations	Small unit forces move ashore swiftly from ships at sea for a specific short-term mission. These operations with as few personnel as possible.
Strike Warfare (SW)	
Bombing Exercise Air-to-Ground (BOMBEX A-G)	Fixed-wing aircraft drop non-explosive bombs against a land target.
Anti-Surface Warfare (ASUW)	
Maritime Security Operations (MSO)	Helicopter and surface ship crews conduct a suite of maritime security operations (e.g., visit, board, search, and seizure; maritime interdiction operations; force protection; and anti-piracy operation).
Gunnery Exercise (Surface-to-Surface) (Ship) (GUNEX [S-S] – Ship)	Ship crews engage surface targets with ship's small, medium, and large caliber guns.
Gunnery Exercise (Surface-to-Surface) (Boat) (GUNEX [S-S] – Boat)	Small boat crews engage surface targets with small and medium-caliber guns.
Missile Exercise (Surface-to-Surface) (MISSILEX [S-S])	Surface ship crews defend against threat missiles and other surface ships with missiles.
Gunnery Exercise (Air-to-Surface) (GUNEX [A-S])	Fixed-wing and helicopter aircrews, including embarked personnel, use small and medium-caliber guns to engage surface targets.
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	Fixed-wing and helicopter aircrews fire both precision-guided missiles and unguided rockets against surface targets.
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Fixed-wing aircrews deliver bombs against surface targets.

Laser Targeting	Fixed-winged, helicopter, and ship crews use single or multi-beam lasers to illuminate enemy targets or to defend against approaching hostile forces.
Sinking Exercise (SINKEX)	Aircraft, ship, and submarine crews deliver ordnance on a seaborne target, usually a deactivated ship, which is deliberately sunk using multiple weapon systems.
Anti-Submarine Warfare (ASW)	
Tracking Exercise/ Torpedo Exercise –Submarine (TRACKEX/TORPEX-Sub)	Submarine crews search, track, and detect submarines. Exercise torpedoes may be used during this event.
Tracking Exercise/ Torpedo Exercise –Surface (TRACKEX/TORPEX-Surface)	Surface ship crews search, track and detect submarines. Exercise torpedoes may be used during this event.
Tracking Exercise/ Torpedo Exercise –Helicopter (TRACKEX/TORPEX-Helo)	Helicopter crews search, detect and track submarines. Recoverable air launched torpedoes may be employed against submarine targets.
Tracking Exercise/ Torpedo Exercise - Maritime Patrol Aircraft (TRACKEX/TORPEX - MPA)	Maritime patrol aircraft crews search, detect, and track submarines. Recoverable air launched torpedoes may be employed against submarine targets.
Tracking Exercise-Maritime Patrol Aircraft Extended Echo Ranging Sonobuoy (TRACKEX–MPA sonobuoy)	Maritime patrol aircraft crews search, detect, and track submarines with extended echo ranging sonobuoys. Recoverable air launched torpedoes may be employed against submarine targets.
Electronic Warfare (EW)	
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 actions.
Counter Targeting - Flare Exercise (FLAREX)	Fixed-winged aircraft and helicopters crews defend against an attack by deploying flares to disrupt threat infrared missile guidance systems.
Counter Targeting - Chaff Exercise (CHAFFEX)	Surface ships, fixed-winged aircraft and helicopter crews defend against an attack by deploying chaff, a radar reflective material, which disrupt threat targeting and missile guidance radars.
Mine Warfare (MIW)	
Mine Countermeasures Exercise (MCM) - Ship Sonar	Littoral combat ship crews detect and avoid mines while navigating restricted areas or channels using active sonar.
Explosive Ordnance Disposal (EOD)/Mine Neutralization	Personnel disable threat mines. Explosive charges may be used.
Mine Countermeasures -Towed Mine Neutralization	Ship crews and helicopter aircrews tow systems (e.g., Organic and Surface Influence Sweep, MK 104/105) through the water designed to disable and/or trigger mines.
Mine Countermeasures - Mine Detection	Ship crews and helicopter aircrews detect mines using towed and laser mine detection systems (e.g., AN/AQS-20, Airborne Laser Mine Detection System).
Mine Countermeasures – Mine Neutralization	Ship crews and helicopter aircrews disable mines by firing small and medium-caliber projectiles.
Mine Countermeasures - Mine Neutralization – Remotely Operated Vehicles	Ship crews and helicopter aircrews disable mines using remotely operated underwater vehicles.
Mine Laying	Fixed-winged aircraft and submarine crews drop/launch non explosive mine shapes.
Civilian Port Defense	Maritime security operations for military and civilian ports and harbors. Because other stressors associated with this activity were concluded by Navy to be "no effect", only the sonar portion of this activity is analyzed in this document.

2.1.1 Naval Special Warfare

The mission of naval special warfare is to conduct unconventional warfare, direct action, combat terrorism, special reconnaissance, security assistance, counter-drug operations, and recovery of personnel from hostile situations. Naval special warfare operations are highly specialized and require continual and intense training.

Naval special warfare units utilize a combination of specialized training, equipment, and tactics, including insertion and extraction operations using parachutes, submerged vehicles, rubber boats, and helicopters; boat-to-shore and boat-to-boat gunnery; underwater demolition training; reconnaissance; and small arms training.

2.1.2 Major Training Exercises

Major training exercises are not included in the primary mission areas but comprise many smaller or "unit level" range exercises conducted by several units operating together while commanded and controlled by a single commander. These exercises typically employ an exercise scenario developed to train and evaluate the strike group in naval tactical tasks. In a major training event, most of the operations and activities being directed and coordinated by the strike group commander are identical in nature to the operations conducted during individual, crew, and smaller-unit training events. In a major training event, however, these disparate training tasks are conducted in concert, rather than in isolation. Typical major training exercises are described in Table 2.

Table 2. Typical Major Training Exercises proposed to occur in the Hawaii-Southern California Training and Testing Study Area.

Activity Name	Activity Description
Composite Training Unit Exercise (COMPTUEX)	Intermediate level exercise designed to create a cohesive Strike Group prior to deployment or joint task force exercise. Typically seven surface ships, helicopters, maritime patrol aircraft, two submarines, and various unmanned vehicles. Marine mammal systems may be used during the exercise.
Joint Task Force Exercise (JTFEX) / Sustainment Exercise (SUSTAINEX)	Final fleet exercise prior to deployment of the Strike Group. Serves as a ready-to-deploy certification for all units involved. Typically nine surface ships, helicopters, maritime patrol aircraft, two submarines, and various unmanned vehicles. Marine mammal systems may be used during the exercise.
Anti-Submarine Warfare-Composite Training Unit Exercise (COMPTUEX)	Anti-submarine warfare activities conducted during a composite training unit exercise.
Integrated Anti-Submarine Warfare Course (IAC)	Multiple ships, aircraft, and submarines coordinate use of sensors, including sonobuoys, to search, detect and track threat submarines. Integrated Anti-Submarine Warfare Course is an intermediate level training event and can occur in conjunction with other major exercises.
Group Sail	Multiple ships and helicopters integrate the use of sensors, including sonobuoys, to search, detect and track a threat submarine. Group sails are not dedicated anti-submarine warfare events and involve multiple warfare areas.

2.1.3 Other Training Exercises

Other training exercises that do not fall under a particular category are described in Table 3.

Table 3. Typical Other Training Exercises proposed to occur in the Hawaii-SOCALTraining and Testing Study Area.

Activity Name	Activity Description
Precision Anchoring	Ship crews train in releasing of anchors in designated locations.
Submarine Navigational (SUB NAV)	Submarine crews locate underwater objects and ships while transiting in and out of port.
Submarine Navigation Under Ice Certification	Submarine crews train to operate under ice. During training and certification other submarines and ships simulate ice.
Surface Ship Sonar Maintenance	Pierside and at-sea maintenance of sonar systems.
Submarine Sonar Maintenance	Pierside and at-sea maintenance of sonar systems.

2.1.4 Training Activity Levels

Table 4 provides a summary of training exercises (as described above), including tempo and quantities of inert and live munitions that the U.S. plans to expend during training, that were analyzed by the U.S. Navy. Items with high explosives (HE) that may have greater effects are bolded in the table.

Table 4. Training exercises in the HSTT Study Area (adapted from Table 2.8-1, Alternative 2, U.S. Navy Final EIS/OEIS, August 2013).

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
Anti-Air Warfare (AAW)			
Air Combat Maneuver (ACM)	814	n/a	Hawaii Range Complex
	3,970	n/a	SOCAL Range Complex
Air Defense Exercise (ADEX)	185	n/a	Hawaii Range Complex
	550	n/a	SOCAL Range Complex
Gunnery Exercise (Air-to-Air) – Medium-Caliber (GUNEX [A-A] – Medium-Caliber	3	3,000 rounds	SOCAL Range Complex
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	27	**105 missiles (53 HE)**	Hawaii Range Complex
	25	**52 missiles (26HE)**	SOCAL Range Complex
Gunnery Exercise (Surface-to-Air) – Large-Caliber (GUNEX [S-A]) – Large-Caliber	50	**400 HE rounds**	Hawaii Range Complex
	160	1,300 rounds	SOCAL Range Complex
Gunnery Exercise (Surface-to-Air) – Medium-Caliber (GUNEX [S-A]) – Medium-Caliber	70	140,000 rounds	Hawaii Range Complex
	190	380,000 rounds	SOCAL Range Complex
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	30	**30 HE missiles**	Hawaii Range Complex
	20	**20 HE missiles**	SOCAL Range Complex
Missile Exercise- Man-portable Air Defense System (MISSILEX–	4	**68 HE missiles**	SOCAL Range Complex

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
MANPADS)			
Amphibious Warfare (AMW)			
Naval Surface Fire Support Exercise – Land-Based Target (FIREX [Land])[4]	52	8,500 rounds- all ashore	SOCAL Range Complex
Naval Surface Fire Support Exercise – At Sea (FIREX [At Sea])	12	1,000 NEPM rounds; **840 HE rounds**	Hawaii Range Complex
Amphibious Assault	12	n/a	Hawaii Range Complex
	18	n/a	SSTC boat Lanes 11-14
Amphibious Raid – Battalion Landing	2	n/a	SOCAL Range Complex
Amphibious Raid	2,342	n/a	SOCAL Range Complex
	84	n/a	SSTC Boat Lanes 1-8
Expeditionary Fires Exercise/ Supporting Arms Coordination Exercise (EFEX/SACEX)	8	1,045 rounds- all ashore	SOCAL Range Complex
Humanitarian Assistance Operations	2	n/a	Hawaii Range Complex
Strike Warfare (STW)			
Bombing Exercise (Air-to-Ground) (BOMBEX A-G)	60	275 bombs (**no HE**)	Hawaii Range Complex
Gunnery Exercise (Air-to-Ground) (GUNEX A-G)	307	60,000 small and medium calibre rounds	Hawaii Range Complex
Anti-Surface Warfare (ASUW)			
Maritime Security Operations (MSO)	70	n/a	Hawaii OPAREA
	150	n/a	SOCAL Range Complex
	42	n/a	SSTC Boat Lanes 1-10
Gunnery Exercise (Surface-to-Surface) – Ship Small-Caliber (GUNEX [S-S] – Ship) Small-Caliber	60	318,000 rounds	Hawaii Range Complex
	350	1,855,000 rounds	SOCAL Range Complex
	16	84,000 rounds	HSTT Transit Corridor
Gunnery Exercise (Surface-to-Surface) – Ship Medium-Caliber (GUNEX [S-S] – Ship) Medium-Caliber	44	4,800 rounds (**440 HE**)	Hawaii Range Complex
	164	20,800 rounds (**1,640 HE**)	SOCAL Range Complex
	32	6,400 rounds (**320 HE**)	HSTT Transit Lane
Gunnery Exercise (Surface-to-Surface) - Ship Large-Caliber (GUNEX [S-S] - Ship) - Large-Caliber	60	1,000 rounds (**934 HE**)	Hawaii Range Complex
	190	8,500 rounds (**4,204 HE**)	SOCAL Range Complex
	16	400 rounds (**20 HE**)	HSTT Transit Lane
Gunnery Exercise (Surface-to-Surface) - Boat	200	600,000	SOCAL Range Complex

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
Small-Caliber (GUNEX [S-S] - Boat) - Small-Caliber			
Gunnery Exercise (Surface-to-Surface) – Boat Medium-Caliber (GUNEX [S-S] - Boat) - Medium-Caliber	10	**100 HE rounds, 100 HE grenades**, 200 NEPM rounds	Hawaii Range Complex
	14	**140 HE rounds, 140 HE grenades**, 240 NEPM rounds	SOCAL Range Complex
Missile Exercise (Surface-to-Surface) (MISSILEX [S-S])	12	12 Missiles	Hawaii Range Complex
	4	4 Missiles	SOCAL Range Complex
Gunnery Exercise (Air-to-Surface) – Small-Caliber (GUNEX [A-S]) – Small-Caliber	275	74,000 rounds	Hawaii Range Complex
	131	104,800	SOCAL Range Complex
Gunnery Exercise [Air-to-Surface] – Medium-Caliber (GUNEX [A-S]) – Medium-Caliber	130	27,000 (**6,000 HE**)	Hawaii Range Complex
	100	48,000 rounds (**12,000 HE**)	SOCAL Range Complex
Missile Exercise (Air-to-Surface) – Rocket (MISSILEX [A-S]) - Rocket	20	**760 HE rockets**	Hawaii Range Complex
	130	**3,800 HE rockets**	SOCAL Range Complex
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	57	**57 HE missiles**	Hawaii Range Complex
	214	**214 HE missiles**	SOCAL Range Complex
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	28	180 bombs (**56 HE**)	Hawaii Range Complex
	120	1,280 bombs (**160 HE**)	SOCAL Range Complex
Laser Targeting	50	n/a	Hawaii Range Complex
	250	n/a	SOCAL Range Complex
Sinking Exercise (SINKEX)	6	36 bombs (**18 HE**), 10 missiles (**6 HE**), 300 large-caliber rounds (**120 HE**), 6 **MK 48 HE torpedoes**, 12,000 medium-caliber NEPM	Hawaii Range Complex
	2	12 bombs (**6 HE**), 4 missiles (**2 HE**), 100 large-caliber rounds (**40 HE**), **2 MK 48 HE torpedoes**, 4,000 medium-caliber NEPM	SOCAL Range Complex
Anti-Submarine Warfare (ASW)			
Tracking Exercise/ Torpedo Exercise – Submarine (TRACKEX/	127	244 MK 48 EXTORP	Hawaii Range Complex
	63	76 MK 48 EXTORP	SOCAL Range Complex

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
TORPEX – Sub)	7	n/a	HSTT Transit Lane
Tracking Exercise/ Torpedo Exercise – Surface (TRACKEX/ TORPEX – Surface)	274	20 EXTORP, 30 REXTORP	Hawaii Range Complex
	540	48 EXTORP, 69 REXTORP	SOCAL Range Complex
Tracking Exercise/ Torpedo Exercise – Helicopter (TRACKEX/ TORPEX – Helo)	165	6 EXTORP, 110 REXTORP	Hawaii Range Complex
	628	6 EXTORP, 200 REXTORP	SOCAL Range Complex
	6	n/a	HSTT Transit Lane
Tracking Exercise/Torpedo Exercise - Maritime Patrol Aircraft (TRACKEX/TORPEX – MPA)	296	20 EXTORP, 210 REXTORP	Hawaii Range Complex
	116	24 EXTORP, 17 REXTORP	SOCAL Range Complex
Tracking Exercise - Maritime Patrol Advanced Extended Echo Ranging Sonsbuoys	96	480 IEER buoys, 1,440 MAC buoys	Hawaii Range Complex
	48	120 IEER buoys, 360 MAC buoys	SOCAL Range Complex
Kilo Dip-Helicopter	1,060	n/a	SOCAL Range Complex
Submarine Command Course (SCC) Operations	2	30 MK 54, 72 MK 48, EXTORP	Hawaii Range Complex
Electronic Warfare (EW)			
Electronic Warfare Operations (EW Ops)	33	n/a	Hawaii OPAREA
	350	n/a	SOCAL Range Complex
Counter Targeting Flare Exercise (FLAREX)	8	n/a	Hawaii OPAREA
	25	n/a	SOCAL Range Complex
Counter Targeting Chaff Exercise (CHAFFEX) - Ship	37	n/a	Hawaii OPAREA
	125	n/a	SOCAL Range Complex
Counter Targeting Chaff Exercise (CHAFFEX) – Aircraft	30	n/a	Hawaii OPAREA
	250	n/a	SOCAL Range Complex
Mine Warfare (MIW)			
Mine Countermeasures Exercise (MCM) – Ship Sonar	30	n/a	Hawaii Range Complex
	92	n/a	SOCAL Range Complex
Mine Countermeasure Exercise- Surface (SMCMEX)	266	n/a	SOCAL Range Complex
Mine Neutralization – Explosive Ordnance Disposal (EOD)	22	**82 HE**	Hawaii Range Complex
	75	**300 HE**	SOCAL Range Complex
	279	**414 HE**	SSTC Boat Lanes 1-14
Mine Countermeasure (MCM)- Towed Mine Neutralization	240	n/a	SOCAL Range Complex
	100	n/a	SSTC Boat Lanes 1-14

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
Mine Countermeasure (MCM) – Mine Neutralization	36	360 rounds	SOCAL Range Complex
Airborne Mine Countermeasure (AMCM) – Mine Detection	630	n/a	SOCAL Range Complex
	372	n/a	SSTC Boat Lanes 1-14
Mine Countermeasure (MCM) - Mine Neutralization – Remotely Operated Vehicle	60	**8 HE**	SOCAL Range Complex
	312	**20 HE**	SSTC Boat Lanes 1-14
Mine Laying	32	384 mine shapes	Hawaii Range Complex
	18	750 mine shapes	SOCAL Range Complex
Marine Mammal System	10	n/a	Hawaii Range Complex
	175	**8 HE**	SSTC Boat Lanes 1-14
Shock Wave Action Generator	90	**90 HE**	SSTC Boat Lanes 1-14
Surf Zone Test Detachment/Equipment Test and Evaluation	200	n/a	SSTC Boat Lanes 1-14
Submarine Mine Exercise	34	n/a	Hawaii Range Complex
	32	n/a	SOCAL Range Complex
Civilian Port Defense	1	**4 HE**	Pearl Harbor
	1	**4 HE**	San Diego
Naval Special Warfare (NSW)			
Personnel Insertion/Extraction- Submarine	145	n/a	SSTC Boat Lanes 1-10
Personnel Insertion/Extraction-Non-Submarine	15	n/a	SOCAL Range Complex
	394	n/a	SSTC Boat Lanes 1-14
Underwater Demolition Multiple Charge- Mat Weave and Obstacle Loading	18	18 HE	SOCAL Range Complex
Underwater Demolition Qualification/Certification	24	30 HE	SSTC Boat Lanes 1-14
Major Training Events			
Composite Training Unit Exercise (COMPTUEX)6	4	Exercise is comprised of various activities accounted for elsewhere within Table	SOCAL Range Complex
Joint Task Force Exercise (JTFX)/ Sustainment Exercise (SUSTAINEX)	5	Exercise is comprised of various activities accounted for elsewhere within Table	SOCAL Range Complex
Rim of the Pacific (RIMPAC) Exercise	1	Exercise is comprised of various activities	Hawaii Range Complex/SOCAL Range

Range Activity	No. of Events (per Year)	Ordnance (Number per Year)	Location
		accounted for elsewhere within Table	Complex
Multi-Strike Group Exercise	1	n/a	Hawaii Range Complex
Integrated Anti-Submarine Warfare Course (IAC)	4	Exercise is comprised of various activities accounted for elsewhere within Table	SOCAL Range Complex
Group Sail	2	Exercise is comprised of various activities accounted for elsewhere within Table	Hawaii Range Complex
	8	Exercise is comprised of various activities accounted for elsewhere within Table	SOCAL Range Complex
Undersea Warfare Exercise (USWEX)	5	Exercise is comprised of various activities accounted for elsewhere within Table	Hawaii Range Complex
Ship ASW Readiness and Evaluation Measuring (SHAREM)	2	8 MK 48 EXTORP, 16 MK 46/54 EXTORP	SOCAL Range Complex
Other			
Precision Anchoring	18	n/a	Hawaii Range Complex
	72	n/a	SSTC Anchorages
Small Boat Attack	6	2,100 small-caliber rounds	Hawaii OPAREA
	36	10,500 blank rounds	SSTC Boat Lans 1-10
Offshore Petroleum Discharge System (OPDS)	6	n/a	SSTC Boat Lanes 1-10
Elevated Causeway System (ELCAS)	4	n/a	SSTC Boat Lanes 1-10
Submarine Navigation Exercise	216	n/a	Pearl Harbor
	84	n/a	Subase Pt. Loma
Submarine Under Ice Certification	12	n/a	Hawaii OPAREA
	6	n/a	SOCAL Range Complex
Savage Operations	3	n/a	Hawaii Range Complex
Surface Ship Sonar Maintenance	148	n/a	Hawaii Range Complex
	488	n/a	SOCAL Range Complex
	4	n/a	HSTT Transit Lane
Submarine Sonar Maintenance	132	n/a	Hawaii Range Complex
	68	n/a	SOCAL Range Complex
	4	n/a	HSTT Transit Lane

2.2 Navy Testing Activities

The Navy's research and acquisition community engages in a broad spectrum of testing activities in support of the Navy. 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 U.S. Navy missions and give a technological edge over adversaries.The individual commands within the research and acquisition community are the Naval Air Systems Command, the Naval Sea Systems Command, Space and Naval Warfare Systems Command, and the Office of Naval Research and Naval Research Laboratory. Some testing activities are similar to training exercises conducted by the Fleet. For example, both the Fleet and the research and acquisition community fire 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 to ensure the torpedo meets performance specifications and operational requirements. These differences may result in different analysis and mitigations for the activity. While many of these systems will eventually be used by the Fleet during normal training exercises and are addressed in the HSTT FEIS/OEIS for those Fleet activities, testing and development activities involving the same or similar systems as will be used by operational Fleet units may be used in different locations and manners than when actually used by operational Fleet units. Hence, the analysis for testing events and training of Fleet units may differ.

The Navy’s activities in anti-air warfare, strike warfare, and electronic warfare do not involve stressors that may affect ESA-listed species. Therefore, these activities are not discussed further.

2.2.1 Naval Air Systems Command Testing

Naval Air Systems Command testing activities generally fall in the primary mission areas used by the fleets. Naval Air Systems Command activities include, but are not limited to, the testing of new aircraft platforms, weapons, and systems before those platforms, weapons, and systems are integrated into the fleet. In addition to the testing of new platforms, weapons, and systems, Naval Air Systems Command also conducts lot acceptance testing of weapons and systems, such as sonobuoys (Table 5).

The majority of testing and development activities conducted by Naval Air Systems Command are similar to Pacific Fleet training events, and many platforms (e.g., the MH-60 helicopter) and systems (e.g., Airborne Towed Mine-hunting System (AN/AQS-20A)) currently being tested are already being used by the Fleet or will ultimately be integrated into Fleet training exercises. However, some testing and development may be conducted in different locations and in a different manner than the fleet and, therefore, though the environmental effects may be the same, the analysis for those events may differ.

Table 5. Typical Naval Air Systems Command Testing Activities proposed to occur in the Hawaii-Southern California Training and Testing Study Area.

Activity Name	Activity Description
Anti-Air Warfare (AAW)	
Air Combat Maneuver	This event is identical to the air combat maneuver training event. Test

Activity Name	Activity Description
(ACM) Test	events involve two or more aircraft, each engaged in continuous proactive and reactive changes in aircraft attitude, altitude, and airspeed. No weapons are fired during air combat maneuver test activities.
Air Platform/Vehicle Test	Testing performed to quantify the flying qualities, handling, airworthiness, stability, controllability, and integrity of an air platform or vehicle. No weapons are released during an air platform/vehicle test. In-flight refueling capabilities are tested.
Air Platform Weapons Integration Test	Testing performed to quantify the compatibility of weapons with the aircraft from which they would be launched or released. Mostly non-explosive weapons or shapes are used, but some tests may require the use of high-explosive weapons.
Air-to-Air (A-A) Weapons System Test	Test to evaluate the effectiveness of air-launched weapons against designated airborne targets. Fixed-wing or rotary-wing aircraft may be used. No testing of high-explosive weapons is planned.
Intelligence, Surveillance, and Reconnaissance Test	Test to evaluate communications capabilities of fixed wing and rotary-wing aircraft, including unmanned systems that can carry cameras, sensors, communications equipment, or other payloads. New systems are tested at sea to ensure proper communications between aircraft and ships.
Anti-Surface Warfare (ASUW)	
Air-to-Surface Missile Test	This event is similar to the training event missile exercise (air-to-surface). Test may involve both fixed-wing and rotary-wing aircraft launching missiles at surface maritime targets to evaluate the weapon system or as part of another systems integration test.
Air-to-Surface Gunnery Test	This event is similar to the training event gunnery exercise (air-to-surface). Strike fighter and helicopter aircrews evaluate new or enhanced aircraft guns against surface maritime targets to test that the gun, gun ammunition, or associated systems meet required specifications or to train aircrew in the operation of a new or enhanced weapon system.
Rocket Test	Rocket testing evaluates the integration, accuracy, performance, and safe separation of laser-guided and unguided 2.75-in. rockets fired from a hovering or forward flying helicopter or from a fixed-wing strike aircraft.
Laser Targeting Test	Aircrew use laser targeting devices integrated into aircraft or weapon systems to evaluate targeting accuracy and precision and to train aircrew in the use of newly developed or enhanced laser targeting devices. Lasers are designed to illuminate designated targets for engagement with laser-guided weapons.
Electronic Warfare (EW)	
Electronic Systems Evaluation	Test that evaluates the effectiveness of electronic systems to control, deny, or monitor critical portions of the electromagnetic spectrum. In general, electronic warfare testing will assess the performance of three types of electronic warfare systems: electronic attack, electronic protect, and electronic support.
Anti-Submarine Warfare (ASW)	
Anti-Submarine Warfare Torpedo Test	This event is similar to the training event torpedo exercise. The test evaluates antisubmarine warfare systems onboard rotary-wing and fixed-wing aircraft and the ability to search for, detect, classify, localize, and track a submarine or similar target.
Kilo Dip	A kilo dip is the operational term used to describe a functional check of a helicopter deployed dipping sonar system. The sonar system is briefly activated to ensure all systems are functional. A kilo dip is simply a

Activity Name	Activity Description
	precursor to more comprehensive testing.
Sonobuoy Lot Acceptance Test	Sonobuoys are deployed from surface vessels and aircraft to verify the integrity and performance of a lot, or group, of sonobuoys in advance of delivery to the fleet for operational use.
Anti-Submarine Warfare Tracking Test—Helicopter	This event is similar to the training event anti-submarine warfare tracking exercise/torpedo exercise - helicopter. The test evaluates the sensors and systems used to detect and track submarines and to ensure that helicopter systems used to deploy the tracking systems perform to specifications.
Anti-Submarine Warfare Tracking Test—Maritime Patrol Aircraft	This event is similar to the training event anti-submarine warfare tracking exercise/torpedo exercise -Maritime Patrol Aircraft extended echo ranging sonobuoy. The test evaluates the sensors and systems used by Maritime Patrol Aircraft to detect and track submarines and to ensure that aircraft systems used to deploy the tracking systems perform to specifications and meet operational requirements.
Mine Warfare (MIW)	
Airborne Mine Neutralization Test – AN/ASQ-235 (AMNS)	Airborne mine neutralization tests of the AN/ASQ-235 evaluate the system's ability to detect and destroy mines. The AN/ASQ-235 uses up to four unmanned underwater vehicles equipped with high-frequency sonar, video cameras, and explosive neutralizers.
Airborne Projectile-based Mine Clearance System	A helicopter uses a laser-based detection system to search for mines and to fix mine locations for neutralization with an airborne projectile-based mine clearance system. The system neutralizes mines by firing a small or medium-caliber inert, supercavitating projectile from a hovering helicopter.
Airborne Towed Minesweeping Test – AN/ALQ-220 (OASIS)	Tests of the Organic Airborne and Surface Influence Sweep (OASIS) would be conducted by a helicopter to evaluate the functionality of Organic Airborne and Surface Influence Sweep at sea. The Organic Airborne and Surface Influence Sweep is towed from a forward flying helicopter and works 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 explode.
Airborne Towed Minehunting Sonar Test	Tests of the Airborne Towed Minehunting Sonar System to evaluate the search capabilities of this towed, mine hunting, detection, and classification system. The sonar on the Airborne Towed Minehunting Sonar System identifies mine-like objects in the deeper parts of the water column.
Airborne Laser-Based Mine Detection System Test (ALMDS)	An airborne mine hunting test of the AN/AES-1 Airborne Laser Mine Detection System, or "ALMDS" evaluates the system's ability to detect, classify, and fix the location of floating and near-surface, moored mines. The system uses a laser to locate mines and may operate in conjunction with an airborne projectile-based mine detection system to neutralize mines.
Other Testing Activities	
Test and Evaluation Catapult Launch	Tests evaluate the function of aircraft carrier catapults at sea following enhancements, modifications, or repairs to catapult launch systems. This includes aircraft catapult launch tests. No weapons or other expendable materials would be released.
Air Platform Shipboard Integrate Test	Tests evaluate the compatibility of aircraft and aircraft systems with ships and shipboard systems. Tests involve physical operations and verify and evaluate communications and tactical data links. This test function also

Activity Name	Activity Description
	includes an assessment of carrier-shipboard suitability and hazards of electromagnetic radiation to personnel, ordnance, and fuels.
Shipboard Electronic Systems Evaluation	Tests measure ship antenna radiation patterns and test communication systems with a variety of aircraft.

2.2.2 Proposed Testing Activity Levels / Naval Air Systems Command

Table 6 provides a summary of testing activities including tempo and quantities of inert and live munitions that the U.S. Navy proposes to expend during testing activities.

Table 6. Proposed Naval Air Systems Command Testing Activities in the Hawaii-Southern California Training and Testing (HSTT) Study Area (adapted from Table 2.8-2, Alternative 2, U.S. Navy Final EIS/OEIS, August 2013).

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Anti-Air Warfare (AAW)			
Air Combat Maneuver	11	n/a	HRC
	110	n/a	SOCAL
Air Platform/Vehicle Test	50	n/a	HRC
	385	n/a	SOCAL
Air Platform Weapons Integration Test	44	n/a	HRC
	165	28 missiles, 22,000 small-and medium-caliber rounds, 330 rockets	SOCAL
Intelligence, Surveillance, and Reconnaissance Test	11	n/a	HRC
	50	n/a	SOCAL
Anti-Surface Warfare (ASUW)			
Air-to-Surface Missile Test	10	10 missiles (**5 HE**)	HRC
	100	156 missiles (**48 HE**)	SOCAL
Air-to-Surface Gunnery Test	55	44,000 medium caliber rounds (**11,000 HE**)	SOCAL
Rocket Test	66	748 rockets (**202 HE**)	SOCAL
Laser Targeting Test	6	n/a	SOCAL
Electronic Warfare (EW)			
Electronic Systems Evaluation	670	n/a	SOCAL
Anti-Submarine Warfare (ASW)			
Anti-submarine Warfare Torpedo Test	12	22 torpedoes	HRC
	36	70 torpedoes	SOCAL
Kilo Dip	5	n/a	HRC
	5	n/a	SOCAL
Sonobuoy Lot Acceptance Test	36	**744 HE sonobuoys**	SOCAL
Anti-submarine Warfare Tracking Test – Helicopter	122	**211 HE sonobuoys**	HRC
	188	**1,267 HE sonobuoys**	SOCAL
Anti-submarine Warfare Tracking Test – Maritime Patrol Aircraft	14	**308 HE sonobuoys**	HRC
	33	**1,004 HE sonobuoys**	SOCAL
Mine Warafare (MIW)			

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Airborne Mine Neutralization System Test (AMNS)	17	**53 HE neutralizers**	SOCAL
Airborne Towed Minehunting Sonar System Test	17	n/a	SOCAL
Airborne Towed Minesweeping System Test	17	n/a	SOCAL
Airborne Laser- Based Mine Detection System Test – ALMDS	17	n/a	SOCAL
Airborne Projectile-based Mine Clearance System Test	17	330 medium caliber rounds, **6 HE mines**	SOCAL
Other Testing			
Test and Evaluation – Catapult Launch	9,570	n/a	HSTT
Air Platform Shipboard Integration Test	136	n/a	HSTT
Shipboard Electronic Systems Evaluation	136	n/a	HSTT

2.2.3 Naval Sea Systems Command Testing

Naval Sea Systems Command Testing (NAVSEA) testing activities are aligned with its mission of new ship construction, life cycle support, and other weapon systems development and testing.

New Ship Construction Activities – Ship construction activities include pierside testing of ship systems, tests to determine how the ship performs at sea (sea trials), and developmental and operational test and evaluation programs for new technologies and systems. Pierside and at-sea testing of systems aboard a ship may include sonar, acoustic countermeasures, radars, and radio equipment. During sea trials, each new ship propulsion engine is operated at full power and subjected to high-speed runs and steering tests. At-sea test firing of shipboard weapon systems, including guns, torpedoes, and missiles, are also conducted.

Life Cycle Activities – Testing activities are conducted throughout the life of a Navy ship to verify performance and mission capabilities. Sonar system testing occurs pierside during maintenance, repair, and overhaul availabilities, and at sea immediately following most major overhaul periods. A Combat System Ship Qualification Trial is conducted for new ships and for ships that have undergone modification or overhaul of their combat systems. Radar cross signature testing of surface ships is conducted on new vessels and periodically throughout a ship’s life to measure how detectable the ship is by radar. Electromagnetic measurements of off-board electromagnetic signature are also conducted for submarines, ships, and surface craft periodically.

Other Weapon Systems Development and Testing – Numerous test activities and technical evaluations, in support of NAVSEA’s systems development mission, often occur with Fleet activities within the Study Area. Tests within this category include, but are not limited to, anti-surface, anti-submarine, and mine warfare, using torpedoes, sonobuoys, and mine detection and neutralization systems.

Table 15. Typical Naval Sea Systems Command Testing Activities proposed to occur in the Hawaii-Southern California Training and Testing Study Area.

Activity Name		Activity Description
New Ship Construction		
Surface Combatant Sea Trials	Pierside Sonar Testing	Tests ship's sonar systems pierside to ensure proper operation.
	Propulsion Testing	Ship is run at high speeds in various formations (e.g., straight-line and reciprocal paths).
	Gun Testing	Gun systems are tested using non-explosive rounds.
	Missile Testing	Explosive and non-explosive missiles are fired at target drones to test the launching system.
	Decoy Testing	Includes testing of the MK 36 Decoy Launching system
	Surface Warfare Testing	Ships defend against surface targets with large-caliber guns.
	Anti-Submarine Warfare Testing	Ships demonstrate capability of countermeasure systems and underwater surveillance and communications systems.
Other Ship Class Sea Trials	Propulsion Testing	Ship is run at high speeds in various formations (e.g., straight-line and reciprocal paths). ("Other Ship" indicates class of vessels without hull-mounted sonar. Example ship classes include LCS, MLP, and T-AKE.)
	Gun Testing – Small Caliber	Gun systems are tested using non-explosive rounds.
Mission Package Testing	Anti-Submarine Warfare	Ships and their supporting platforms (e.g., helicopters, unmanned aerial vehicles) detect, localize, and prosecute submarines.
	Surface Warfare	Ships defense against surface targets with small-, medium-, and large-caliber guns and medium range missiles.
	Mine Countermeasures	Ships conduct mine countermeasure operations.
Post-Homeporting Testing (all classes)		Tests all ship systems, including navigation and propulsion systems.
Ship Signature Testing		Tests ship and submarine radars and electromagnetic signatures.
Surface Ship Sonar Testing/ Maintenance (in OPAREAs and Ports)		Pierside and at-sea testing of surface ship systems occurs periodically following major maintenance periods and for routine maintenance.
Submarine Sonar Testing/ Maintenance (in OPAREAs and Ports)		Pierside and at-sea testing of submarine systems occurs periodically following major maintenance periods and for routine maintenance.

Activity Name		**Activity Description**
Combat System Ship Qualification Trial (CSSQT)	In-port Maintenance Period	Each combat system is tested to ensure they are functioning in a technically acceptable manner and are operationally ready to support at-sea Combat System Ship Qualification Trials.
	Air Defense (AD)	Tests the ship’s capability to detect, identify, track, and successfully engage live and simulated targets.
	Surface Warfare (SUW)	Tests shipboard sensors capabilities to detect and track surface targets, relay the data to the gun weapon system, and engage targets.
	Undersea Warfare (USW)	Tests ships ability to track and engage undersea targets.
Anti-Surface Warfare (ASUW)/Anti-Submarine Warfare (ASW) Testing		
Missile Testing		Missile testing includes various missiles fired from submarines and surface combatants.
Kinetic Energy Weapon Testing		A kinetic energy weapon uses stored energy released in a burst to accelerate a non-explosive projectile.
Electronic Warfare Testing		Testing will include radiation of military and commercial radar and communication systems (or simulators).
Torpedo (Non-explosive) Testing		Air, surface, or submarine crews employ non-explosive torpedoes against submarines or surface vessels. All torpedoes are recovered.
Torpedo (Explosive) Testing		Air, surface, or submarine crews employ high-explosive torpedoes against artificial targets or deactivated ships.
Countermeasure Testing		Various acoustic systems (e.g., towed arrays and surface ship torpedo defense systems) are employed to detect, localize, track, and neutralize incoming weapons.
Pierside Sonar Testing		Pierside testing to ensure systems are fully functional in a controlled pierside environment prior to at-sea test activities.
At-sea Sonar Testing		At-sea testing to ensure systems are fully functional in an open ocean environment.
Mine Warfare (MIW) Testing		
Mine Detection and Classification Testing		Air, surface, and subsurface vessels detect and classify mines and mine-like objects.
Pierside Systems Health Checks		Mine warfare systems are tested in pierside locations to ensure acoustic and electromagnetic sensors are fully functional prior to at-sea test activities.
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/diver threats in harbor environments.
Shipboard Protection Systems Testing		Loudhailers and small caliber munitions are used to protect a ship against small boat threats.
Chemical/Biological Simulant Testing		Chemical/biological agent simulants are deployed against surface ships.
Unmanned Vehicle Testing		

Activity Name	Activity Description
Underwater Deployed Unmanned Aerial System Testing	Unmanned aerial systems are launched by submarines and special operations forces while submerged.
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.
Other Testing Activities	
Special Warfare	Special warfare includes testing of submersibles capable of inserting and extracting personnel or payloads into denied areas from strategic distances.
Acoustic Communications Testing	Acoustic modems, submarines, and surface vessels transmit signals to communicate.
Mine Countermeasure/Neutralization Testing	Air, surface, and subsurface vessels neutralize threat mines that would otherwise restrict passage through an area.

2.2.4 Proposed Testing Activity Levels/Naval Sea Systems Command

Table 7 provides a summary of testing activities including tempo and quantities of inert and live munitions that the U.S. Navy proposes to expend during testing activities.

Table 7. Proposed Naval Sea Systems Command Testing Activities in the Hawaii-Southern California Training and Testing Study Area (adapted from Table 2.8-2, Alternative 2, U.S. Navy Final EIS/OEIS, August 2013).

Event		No. of Events (per Year)	Ordnance (Number per Year)	Location
Ship Construction and Maintenance				
New Ship Construction				
Surface Combatant Sea Trials	Pierside Sonar Testing	2	n/a	Pearl Harbor
		2	n/a	San Diego
	Propulsion Testing	2	n/a	HRC
		2	n/a	SOCAL
	Gun Testing	2	52 large-calber rounds, 1,400 medium-caliber rounds	HRC
		2	52 large-calber rounds, 1,400 medium-caliber rounds	SOCAL
	Missile Testing	2	**4 High Explosives (HE) missiles**	HRC
		2	**4 HE missiles**	SOCAL
	Decoy Testing	2	n/a	HRC
		2	n/a	SOCAL
	Surface Warfare Testing	2	96 large-caliber rounds	HRC
		2	96 large-caliber rounds	SOCAL

Event		No. of Events (per Year)	Ordnance (Number per Year)	Location
	Anti-Submarine Warfare Testing	2	n/a	HRC
		2	n/a	SOCAL
Other Class Ship Sea Trials	Propulsion Testing	21	n/a	SOCAL
	Gun Testing - Small Caliber	6	6,000 rounds	SOCAL
ASW Mission Package Testing		40	40 Torpedoes	SOCAL
		16	16 Torpedoes	HRC
Surface Warfare Mission Package Testing	Gun Testing – Small Caliber	5	2,500 rounds	HRC/SOCAL
	Gun Testing - Medium Caliber	5	7,000 rounds (**3,500 HE**)	HRC/SOCAL
	Gun Testing – Large Caliber	5	7,000 rounds (**4,900 HE**)	HRC/SOCAL
	Missile/Rocket Testing	15	30 missilies/rockets (**15 HE**)	HRC/SOCAL
MCM Mission Package Testing		4	n/a	SOCAL / CPAAA
		8	128 neutralizers (**64 HE**)	SOCAL
		4	n/a	SOCAL / Tanner Bank Minefield
		4	128 neutralizers (**64 HE**)	HRC
Life Cycle Activities				
Post Homeporting Testing (All Classes)		22	n/a	HRC
		22	n/a	SOCAL
Ship Signature Testing		3	n/a	HRC
		6	n/a	Pearl Harbor
		39	n/a	SOCAL
Surface Ship Sonar Testing/Maintenance (in OPAREAs and Ports)		17	n/a	HRC
		10	n/a	SOCAL
Submarine Sonar Testing/Maintenance (in OPAREAs and Ports)		18	n/a	HRC
		9	n/a	SOCAL
Combat System Ship Qualification Trial (CSSQT) In Port Maintenance Period		2	n/a	Pearl Harbor
		2	n/a	San Diego
Combat System Ship Qualification Trial (CSSQT) – Air Defense (AD)		6	12,000 medium-caliber rounds, 120 large rounds (**48 HE**)**,** 84 missiles (**42 HE**)	HRC
		2	**2 HE missiles**	SOCAL
Combat System Ship Qualification Trial (CSSQT) – Anti-Surface Warfare (ASUW)		6	12,000 medium-caliber rounds, 1,800 large rounds (**678 HE**)**,** 6 missiles	HRC
		13	14,000 medium-caliber rounds, 3,420 large rounds	SOCAL

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
		(**1,511 HE**), 9 missiles	
Combat System Ship Qualification Trial (CSSQT) – Undersea Warfare (USW)	10	80 torpedoes	HRC
	11	88 torpedoes	SOCAL
Anti-Surface Warfare (ASUW) / Anti-Submarine Warfare (ASW) Testing			
Missile Testing	24	24 missiles	HRC/SOCAL
Kinetic Energy Weapon Testing	55	2,200	HRC
	1 in 5 years	5,000	HRC
Electronic Warfare Testing	106	n/a	Pearl Harbor
	16	n/a	HRC
	54	n/a	SOCAL
Torpedo (Non-Explosive) Testing	9	140 torpedoes	HRC
	10	250 torpedoes	HRC
	2	16 torpedoes	HRC
	17	391 torpedoes	SOCAL
Torpedo (Explosive) Testing	2	28 torpedoes (**8 HE**)	HRC
	2	28 torpedoes (**8 HE**)	SOCAL
Countermeasure Testing	1	n/a	HSTT Transit Corridor
	5	105 torpedoes (**21 HE**)	HRC
	2	84 torpedoes	SOCAL
Pierside Sonar Testing	10	n/a	Pearl Harbor/San Diego
At-Sea Sonar Testing	20	n/a	HRC/SOCAL
Mine Warfare (MIW) Testing			
Mine Detection and Classification Testing	2	n/a	HRC
	3	n/a	HRC: Kahoolawe Training Minefield
	5	n/a	SOCAL
	3	n/a	SOCAL: Mission Bay Training Minefield
Mine Countermeasures / Neutralization Testing	14	**28 HE charges**	SOCAL
	4	n/a	San Diego
Pierside Systems Health Checks	4	n/a	San Diego
Shipboard Protection Systems and Swimmer Defense Testing			
Pierside Integrated Swimmer Defense	5	n/a	San Diego
Shipboard Protection Systems Testing	4	n/a	San Diego
	4	1,300 small caliber rounds	SOCAL
Chemical / Biological Simulant Testing	440	n/a	HRC/SOCAL
Unmanned Vehicle Testing			
Underwater Deployed Unmanned Aerial System Testing	30	n/a	HRC/SOCAL
Unmanned Vehicle Development and Payload Testing	17	n/a	HRC
	26	n/a	SOCAL
Other Testing			
Special Warfare	4	n/a	HRC/SOCAL

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Acoustic Communications Testing	2	n/a	HRC/SOCAL

The Navy proposes to conduct various testing activities in the HSTT Study Area. Detailed information about each proposed activity (stressor, testing event, description, sound source, duration, and geographic location) can be found in Appendix A of the HSTT FEIS/OEIS. NMFS used the detailed information in Appendix A to analyze the impacts from testing activities on ESA-listed species. Table 8 provides the annual number of impulsive source detonations during testing activities within the HSTT Study Area, and Table 9 shows the annual number of hours or items of non-impulsive sources used during testing within the HSTT Study Area.

Table 8. Proposed annual number of impulsive source detonations during testing activities within the Hawaii-Southern California Training and Testing Study Area.

Explosive Class	Net Explosive Weight (NEW)	Annual In-Water Detonations
E1	(0.1 lb. – 0.25 lb.)	14,501
E2	(0.26 lb. – 0.5 lb.)	0
E3	(>0.5 lb. – 2.5 lb.)	2,990
E4	(>2.5 lb.-5 lb.)	753
E5	(>5 lb.-10 lb.)	202
E6	(>10 lb.-20 lb.)	37
E7	(>20 lb.-60 lb.)	21
E8	(>60 lb.-100 lb.)	12
E9	(>100 lb. – 250 lb.)	0
E10	(>250 lb. – 500 lb.)	31
E11	(>500 lb. – 650 lb.)	14
E12	(>650 lb. – 1,000 lb.)	0
E13	(>1,000 lb. – 1,740 lb.)	0

Table 9. Annual hours and items of non-impulsive sources used during testing activities proposed to occur in the Hawaii-Southern California Training and Testing Study Area.

Source Class Category	Source Class	Annual Use
Low-Frequency (LF) Sources that produce signals less than 1 kHz	LF4	52 hours
	LF5	2,160 hours
	LF6	192 hours
Mid-Frequency (MF) Tactical and non-tactical sources that produce signals from 1 to 10 kHz	MF1	180 hours
	MF1K	18 hours
	MF2	84 hours
	MF3	392 hours
	MF4	693 hours

Source Class Category	Source Class	Annual Use
	MF5	5,024 items
	MF6	540 items
	MF8	2 hours
	MF9	3,039 hours
	MF10	35 hours
	MF12	336 hours
High-Frequency (HF) and Very High-Frequency (VHF): Tactical and non-tactical sources that produce signals greater than 10kHz but less than 200kHz	HF1	1,025 hours
	HF3	273 hours
	HF4	1,336 hours
	HF5	1,094 hours
	HF6	3,460 hours
Anti-Submarine Warfare (ASW) Tactical sources used during anti-submarine warfare	ASW1	224 hours
	ASW2	2,260 items
	ASW2	255 hours
	ASW3	1,278 hours
	ASW4	477 items
Torpedoes (TORP) Source classes associated with active acoustic signals produced by torpedoes	TORP1	701 items
	TORP2	732 items
Acoustic Modems (M) Transmit data acoustically through the water	M3	4,995 hours
Swimmer Detection Sonar (SD) Used to detect divers and submerged swimmers	SD1	38 hours
Airguns (AG) Used during swimmer defense and diver deterrent	AG	5 uses
Synthetic Aperture Sonar (SAS): Sonar in which active acoustic signals are post-processed to form high-resolution images of the seafloor	SAS1	2,700 hours
	SAS2	4,956 hours
	SAS3	3,360 hours

2.2.5 Space and Naval Warfare Systems Command Testing

The mission of Space and Naval Warfare Systems Command Testing (SPAWAR) is to acquire, develop, deliver, and sustain decision superiority for the warfighter at the right time and for the right cost. SPAWAR Systems Center Pacific is the research and development part of SPAWAR focused on developing and transitioning technologies in the area of command, control, communications, computers, intelligence, surveillance, and reconnaissance. SPAWAR Systems Center Pacific conducts research, development, test, and evaluation projects to support emerging technologies for intelligence, surveillance, and reconnaissance; anti-terrorism and force protection; mine countermeasures; anti-submarine warfare; oceanographic research; remote sensing; and communications. These activities include, but are not limited to, the testing of unmanned undersea and surface vehicles, a wide variety of intelligence, surveillance, and reconnaissance sensor systems, underwater surveillance technologies, and underwater communications (Table 10).

Table 10. Typical Space and Naval Warfare Systems Command Testing Activities in the Hawaii-Southern California Training and Testing Study Area.

Activity Name	Activity Description
Autonomous Undersea Vehicle (AUV) Anti-Terrorism/Force Protection (AT/FP) Mine Countermeasures	Autonomous undersea vehicle shallow water mine countermeasure testing is focused on the testing of unmanned undersea vehicles with mine hunting sensors in marine environments in and around rocky outcroppings. Anti-terrorism/force protection mine countermeasures testing is focused on mine countermeasure missions in confined areas between piers and pilings
Autonomous Undersea Vehicle (AUV) Underwater Communications	This testing is focused on providing two-way networked communications below the ocean surface while maintaining mission profile.
Fixed System Underwater Communications	Fixed underwater communications systems testing is focused on testing stationary or free floating equipment that provides two-way networked communications below the ocean surface while maintaining mission profile
Autonomous Undersea Vehicle (AUV) Autonomous Oceanographic Research and Meteorology and Oceanography (METOC)	The research is comprised of ocean gliders and autonomous undersea vehicles. Gliders are portable, long-endurance buoyancy driven vehicles that provide a means to sample and characterize ocean water properties. Autonomous undersea vehicles are larger, shorter endurance vehicles.
Fixed Autonomous Oceanographic Research and Meteorology and Oceanography (METOC)	The goal of these systems is to develop, integrate, and demonstrate deployable autonomous undersea technologies that improve the Navy’s capability to conduct effective anti-submarine warfare and intelligence, surveillance, and reconnaissance operations in littoral waters
Passive Mobile Intelligence, Surveillance, and Reconnaissance Sensor Systems	These systems use passive arrays hosted by surface and subsurface vehicles and vessels for conducting submarine detection and tracking experiments and demonstrations
Fixed Intelligence, Surveillance, and Reconnaissance Sensor Systems	These systems use stationary fixed arrays for conducting submarine detection and tracking experiments and demonstrations.
Anti-Terrorism/Force Protection (AT/FP) Fixed Sensor Systems	These systems use stationary fixed arrays for providing protection of Navy assets from underwater threats.

2.2.6 Proposed Testing Activity Levels/Space and Naval Warfare Systems Command

Table 11 provides a summary of testing activities including tempo and location where the U.S. Navy proposes to conduct during testing.

Table 11. Proposed Space and Naval Warfare Systems Command Testing Activities proposed to occur in the Hawaii-Southern California Training and Testing Study Area (adapted from Table 2.8-4, Alternative 2, U.S. Navy Final EIS/OEIS, August 2013).

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Autonomous Undersea Vehicle (AUV)	92	n/a	SOCAL

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Anti-Terrorism/Force Protection (AT/FP) Mine Countermeasures	20	n/a	HRC
AUV Underwater Communications	92	n/a	SOCAL
	20	n/a	HRC
Fixed System Underwater Communications	37	n/a	SOCAL
AUV Autonomous Oceanographic Research and Meteorology and Oceanography (METOC)	92	n/a	SOCAL
	20	n/a	HRC
Fixed Autonomous Oceanographic Research and METOC	26	n/a	SOCAL
Passive Mobile Intelligence, Surveillance, and Reconnaissance Sensor Systems	27	n/a	SOCAL
Fixed Intelligence, Surveillance, and Reconnaissance Sensor Systems	39	n/a	SOCAL
	4	n/a	HRC
Anti-Terrorism/Force Protection (AT/FP) Fixed Sensor Systems	11	n/a	SOCAL

2.2.7 Office of Naval Research Testing

As the Navy’s science and technology provider, Office of Naval Research and Naval Research Laboratory Testing (ONR and NRL) provide technology solutions for Navy and Marine Corps needs. ONR’s mission is to plan, foster, and encourage scientific research in recognition of its paramount importance as related to the maintenance of future naval power, and the preservation of national security. Further, ONR manages the Navy’s basic, applied, and advanced research to foster transition from science and technology to higher levels of research, development, test, and evaluation. The Ocean Battlespace Sensing Department explores science and technology in the areas of oceanographic and meteorological observations, modeling, and prediction in the battlespace environment; submarine detection and classification (anti-submarine warfare); and mine warfare applications for detecting and neutralizing mines in both the ocean and littoral environment. ONR events include research, development, test, and evaluation activities; surface processes acoustic communications experiments; shallow water acoustic communications experiments; sediment acoustics experiments; shallow water acoustic propagation experiments; and long range acoustic propagation experiments Table 12.

Table 12. Typical Office of Naval Research Testing Activity in the Hawaii-Southern California Training and Testing Study Area.

Activity Name	Activity Description

Activity Name	Activity Description
Kauai Acoustic Communications Experiment (Coastal)	The primary purpose of the Kauai Acoustic Communications Experiment is to collect acoustic and environmental data appropriate for studying the coupling of oceanography, acoustics, and underwater communications.

2.2.8 Proposed Testing Activity Levels/ Office of Naval Research

Table 13 provides a summary of testing activities including tempo and location where the U.S. Navy proposes to conduct during testing.

Table 13. Proposed Office of Naval Reasearch Testing Activities in the Hawaii-Southern California Training and Testing Study Area (adapted from Table 2.8-5, Alternative 2, U.S. Navy Final EIS/OEIS, August 2013).

Event	No. of Events (per Year)	Ordnance (Number per Year)	Location
Kauai Acoustic Communications Experiment	2	n/a	HRC

2.2.9 Sonar, Ordnance, Targets, and Other Systems

The Navy uses a variety of sensors, platforms, weapons, and other devices to meet its mission. Training and testing with these systems may introduce acoustic (sound) energy into the environment. This section describes and organizes sonar systems, ordnance, munitions, targets, and other systems to facilitate understanding of the activities in which these systems are used. Underwater sound is described as one of two types for the purposes of the Navy's proposed action: impulsive and non-impulsive. Underwater detonations of explosives and other percussive events are impulsive sounds. Sonar and similar sound producing systems are categorized as non-impulsive sound sources.

Sonar and Other Non-impulsive Sources – Modern sonar technology includes a variety of sonar sensor and processing systems. The simplest active sonar emits sound waves, or "pings," sent out in multiple directions and the sound waves then reflect off of the target object in multiple directions. The sonar source calculates the time it takes for the reflected sound waves to return; this calculation determines the distance to the target object. More sophisticated active sonar systems emit a ping and then rapidly scan or listen to the sound waves in a specific area. This provides both distance to the target and directional information. Even more advanced sonar systems use multiple receivers to listen to echoes from several directions simultaneously and provide efficient detection of both direction and distance. The Navy rarely uses active sonar continuously throughout activities. When sonar is in use, the pings occur at intervals, referred to as a duty cycle, and the signals themselves are very short in duration. For example, sonar that emits a 1-second ping every 10 seconds has a 10-percent duty cycle. The Navy utilizes sonar systems and other acoustic sensors in support of a variety of mission requirements. Primary uses include the detection of and defense against submarines (anti-submarine warfare) and mines (mine warfare); safe navigation and effective communications; use of unmanned undersea vehicles; and oceanographic surveys.

Ordnance and Munitions – Most ordnance and munitions used during training exercises and testing events fall into three basic categories: projectiles (such as gun rounds), missiles (including rockets), and bombs. Ordnance can be further defined by their net explosive weight, which considers the type and quantity of the explosive substance without the packaging, casings, bullets, etc. Net explosive weight (NEW) is the trinitrotoluene (TNT) equivalent of energetic material, which is the standard measure of strength of bombs and other explosives. For example, a 12.7-centimeter (cm) shell fired from a Navy gun is analyzed at about 9.5 pounds (lb) (4.3 kilograms (kg)) of NEW. The Navy also uses non-explosive ordnance in place of high explosive ordnance in many training exercises and testing events. Non-explosive ordnance and munitions look and perform similarly to high explosive ordnance, but lack the main explosive charge.

Defensive Countermeasures – Naval forces depend on effective defensive countermeasures to protect themselves against missile and torpedo attack. Defensive countermeasures are devices designed to confuse, distract, and confound precision guided munitions. Defensive countermeasures analyzed in this biological opinion application include acoustic countermeasures, which are used by surface ships and submarines to defend against torpedo attack. Acoustic countermeasures are either released from ships and submarines, or towed at a distance behind the ship.

Mine Warfare Systems – The Navy divides mine warfare systems into two categories: mine detection and mine neutralization. Mine detection systems are used to locate, classify, and map suspected mines, on the surface, in the water column, or on the sea floor. The Navy analyzed the following mine detection systems for impacts to marine mammals:

- Towed or hull-mounted mine detection systems. These detection systems use acoustic and laser or video sensors to locate and classify suspect mines. Fixed and rotary wing platforms, ships, and unmanned vehicles are used for towed systems, which can rapidly assess large areas.

- Unmanned/remotely operated vehicles. These vehicles use acoustic and video or lasers to locate and classify mines and provide unique capabilities in nearshore littoral areas, surf zones, ports, and channels.

- Marine mammal systems. The Navy deploys trained Atlantic bottlenose dolphins (*Tursiops truncatus*) and California sea lions (*Zalophus californianus*) for integrated training involving two primary mission areas: to find objects such as inert mine shapes, and to detect swimmers or other intruders around Navy facilities such as piers. These systems also include one or more motorized small boats and several crew members for each trained marine mammal. When not engaged in training, Navy marine mammals are housed in temporary enclosures either on land or aboard ships.

Mine Neutralization Systems – Mine neutralization systems disrupt, disable, or detonate mines to clear ports and shipping lanes, as well as littoral, surf, and beach areas in support of naval amphibious operations. The Navy analyzed the following mine neutralization systems for impacts to ESA-listed species:

- Towed influence mine sweep systems. These systems use towed equipment that mimic a particular ship's magnetic and acoustic signature triggering the mine and causing it to explode.
- Unmanned/remotely operated mine neutralization systems. Surface ships and helicopters operate these systems, which place explosive charges near or directly against mines to destroy the mine.
- Airborne projectile-based mine clearance systems. These systems neutralize mines by firing a small or medium-caliber non-explosive, supercavitating projectile from a hovering helicopter.
- Diver emplaced explosive charges. Operating from small craft, divers put explosive charges near or on mines to destroy the mine or disrupt its ability to function.

2.2.9.1 *Classification of Non-impulsive and Impulsive Sources Analyzed*

In order to better organize and facilitate the analysis of about 300 sources of underwater non-impulsive sound or impulsive energy, the Navy developed a series of source classifications, or source bins. This method of analysis provides the following benefits:

- Allows for new sources to be covered under existing authorizations, as long as those sources fall within the parameters of a "bin;"
- Simplifies the data collection and reporting requirements anticipated under the MMPA;
- Ensures a conservative approach to all impact analysis because all sources in a single bin are modeled as the loudest source (e.g., lowest frequency, highest source level, longest duty cycle, or largest net explosive weight within that bin);
- Allows analysis to be conducted more efficiently, without compromising the results;
- Provides a framework to support the reallocation of source usage (hours/explosives) between different source bins, as long as the total number and severity of marine mammal and sea turtle takes remain within the overall analyzed and authorized limits.

> This flexibility is required to support evolving Navy training exercises and testing requirements, which are linked to real world events.

Non-impulsive sources are grouped into bins based on the frequency, source level when warranted, and how the source would be used. Impulsive bins are based on the net explosive weight of the munitions or explosive devices. The following factors further describe how non-impulsive sources are divided:

- Frequency of the non-impulsive source:
    - Low-frequency sources operate below 1 kilohertz (kHz)
    - Mid-frequency sources operate at or above 1 kHz, up to and including 10 kHz
    - High-frequency sources operate above 10 kHz, up to and including 100 kHz
    - Very high-frequency sources operate above 100, but below 200 kHz
- Source level of the non-impulsive source:
    - Greater than 160 decibels (dB), but less than 180 dB
    - Equal to 180 dB and up to 200 dB
    - Greater than 200 dB

How a sensor is used determines how the sensor’s acoustic emissions are analyzed. Factors to consider include pulse length (time source is on); beam pattern (whether sound is emitted as a narrow, focused beam, or, as with most explosives, in all directions); and duty cycle (how often a transmission occurs in a given time period during an event).

There are also non-impulsive sources with characteristics that are not anticipated to result in takes of marine mammals and sea turtles. These sources have low source levels, narrow beam widths, downward directed transmission, short pulse lengths, frequencies beyond known hearing ranges of marine mammals, or some combination of these factors. These sources were not modeled by the Navy.

2.3 NMFS’ Promulgation of Regulations Pursuant to the Marine Mammal Protection Act (MMPA)

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 HSTT Action 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 determined 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 and interactions with vessels. 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 regulations are all consistent with the Navy's action described previously in this Opinion so they will not be reiterated here.

The take of ESA-listed species by harassment incidental to the Navy's training exercises in the HSTT Action Area authorized pursuant to NMFS Permit Division's MMPA rule is presented in Table 14.

Table 14. Level B harassment take of listed species authorized pursuant to the Marine Mammal Protection Act by NMFS Permits Division for training exercises within the HSTT Action Area.

Species	Annual takes	5-year total takes
Blue Whale	up to 4,325	21,559
Fin Whale	up to 1,719	8,531
Western North Pacific Gray Whale	up to 10	50
Humpback Whale	up to 9,273	46,365
Sei Whale	up to 630	3,150
Main Hawaiian Islands Insular False Killer Whale	up to 49	245
Sperm Whale	up to 3,332	15,920
Guadalupe Fur Seal	average of 2,603	13,015
Hawaiian Monk Seal	Average of 1,292	6,334

The Final Rule also authorized lethal take of up to 12 ESA-listed large whales by vessel strike during training activities. Since issuance of the Final Rule, NMFS and the Navy determined for the reasons discussed in more detail in section 6.6.2 below that strike of up to 12 ESA-listed large whales would not be reasonably expected to occur over the five-year period and that this estimate was an overestimation of the strike risk to these species. Additionally, this estimation did not apportion the risk by species, and therefore potentially overestimated risk for some species while conceivably underestimating risk for others. Consequently, for this Opinion, we have performed a new assessment of strike risk in section 6.6.2 below.

The take of ESA-listed species by harassment incidental to the Navy's testing exercises in the HSTT Action Area authorized pursuant to NMFS Permit Division's MMPA rule is presented in Table 10.

Table 15. Level B harassment take of listed species authorized pursuant to the Marine Mammal Protection Act by NMFS' Permits Division for testing activities within the HSTT Action Area.

Species	Annual takes	5-year total takes
Blue whale	up to 426	2,140
Fin whale	up to 225	1,125
Western North Pacific Gray Whale	up to 2	10
Humpback whale	up to 927	4,635
Sei whale	up to 51	255
Main Hawaiian Islands Insular False killer whale	up to 4	20
Sperm whale	up to 263	1,315
Guadalupe fur seal	average of 269	1,345
Hawaiian monk seal	average of 358	1,790

The Final Rule also authorized lethal take of one ESA-listed large whale per year by vessel strike during testing activites. Since issuance of the Final Rule, NMFS and the Navy determined that strike of up to one ESA-listed large whale per year would not be reasonably expected to occur. Additionally, this estimation did not apportion the risk by species and therefore potentially overestimatedg risk for some species while conceivably underestimating risk for others. Consequently, for this Opinion, we have performed a new assessment of strike risk in section 6.6.2 below.

NMFS' Permits Division also set forth mitigation, monitoring, and reporting requirements in the MMPA rule. Here we provide a summary of those that relate to the avoidance or minimization of impacts to marine mammals found in the Final Rule. Items in the MMPA regulations that are related to the LOA application process or specific requirements that must be included in each LOA issued under the rule, renewal process, and modification process of LOAs are not reiterated here.

2.3.1 Mitigation Measures Employed by the U.S. Navy

NMFS Permits Division, NMFS ESA Interagency Cooperation Division, and the Navy have reached agreement regarding mitigation measures that will be implemented by the Navy. These measures satisfy the requirements of the MMPA and many were carried over as requirements from previous section 7 consultations. The Navy will implement measures that will allow their training exercises to have the least practicable adverse impact on marine mammal species or stocks (which includes considerations of personnel safety, practicality of implementation, and impact on the effectiveness of the "military readiness activity"). Those measures are summarized in this section of this Opinion; for a complete description of all of the measures applicable to the proposed exercises, readers should refer to the U.S. Navy's request for a letter of authorization and the Permit Division's regulations and LOAs. Many of these measures would minimize the impacts to other ESA-listed species. Mitigation measures in the final rule are divided into four categories: (1) lookouts, (2) mitigation zones, (3) humpback whale cautionary area, and (4) stranding response.

2.3.1.1 ***Lookouts***

The use of lookouts is a critical component of Navy procedural measures and implementation of mitigation zones. Navy lookouts are highly qualified and experienced observers of the marine environment. They are required to report all objects sighted in the water to the Officer of the Deck (OOD) (e.g., trash, a periscope, marine mammals, sea turtles) 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.

The Navy would have two types of lookouts for purposes of conducting visual observations: (1) those positioned on surface ships, and (2) those positioned in aircraft or on boats. Lookouts positioned on surface ships would be dedicated solely to diligent observation of the air and surface of the water. They would have multiple observation objectives, which 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. Due to aircraft and boat manning and space restrictions, lookouts positioned in aircraft or on boats would consist of the aircraft crew, pilot, or boat crew.

Lookouts positioned in aircraft and boats may necessarily be responsible for tasks in addition to observing the air or surface of the water (for example, navigation of a helicopter or rigid hull inflatable boat). However, aircraft and boat lookouts would, to the maximum extent practicable and consistent with aircraft and boat safety and training and testing requirements, comply with the observation objectives described above for lookouts positioned on surface ships.

All divers placing the charges on mines will support the Lookouts while performing their regular duties. The divers will report all marine mammal, scalloped hammerhead shark, and sea turtle sightings to their dive support vessel.

2.3.1.2 ***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 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) When marine mammals are visually detected, the Navy shall ensure that low-frequency and hull-mounted mid-frequency active sonar transmission levels are limited to at least 6 dB below normal operating levels, for sources that can be powered down, if any detected marine mammals are within 1,000 yd (914 m) of the sonar dome (the bow).[1]

(B) The Navy shall ensure that low-frequency and hull-mounted mid-frequency active sonar transmissions are limited to at least 10 dB below the equipment's normal operating level, for sources that can be powered down, if any detected marine mammals are within 500 yd (457 m) of the sonar dome.

(C) The Navy shall ensure that low-frequency sonar and hull-mounted mid-frequency active sonar transmissions are ceased, for sources that can be turned off during the activity, if any visually detected marine mammals are within 200 yd (183 m) of the sonar dome. Transmissions will not resume until one of the following conditions is met: the animal is observed exiting the mitigation zone; the animal is thought to have exited the mitigation zone based on a determination of its course and speed and the relative motion between the animal and the source; the mitigation zone has been clear from any additional sightings for a period of 30 minutes; the ship has transited more than 2,000 yd (1.8 km) beyond the location of the last sighting; or the ship concludes that dolphins are deliberately closing in on the ship to ride the ship's bow wave (and there are no other marine mammal sightings within the mitigation zone). Active transmission may resume when dolphins are bow riding because they are out of the main transmission axis of the active sonar while in the shallow-wave area of the bow.

(D) The Navy shall ensure that low-frequency and hull-mounted mid-frequency active sonar transmissions are ceased for sources that cannot be powered down during the activity, if any visually detected marine mammals are within 200 yd (183 m) of the source. Transmissions will not resume until one of the following conditions is met: the animal is observed exiting the mitigation zone; the animal is thought to have exited the mitigation zone based on a determination of its course and speed and the relative motion between the animal and the source; the mitigation zone has been clear from any additional sightings for a period of 30 minutes; or the ship has transited more than 400 yd (366 m) beyond the location of the last sighting.

(E) When marine mammals are visually detected, the Navy shall ensure that high-frequency and non-hull-mounted mid-frequency active sonar transmission levels are ceased if any visually detected marine mammals are within 200 yd (183 m) of the source. Transmissions will not resume until one of the following conditions is met: the animals is observed exiting the mitigation zone; the animal is thought to have exited the mitigation zone based on a determination of its course and speed and the relative motion between the animal and the source; 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

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[1] This measure does not apply to SURTASS LFA Sonar which is addressed in a separate MMPA rule and biological opinion.

period of 30 minutes for a vessel-deployed source; the vessel or aircraft has repositioned itself more than 400 yd (366 m) away from the location of the last sighting; or the vessel concludes that dolphins are deliberately closing to ride the vessel's bow wave (and there are no other marine mammal sightings within the mitigation zone).

(iv) Mitigation zones for explosive and impulsive sound:

(A) A mitigation zone with a radius of 600 yd (549 m) shall be established for IEER sonobuoys (bin E4).

(B) A mitigation zone with a radius of 350 yd (320 m) shall be established for explosive sonobuoys using 0.6 to 2.5 lb net explosive weight (bin E3).

(C) A mitigation zone with a radius of 200 yd (183 m) shall be established for anti-swimmer grenades (bin E2).

(D) A mitigation zone ranging from 600 yd (549 m) to 2,100 yd (1.9 km), dependent on charge size, shall be established for general mine countermeasure and neutralization activities using positive control firing devices. Mitigation zone distances are specified for charge size in Table 11-2 of the Navy's application.

(E) A mitigation zone ranging from 350 yd (320 m) to 850 yd (777 m), dependent on charge size, shall be established for mine countermeasure and neutralization activities using diver-placed positive control firing devices. Mitigation zone distances are specified for charge size in Table 11-2 of the Navy's application.

(F) A mitigation zone with a radius of 1,000 yd (914 m) shall be established for mine neutralization diver placed mines using time-delay firing devices (bin E7).

(G) 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 (bin E2).

(H) A mitigation zone with a radius of 600 yd (549 m) shall be established for large-caliber gunnery exercises with a surface target (bin E5).

(I) A mitigation zone with a radius of 900 yd (823 m) shall be established for missile exercises (including rockets) with up to 250 lb net explosive weight and a surface target (bin E9).

(J) A mitigation zone with a radius of 2,000 yd (1.8 km) shall be established for missile exercises with 251 to 500 lb net explosive weight and a surface target (E10)

(K) A mitigation zone with a radius of 2,500 yd (2.3 km) shall be established for bombing exercises (bin E12).

(L) A mitigation zone with a radius of 2,100 yd (1.9 km) shall be established for torpedo (explosive) testing (bin E11).

(M) A mitigation zone with a radius of 2.5 nautical miles shall be established for sinking exercises (bin E12).

(N) A mitigation zone with a radius of 1,600 yd (1.4 km) shall be established for at-sea explosive testing (bin E5).

(O) A mitigation zone with a radius of 60 yd (55 m) shall be established for elevated causeway system pile driving.

(P) A mitigation zone with a radius of 70 yd (64 m) within 30 degrees on either side of the gun target line on the firing side of the vessel for explosive and non-explosive large-caliber gunnery exercises.

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

(A) A mitigation zone of 500 yd (457 m) for observed whales and 200 yd (183 m) for all other marine mammals (except bow riding dolphins) shall be established for all vessel movement, providing it is safe to do so.

(B) A mitigation zone of 250 yd (229 m) for any observed marine mammal shall be established for all towed in-water devices that are towed from a manned platform, 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 with non-explosive practice munitions.

(B) A mitigation zone of 1,000 yd (914 m) shall be established for bombing exercises with non-explosive practice munitions.

(C) A mitigation zone of 900 yd (823 m) shall be established for missile exercises (including rockets) using a surface target.

(vii) Mitigation zones for the use of Navy sea lions:

(A) If a monk seal is seen approaching or within 100 m of a Navy sea lion, the handler will hold the Navy sea lion in the boat or recall the Navy sea lion immediately if it has already been released.

2.3.1.3 *Humpback Whale Cautionary Area*

The Navy will maintain a 5-km (3.1-mi) buffer zone in the Study Area between December 15th and April 15th where conducting mid-frequency active sonar exercises will require authorization by the Commander, U.S. Pacific Fleet (CPF). If authorized, the CPF will provide specific direction on required mitigation prior to operational units transiting to and training in the area. The Navy will provide NMFS with advance notification of any mid-frequency active sonar training and testing activities in the humpback whale cautionary area between December 15th and April 15th.

2.3.1.4 *Stranding Response*

The proposed MMPA regulations include a stranding response plan as follows (78 FR 6978).

(4) Stranding Response Plan

> (i) The Navy shall abide by the letter of the “Stranding Response Plan for Major Navy Training Exercises in the HSTT Study Area,” to include the following measures:
>
> > (A) Shutdown Procedures - When an Uncommon Stranding Event (USE - defined in § 218.71 (b)(1)) occurs during a Major Training Exercise (MTE) in the HSTT Study Area, the Navy shall implement the procedures described below.

(1) The Navy shall implement a shutdown (as defined § 218.71 (b)(2)) when advised by a NMFS Office of Protected Resources Headquarters Senior Official designated in the HSTT Study Area Stranding Communication Protocol that a USE involving live animals has been identified and that at least one live animal is located in the water. NMFS and the Navy will maintain a dialogue, as needed, regarding the identification of the USE and the potential need to implement shutdown procedures.

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

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

(4) In the event, following a USE, that qualified individuals are attempting to herd animals back out to the open ocean and animals are not willing to leave, or animals are seen repeatedly heading for the open ocean but turning back to shore, NMFS and the Navy shall

coordinate (including an investigation of other potential anthropogenic stressors in the area) to determine if the proximity of mid-frequency active sonar training activities or explosive detonations, though farther than 14 nautical miles from the distressed animal(s), is likely contributing to the animals' refusal to return to the open water. If so, NMFS and the Navy will further coordinate to determine what measures are necessary to improve the probability that the animals will return to open water and implement those measures as appropriate.

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

2.3.2 **Monitoring and Reporting Requirements**

Monitoring and reporting is required by the MMPA regulations as described in the final MMPA rule. These requirements are generally consistent with monitoring and reporting required of, and undertaken by, the Navy in the past for the SOCAL and Hawaii Range Complexes under prior biological opinions, rules, and permits. These data have helped to inform subsequent analyses, informing assumptions we have made in the past and better informing our new analyses. As such, we continue to require these components with occassional modification to reflect revised assumptions and analyses.

§ 218.75 Requirements for monitoring and reporting.

(a) As outlined in the HSTT Study Area Stranding Communication Plan, the Holder of the Authorization must notify NMFS immediately (or as soon as operational security considerations allow) if the specified activity identified in § 218.70 is thought to have resulted in the mortality or injury of any marine mammals, or in any take of marine mammals not identified in § 218.71.

(b) The Holder of the LOA must conduct all monitoring and required reporting under the LOA, including abiding by the HSTT Monitoring Plan.

(c) General Notification of Injured or Dead Marine Mammals - Navy personnel shall ensure that NMFS (regional stranding coordinator) is notified immediately (or as soon as operational security considerations allow) if an injured or dead marine mammal is found during or shortly after, and in the vicinity of, an Navy training or testing activity utilizing mid- or high-frequency active sonar, or underwater explosive detonations. The Navy shall provide NMFS with species or description of the animal(s), the condition of the animal(s) (including carcass condition if the animal is dead), location, time of first discovery, observed behaviors (if alive), and photo or video (if available). The Navy shall consult the Stranding Response Plan to obtain more specific reporting requirements for specific circumstances.

(d) Vessel Strike – In the event that a Navy vessel strikes a whale, the Navy shall do the following:

(1) Immediately report to NMFS (pursuant to the established Communication Protocol) the:

(i) Species identification if known;

(ii) Location (latitude/longitude) of the animal (or location of the strike if the animal has disappeared);

(iii) Whether the animal is alive or dead (or unknown); and

(iv) The time of the strike.

(2) As soon as feasible, the Navy shall report to or provide to NMFS, the:

(i) Size, length, and description (critical if species is not known) of animal;

(ii) An estimate of the injury status (e.g., dead, injured but alive, injured and moving, blood or tissue observed in the water, status unknown, disappeared, etc.);

(iii) Description of the behavior of the whale during event, immediately after the strike, and following the strike (until the report is made or the animal is no long sighted);

(iv) Vessel class/type and operation status;

(v) Vessel length

(vi) Vessel speed and heading; and

(vii) To the best extent possible, obtain a photo or video of the struck animal, if the animal is still in view.

(3) Within 2 weeks of the strike, provide NMFS:

(i) A detailed description of the specific actions of the vessel in the 30-minute timeframe immediately preceding the strike, during the event, and immediately after the strike (e.g., the speed and changes in speed, the direction and changes in the direction, other maneuvers, sonar use, etc., if not classified); and

(ii) A narrative description of marine mammal sightings during the event and immediately after, and any information as to sightings prior to the strike, if available; and

(iii) Use established Navy shipboard procedures to make a camera available to attempt to capture photographs following a ship strike.

(e) Annual HSTT Monitoring Plan Report - (1) The Navy shall submit an annual report for the HSTT Monitoring Plan in April of each year, describing the implementation and results 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 HSTT Monitoring Plan shall, at a minimum, provide the same marine mammal observation data required in § 218.75. (2) As an alternative, the Navy may submit a multi-Range Complex annual Monitoring Plan report to fulfill this requirement. Such a report would describe progress of knowledge made with respect to monitoring plan study questions across all Navy ranges associated with the ICMP. 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.

(f) Annual HSTT Exercise and Testing Reports - The Navy shall submit preliminary reports detailing the status of authorized sound sources within 21 days after the end of the annual authorization cycle. The Navy shall submit detailed reports 3 months after the anniversary of the date of issuance of the LOA. The detailed annual reports shall contain information on Major Training Exercises (MTE), Sinking Exercise (SINKEX) events, and a summary of sound sources used, as described below. 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 detailed reports shall contain information identified in paragraphs (e)(1) through (e)(5) of this section.

(1) Major Training Exercises/SINKEX:

(i) This section shall contain the reporting requirements for Coordinated and Strike Group exercises and SINKEX. Coordinated and Strike Group Major Training Exercises include:

(A) Joint Task Force Exercises (JTFEX) / Sustainment Exercise (SUSTAINEX).

(B) Integrated ASW Course (IAC).

(C) Composite Training Unit Exercises (COMPTUEX).

(D) Undersea Warfare Exercise (USWEX).

(E) Rim of the Pacific (RIMPAC)

(ii) Exercise information for each MTE:

(A) Exercise designator.

(B) Date that exercise began and ended.

(C) Location (operating area).

(D) Number of items or hours (per the LOA) of each sound source bin (impulsive and non-impulsive) used in the exercise.

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

(F) Individual marine mammal sighting info for each sighting for each MTE:

(1) Date/time/location of sighting.

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

(3) Number of individuals.

(4) Initial detection sensor.

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

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

(7) Sea state.

(8) Visibility.

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

(10) Indication of whether animal is <200 yd, 200-500 yd, 500-1,000 yd, 1,000-2,000 yd, or >2,000 yd from sound source.

(11) Mitigation implementation – whether operation of sonar sensor was delayed, or sonar was powered or shut down, and how long the delay was; or whether navigation was changed or delayed.

(12) If source in use is a hull-mounted sonar, relative bearing of animal from ship and estimation of anima's motion relative to ship (opening, closing, parallel).

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

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

(iii) Exercise information for each SINKEX:

(A) List of the vessels and aircraft involved in the SINKEX.

(B) Location (operating area).

(C) Chronological list of events with times, including time of sunrise and sunset, start and stop time of all marine species surveys that occur before, during, and after the SINKEX, and ordnance used.

(D) Visibility and/or weather conditions, wind speed, cloud cover, etc. throughout exercise if it changes.

(E) Aircraft used in the surveys, flight altitude, and flight speed and the area covered by each of the surveys, given in coordinates, map, or square miles.

(F) Passive acoustic monitoring details (number of sonobuoys, area and depth that was heard, detections of biologic activity, etc.).

(G) Individual marine mammal sighting info for each sighting that required mitigation to be implemented:

(1) Date/time/location of sighting.

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

(3) Number of individuals.

(4) Initial detection sensor.

(5) Indication of specific type of platform the observation was made from (including, for example what type of surface vessel or platform).

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

(7) Sea state.

(8) Visibility.

(9) Indication of whether animal is <200 yd, 200-500 yd, 500-1,000 yd, 1,000-2,000 yd, or >2,000 yd from the target.

(10) Mitigation implementation – whether the SINKEX was stopped or delayed and length of delay.

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

(H) List of the ordnance used throughout the SINKEX and net explosive weight (NEW) of each weapon and the combined ordnance NEW.

(2) Summary of Sources Used.

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

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

(B) Total annual expended/detonated rounds (missiles, bombs, etc.) for each explosive bin;

(C) Total annual airgun use; and

(D) Improved Extended Echo-Ranging System (IEER)/sonobuoy summary, including:

(1) Total expended/detonated rounds (buoys).

(2) Total number of self-scuttled IEER rounds.

(3) Sonar Exercise Notification – The Navy shall submit to NMFS (specific contact information to be provided in LOA) either an electronic (preferably) or verbal report within fifteen calendar days after the completion of any major exercise (RIMPAC, USWEX, or Multi Strike Group) indicating:

(i) Location of the exercise.

(ii) Beginning and end dates of the exercise.

(iii) Type of exercise (e.g., RIMPAC, USWEX, or Multi Strike Group).

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

(5) Special Reporting Requirements – To the extent practicable, and as it applies to the specific Study Area, these reports will also include:

(i) The total hours (from December 15th through April 15th) of hull-mounted active sonar operation occurring in the dense humpback areas generally shown on the Mobley map (73 FR 35510, 35520) plus a 5-km buffer, but not including the Pacific Missile Range Facility (as illustrated in the HSTT FEIS/OEIS).

(ii) The total estimated annual hours of hull-mounted active sonar operation conducted in the Humpback Whale Cautionary Area between December 15th through April 15th.

(6) 5-year Close-out Exercise and Testing Report – This report will be included as part of the 2019 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 FEIS 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 NMFS' Issuance of Letters of Authorization (LOAs) Pursuant to the MMPA Regulations

NMFS's Permits Division issued regulations under the MMPA to govern the unintentional taking of marine mammals incidental to training and testing activities conducted in the HSTT Action Area from December 2013 through December 2018. These regulations allow NMFS to issue

Letters of Authorization (LOAs) for the incidental take of marine mammals during the Navy's specified activities and time frames, set forth the permissible methods of taking, set forth other means of minimizing adverse impact on marine mammal species or stocks and their habitat, and set forth requirements pertaining to the monitoring and reporting of the incidental take. The final LOA for HSTT training activities can be found at: http://www.nmfs.noaa.gov/pr/pdfs/permits/hstt_loa_training.pdf, and the LOA for testing can be found at: http://www.nmfs.noaa.gov/pr/pdfs/permits/hstt_loa_testing.pdf.

The levels of take associated with acoustic stressors to ESA-listed marine mammals assessed in this Opinion are identical to those assessed in the MMPA rule. However, take associated with mortality of ESA-listed whale species resulting from vessel strike is lower in this Opinion than in the MMPA rule. Our regulations (50 C.F.R. 402.02) indicate that the jeopardy analysis should be based upon the effects of the action that reasonably would be expected. Our ESA Consultation Handbook likewise indicates that the incidental take evaluated in the biological opinion should be the level of take that is anticipated and reasonably likely to result from the action. Consistent with the regulations and Handbook, we have taken a new approach in this Opinion and analyzed vessel strikes of ESA-listed whales that reasonably would be expected from the full scope of the Navy'sactivities based on historical Navy strike data and relative whale abundances (see section 6.6.2). In contrast, the number of whale strikes published in the MMPA rule represents an improbable worst-case scenario that substantially exceeds actual strike rates and is not reasonably expected to occur.

The ESA requires that take of listed marine mammals specified in an incidental take statement be authorized under the MMPA[2]. Because take of marine mammals from acoustic stressors assessed in this Opinion is identical to that contained in the MMPA rule and LOAs, and take from vessel strikes assessed in this Opinion is equal to or below that authorized in the MMPA rule and LOAs, all take of marine mammals specified in this Opinion and incidental take statement has also been authorized under the MMPA. However, any take from vessel strikes in excess of the levels specified in this Opinion and incidental take statement will require reinitiation of consultation, regardless of whether the take is authorized by the MMPA rule and LOAs.

2.5 Interrelated and Interdependent Activities

Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration. NMFS determined that there are no interrelated actions outside the scope of this consultation. The three interdependent activities were described above.

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[2] *In the litigation challenging the MMPA rule and LOAs, the court entered an order on March 31, 2015, finding that NMFS's negligible impact and least practicable adverse impact determinations were arbitrary and capricious. Under the court's scheduling order, appropriate relief is to be addressed in future proceedings. In the event the MMPA rule and LOAs are vacated, the provisions of the Incidental Take Statement pertaining to listed marine mammals would become operative only upon issuance of new MMPA take authorization or if Navy testing and training activities in the HSTT Action Area are exempted from the requirements of the MMPA pursuant to 16 USC 1371(f).*

2.6 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 CFR 402.02). The Action Area for this Opinion includes the HSTT Study Area and areas outside the study area that are within the range to effects from stressors generated within the Study Area boundary. For example, sound from impulsive and non-impulsive acoustic sources may travel outside of the Study Area boundary.

The HSTT Study Area (Figure 1) is comprised of established operating and warning areas across the north-central Pacific Ocean, from Southern California to Hawaii and the International Date Line. The defined Study Area has expanded beyond the areas included in previous Navy authorizations to include transit routes and pierside locations. The Navy proposes to conduct military readiness training exercises and testing activities throughout the in-water areas around the Hawaiian Islands and off the coast of Southern California, primarily in established operating and military warning areas of the Study Area.

The SOCAL Range Complex (Figure 2) is situated between Dana Point and San Diego, and extends more than 600 nm southwest into the Pacific Ocean, encompassing 120,000 nm^2 of sea space, 113,000 nm^2 of special use airspace, and over 56 square miles (mi^2) of land area.

The Hawaii Range Complex (

Figure 3) consists of ocean areas located around the major islands of the Hawaiian Islands chain. The offshore areas form an area approximately 1,700 nm by 1,600 nm. The component areas of the Hawaii Range Complex include the Hawaii operating area which consists of 235,000 nm^2 of surface and subsurface ocean areas and special use airspace as well as various Navy land ranges and other services' land used for military training exercises and testing activities.

The Silver Strand Training Complex (Figure 4) is composed of oceanside beach and boat training lanes, ocean anchorage areas, bayside water training areas in the San Diego Bay and its bayside beaches.

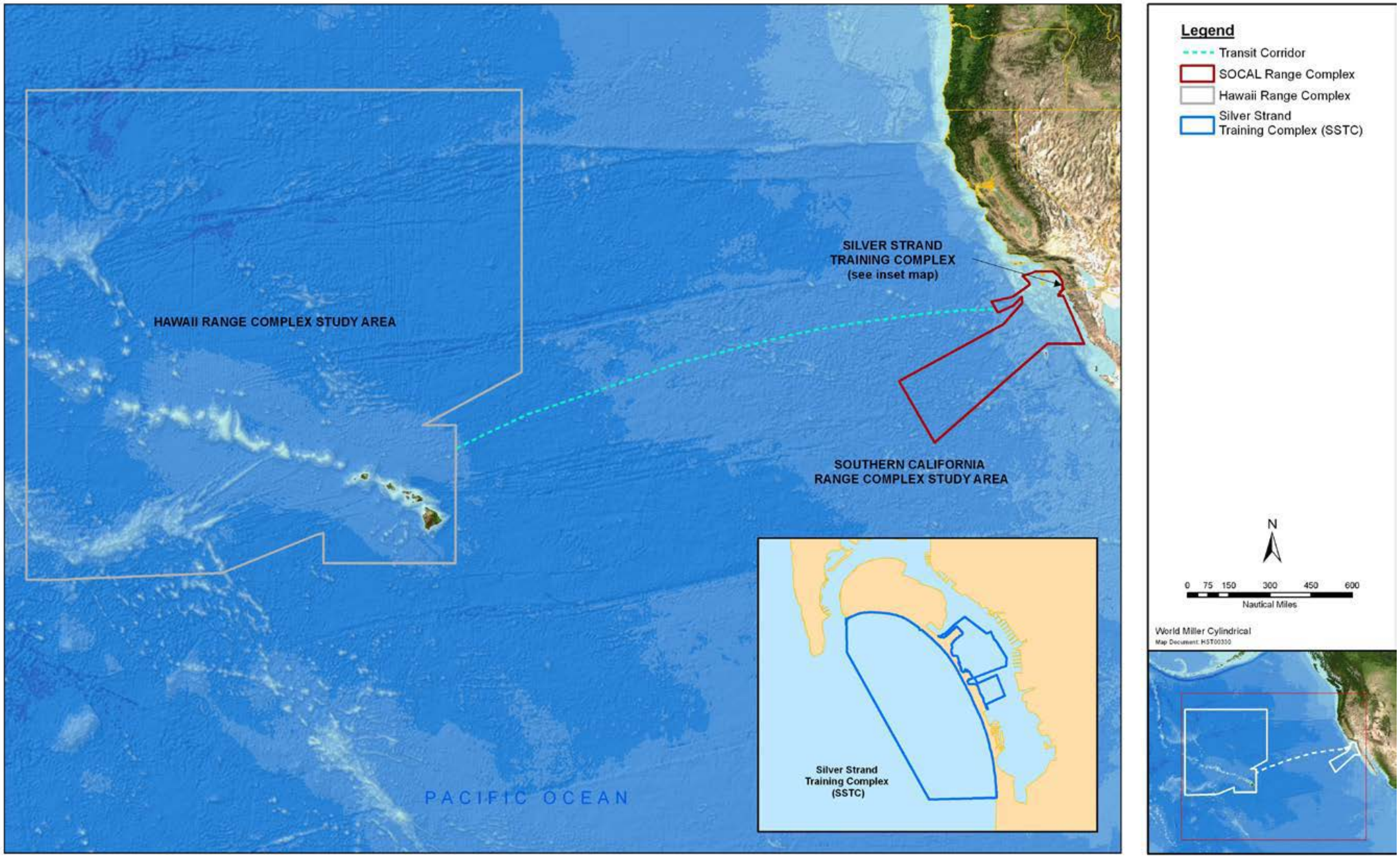

Figure 1. Action Area - U.S. Navy Hawaii – SOCALTraining and Testing Study Area.

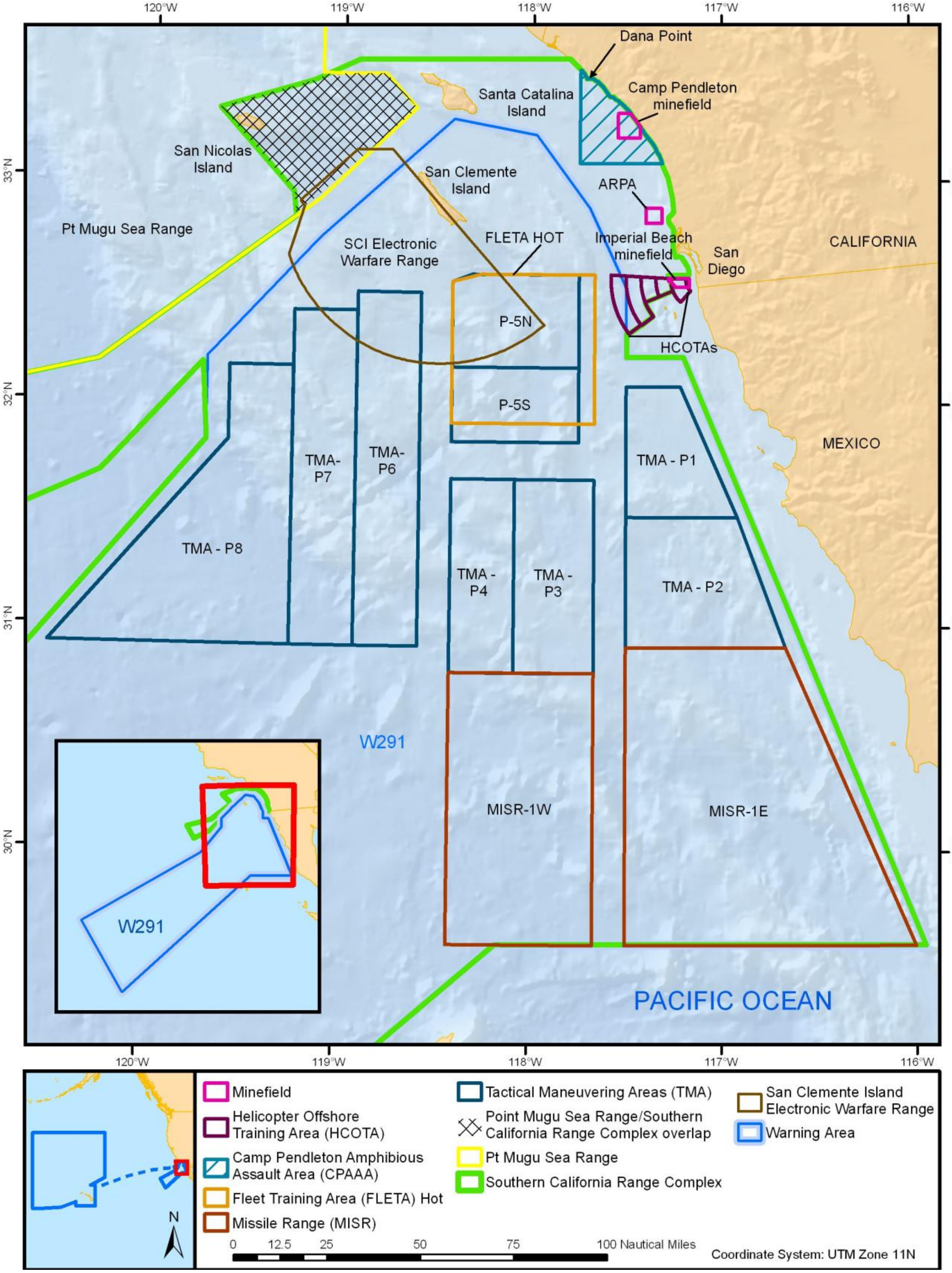

Figure 2. Action Area – Southern California Range Complex

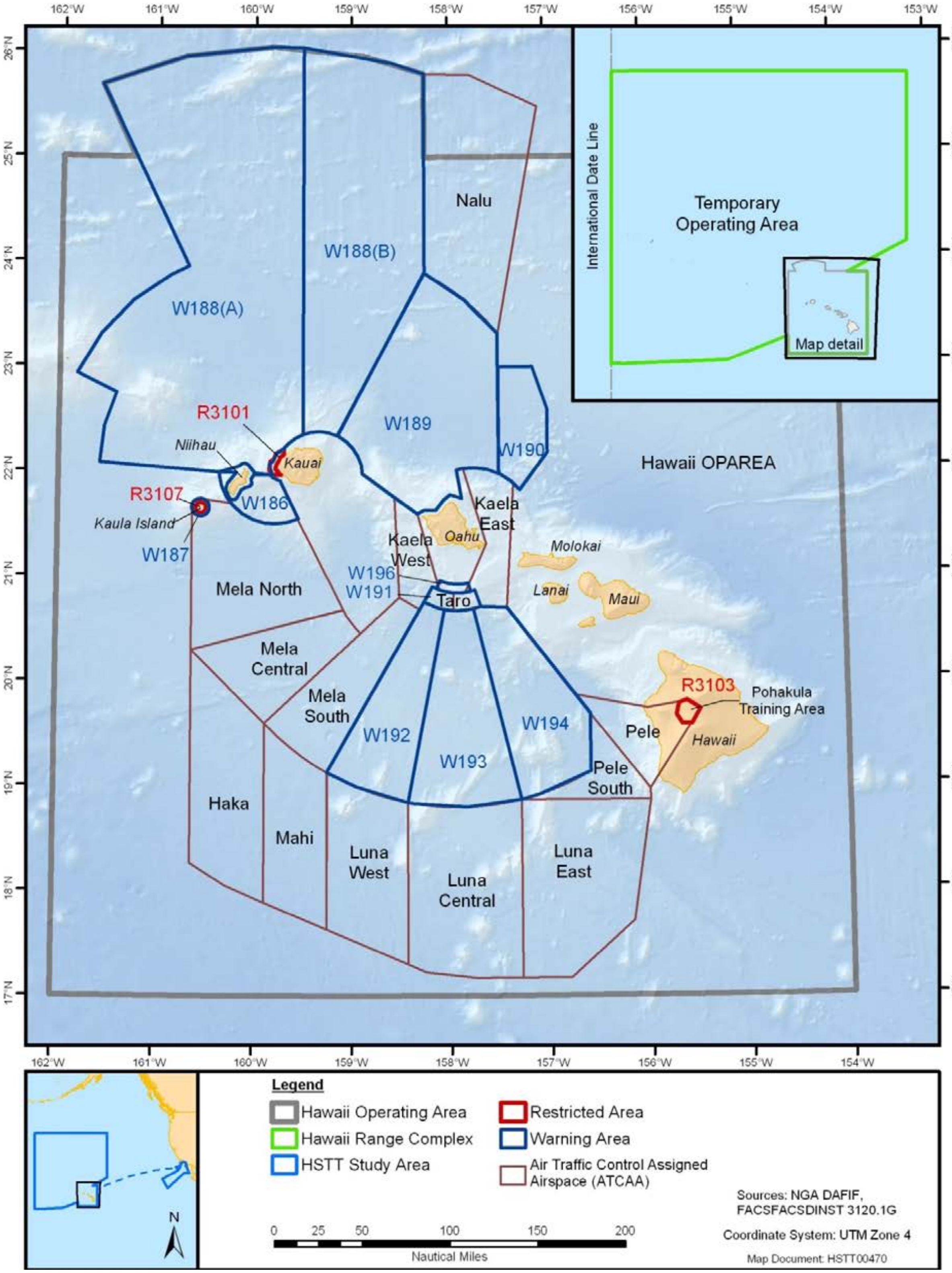

Figure 3. Action Area – Hawaii Range Complex

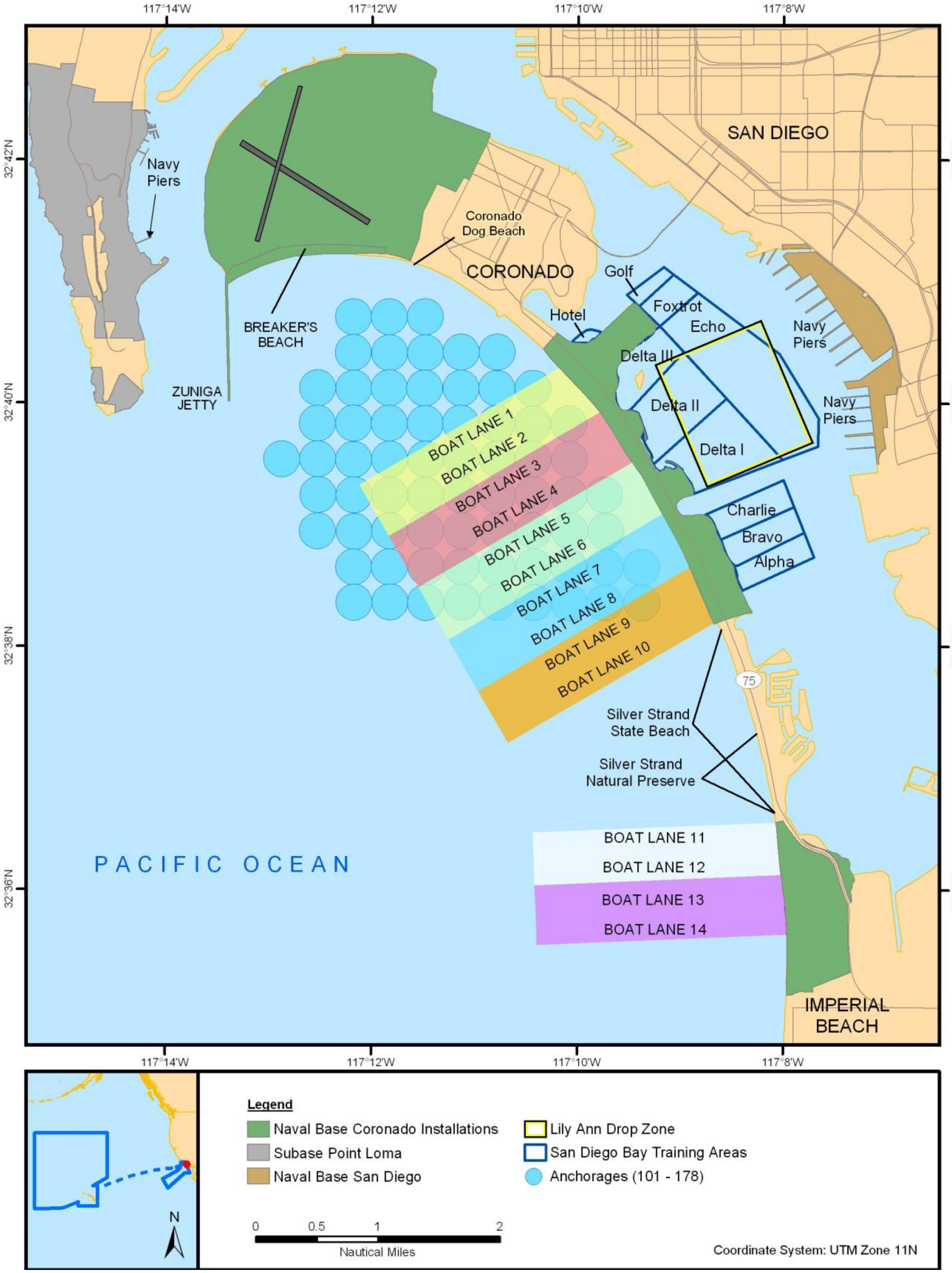

Figure 4. Action Area – Silver Strand Training Complex

3 APPROACH TO THE ASSESSMENT

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 a 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 a 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.

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 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.[3]

3.1 Overview of NMFS' Assessment Framework

We will use the following approach to determine whether the proposed action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Identify the range-wide status of the species and critical habitat likely to be adversely affected by the proposed action; then
- 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 (Figure 1, Figure 2,
    - Figure 3,and Figure 4);
    - 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; then
- Analyze the effects of the proposed action on both species and their habitat; then

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[3] 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).

- Consider how the proposed action would affect the species' reproduction, numbers, and distribution or, in the case of salmon and steelhead, their viable salmonid population (VSP) parameters; then
- Evaluate the proposed action's effects on critical habitat features; then
- Describe any cumulative effects 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;

We integrate and synthesize the above factors to assess the risk that the proposed action poses to species and critical habitat.

- We assess the effects of the action with the impacts to the resources resulting from the *Environmental Baseline* and the *Cumulative Effects* to assess whether the action could reasonably be expected to:
    - Reduce appreciably the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution; and/or
    - Reduce the conservation value of designated or proposed critical habitat. These assessments are made in full consideration of the *Status of Listed Resources* and critical habitat.

Reach jeopardy and adverse modification *Conclusion*. In this step we state our conclusions regarding jeopardy and the destruction or adverse modification of critical habitat (*Integration and Synthesis*).

If necessary, we define a reasonable and prudent alternative to the proposed action. If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of 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 listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

3.2 Defining "Significance"

In biological opinions, we focus on potential physical, chemical, or biotic stressors that are "significant" in the sense of being distinct from ambient or background. We then ask 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) (Johnson 1999).

For populations (or sub-populations), we are concerned 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 designated area.

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[4] largest number of individuals of a particular species that can survive over long periods of time in a given enviroment, this level depends on the effect of the limiting factors

3.3 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 U.S. Navy provided NMFS with a draft and final EIS/OEIS on training and testing that are proposed in the Action Area. We also evaluated the Navy's annual and comprehensive major training exercise and monitoring reports required by the MMPA rule and LOAs and the previous biological opinion to assess effectiveness of mitigation and actual take incidental to actual 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.

3.3.1 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 draft Acoustic Guidance that provides acoustic threshold levels for onset of permanent threshold shift (PTS) and temporary threshold shifts (TTS) in marine mammals for all sound sources. NOAA has since been working to incorporate the relevant information received during the public comment period and to make appropriate changes.

In January 2015, while NOAA was still working to finalize the Guidance, the U.S. Navy provided NOAA with a technical paper by Finneran (2015) describing Navy's proposed methodology for updating auditory weighting functions and numeric thresholds for predicting onset of auditory effects (TTS/PTS thresholds) on marine animals exposed to active sonars and other active acoustic sources utilized during Navy training and testing activities. One of the most significant differences between the criteria employed in prior Navy modeling and those proposed in Finneran (2015) is that the low-frequency cetacean and sea turtle TTS and PTS thresholds are higher at all frequencies. This change is intended to account for the relatively poor auditory sensitivities of sea turtles and the suspected elevated thresholds in low-frequency cetaceans relative to mid-frequency cetaceans.

NOAA is working to evaluate and incorporate the information in Finneran (2015) into its Acoustic Guidance before it becomes final. Before doing so, an independent peer review (i.e., second peer review of the Guidance) of the Navy's technical paper will be necessary, along with a subsequent additional public comment period (i.e., second public comment period for the Guidance). After the peer review and public comment processes are complete, NOAA will determine how best to incorporate the Navy's methodology into its final Acoustic Guidance. The Guidance likely will not be finalized until later this year. Thereafter, any new Navy modeling based on our final Acoustic Guidance would likely take a minimum of several months to complete. Consequently, the results of prior Navy modeling described in this Opinion represent the best available estimate of the number and type of take that may result from the Navy's use of acoustic sources in the HSTT Action Area. 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 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 HSTT 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 ten 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. 2012b; Kastelein et al. 2014a; Kastelein et al. 2014b; Popov et al. 2013; Popov et al. 2011)) (Table 16).

3.3.1.1 ***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. (2013, 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. 2000; i.e., these studies are not expected to result in any changes to the thresholds).

3.3.1.2 *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. 2012b), tones (Kastelein et al. 2013b; Kastelein et al. 2014b), or 1 to 2 kHz sonar (Kastelein et al. 2014a). 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 HSTT 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, 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.

3.3.1.3 *Phocid Pinnipeds*

Two more recent studies for phocid pinnipeds (harbor seal) were completed by Kastelein et al. (2012a; 2013a). 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. (2012a) 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 $\mathrm{SEL}_{\mathrm{cum}}$; note that this exposure level is higher than what is being proposed in the Guidance for PTS onset, which is 197 dB $\mathrm{SEL}_{\mathrm{cum}}$, and that the exposure in Kastelein did result in recovery).

Table 16. 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. 2012a	Octave-band noise (non-impulsive)	Harbor seal (2)
Kastelein et al. 2012b	Octave-band noise (non-impulsive)	Harbor porpoise (1)
Finneran and Schlundt 2013	Tones (non-impulsive)	Bottlenose dolphin (2)
Popov et al. 2013	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. 2014a	1-2 kHz sonar (non-impulsive)	Harbor porpoise (1)
Kastelein et al. 2014b	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 1 March 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.

3.3.2 The U.S. Navy's Exposure Estimates

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

3.3.2.1 *The U.S. Navy Acoustic Effects Model (NAEMO)*

Since 1997, the U.S. 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, 23 August 2013 (updated from 12 March 2012).

3.3.2.1.1 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 U.S. Navy's analyses in the HSTT 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 12 March 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 (2012b) 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 sumsthe 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 HSTT 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.

3.3.2.1.2 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 exercises 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.

3.3.3 Consideration of Potential Biological Removal Estimates in ESA Risk Analyses

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with the Services, to insure that "any action authorized, funded, or carried out by such agency... is not likely to jeopardize the continued existence" of any listed species (16 U.S.C. 1536(a)(2)). NMFS uses the best available science to analyze the effects of federal actions and to determine if those actions are likely to jeopardize the survival and recovery of ESA-listed entities. As part of its review of the best available science for this consultation, NMFS considered the use of a concept developed pursuant to the Marine Mammal Protection Act (MMPA) to assess the effects of incidental take on marine mammals in commercial fishing operations, known as potential biological removal (PBR) level, for its jeopardy analysis. For the reasons discussed below, NMFS concluded that it will not further consider PBR in this consultation.

3.3.3.1 *Assessment of the Background and Use of PBR*

ESA-listed marine mammals receive protections under the ESA and the MMPA. The MMPA aims to prevent species and "stocks" of marine mammals from falling below their optimum sustainable population (OSP). OSP is defined by the MMPA as "the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element" (16 U.S.C. 1362(9)).

In 1988, Congress amended the MMPA, with provisions for the incidental take of marine mammals in commercial fishing operations. Congress directed NMFS to develop and recommend a new long-term regime to govern such incidental taking (see MMC, 1994). The need to set allowable take levels incidental to commercial fishing operations led NMFS to suggest a new and simpler conceptual means for assuring that incidental take authorized under the MMPA does not cause any marine mammal species or stock to be reduced or maintained below the lower limit of its OSP level. That concept, known as PBR, was incorporated in the 1994 amendments to the MMPA, wherein Congress enacted MMPA sections 117 and 118, establishing a new regime governing the incidental taking of marine mammals in commercial fishing operations and stock assessments. PBR is defined by the MMPA as "the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population" (16 U.S.C. 1362(20)). For purposes of implementing sections 117 and 118 of the

MMPA, PBR is one tool that can be used to help evaluate the effects of mortality and serious injury[5] (M/SI) on a marine mammal stock.[6]

PBR appears within the MMPA only in section 117 (relating to periodic stock assessments) and in portions of section 118 describing requirements for take reduction plans for reducing marine mammal bycatch in commercial fisheries. It does not appear in the ESA. Since the introduction of PBR, NMFS has considered the concept almost entirely within the context of implementing sections 117 and 118 and other commercial fisheries management-related provisions of the MMPA, including those within section 101(a)(5)(E) related to the taking of ESA-listed marine mammals incidental to commercial fisheries. Nothing in the MMPA or ESA requires the application of PBR outside the management under the MMPA of commercial fisheries interactions with marine mammals.

Moreover, nothing in the ESA requires NMFS to make a jeopardy determination on the basis of whether a species is at or below its optimum sustainable population, which is the population level upon which PBR is based. When assessing whether an action is likely to jeopardize the continued existence of an ESA-listed species, NMFS examines whether the “action... reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and the recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species” (50 C.F.R. 402.02). Survival is the condition in which an ESA-listed entity continues to exist into the future while retaining the potential for recovery (USFWS and NMFS 1998). Recovery is the improvement of status of the listed entity such that it no longer requires protection under the ESA (USFWS and NMFS 1998), and criteria for recovery are often outlined in a species’ recovery plan. Thus, a marine mammal stock’s OSP may be significantly different than a population size below which a species may be at risk of jeopardy. Neither the ESA nor NMFS’ regulations implementing the statute define jeopardy, survival, or recovery in relation to a species’ optimum sustainable population. Therefore, although PBR may be useful in certain instances for evaluating the impact of human-caused mortality on marine mammal stocks, it is not a measure for jeopardy and should not be used as a determinative indicator of jeopardy.

3.3.3.2 *Comparison of MMPA Stocks and ESA-Listed Entities*

The utility of PBR in this consultation is further limited by the fact that PBR is intended to assess, and calculated for marine mammal “stocks.” The MMPA defines a “stock” as “a group of marine mammals of the same species or smaller taxa in a common spatial arrangement that interbreed when mature” (16 U.S.C. 1362(11)). For the purposes of management under the MMPA, a stock is a management unit that identifies a demographically independent biological

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[5] For purposes of implementing sections 117 and 118 of the MMPA, NMFS defines serious injury as an injury that is more likely than not to result in mortality.

[6] PBR is not designed to assess the impacts of indirect or sublethal effects, such as behavioral response, injury from acoustic stressors, masking, avoidance behaviors, or energetic effects because these are non-lethal impacts.

population. MMPA stocks typically are only defined and assessed for marine mammals within U.S. jurisdictional waters. Marine mammal populations falling outside of U.S. jurisdictional waters, containing only a small portion of their range in U.S. jurisdictional waters, or that occur only in U.S. jurisdictional waters during rare climatic events usually are not assessed as MMPA stocks (NMFS 2005). NMFS' stock assessment reports define the stock and its geographic range, and provide a PBR for the stock.

Under the ESA, a "species" is the listed entity upon which further ESA activities, such as recovery actions, consultations, and permitting decisions, are assessed. The ESA defines "species" to include "any subspecies of fish or wildlife or plants, and any distinct population segment (i.e., DPS) of any species of vertebrate fish or wildlife which interbreeds when mature" (16 U.S.C. 1532(16)). In 1996, the Services published a policy to clarify their interpretation of DPS (61 FR 4722, Feb 7, 1996). The policy outlined two key elements to determine if a population qualified for listing as a DPS: discreteness of the population segment in relation to the remainder of the species to which it belongs, and significance of the population segment to the species to which it belongs. Neither the ESA, nor the DPS policy, explicitly separate ESA-listed entities by U.S. jurisdictional water boundaries. In contrast to MMPA stocks, listed entities for marine mammals under the ESA are often rangewide (i.e., circumglobal) or are broken into DPSs based on criteria established under the ESA. Many ESA-listed species also have a comprehensive recovery plan with clear goals designed to recover the listed entity so it no longer requires protection under the ESA. Species-specific recovery units, as defined in ESA recovery plans, may be based on the ESA-listed entity distribution, MMPA stock distributions, or based on entirely different criteria.

Although MMPA stocks and ESA-listed entities may overlap, they are not identical by definition and typically are not treated identically in practice because of the different conservation and recovery standards required by the two statutes. For example, the fin whale is listed as endangered under the ESA throughout its circumglobal range. By contrast, under the MMPA, fin whales have been divided into four stocks in U.S. jurisdictional waters: Hawaii, California/Oregon/Washington, Alaska (Northeast Pacific), and the western North Atlantic stock. In a recent five year review, NMFS acknowledged MMPA stocks and the ESA-listed entity for fin whales are incongruous (NMFS 2011).

Given that MMPA stocks may constitute only a portion of an ESA-listed entity, as illustrated by the fin whale example, the PBR calculated for an MMPA stock may not provide information useful for assessing the status of the full ESA-listed entity, or the effect of an action on the full ESA-listed entity. Such is the case here. All of the ESA-listed entities that could experience mortality as a result of the proposed action differ significantly from the MMPA stocks that have been defined for the species in the Action Area. Therefore, the PBRs for MMPA stocks that exist in the Action Area do not provide information meaningful for assessing the effect of the action on the ESA-listed entities that we are evaluating for this consultation.

3.3.3.3 *Conclusion*

Section 7 of the ESA requires NMFS to determine whether an action is likely to jeopardize the continued survival and recovery of a listed species. As discussed above, PBR is tied to an MMPA concept (i.e., optimum sustainable population) that is not directly comparable to this standard and is calculated for MMPA management units that typically are not equivalent to ESA-listed entities. NMFS will not further consider PBR in this consultation because the federal action being evaluated does not occur entirely within the geographic distribution of a single MMPA stock and none of the listed entities that could experience mortality as a result of vessel strike are identical to MMPA stocks for which PBR has been calculated.

3.3.4 Applicability of Population Viability Analyses in this Opinion

Population viability analysis (PVA) is a species-specific method of risk assessment frequently used in conservation biology. It is traditionally defined as the process that determines the probability a population will go extinct within a given number of years.

A PVA can be used to estimate the likelihood of a population’s extinction, indicate the urgency of recovery efforts, and identify key life stages or processes that should be the focus of recovery efforts. It is also used to compare proposed management options and assess existing recovery efforts. A PVA can be used in endangered species management to develop a plan of action, rank the pros and cons of different management scenarios, and assess the potential impacts of habitat loss. Each PVA is individually developed for a target population or species, and consequently, each PVA is unique. For the reasons discussed below, the use of PVA, such as those conducted by Van Houtan (2011) for shallow-set fisheries, is not an appropriate tool for analyzing the effects of Navy training and testing in the HSTT Action Area.

The PVA developed by Van Houtan (2011) was facilitated by the collection of 7 years of mortality data and estimates of fishing effort specific to the local fisheries upon which to estimate the take of loggerhead and leatherback sea turtles associated with the proposed action. Estimating take from HSTT activities is more complicated, particularly for sea turtle ship strike where at-sea and near-shore mortality data from large vessel strikes is virtually non-existent and is difficult to attribute to Navy vessels. Although there are limited situations for which mortality can be observed and attributed to HSTT activities (e.g., marine mammal vessel strikes), we determined that a PVA is not the most appropriate or efficient method of determining the effects of such take on the ESA-listed entity (e.g. distinct population segment or entire rangewide species) for the purposes of determining whether the proposed HSTT activities are likely to jeopardize the continued existence or survival of the entity.

The two PVA models created by Van Houtan (2011) were a classical simple model and a climate-based model. These two models were created for loggerhead and leatherback sea turtles to analyze the effects of the Hawaii shallow-set longline fishery on these species. The classical models assumed that demographic forces are the primary factors affecting population dynamics and that the factors remained constant through time; however, as noted by the author, these these two assumptions may not be warranted. The classical model consisted of two variables: initial population size and annual rate of population growth. Each of these variables were estimated from nearby population nesting data that presumably is the source of sea turtles within the fishery Action Area. The climate-based model (Van Houtan 2011) included several climate variables in addition to the age of first breeding and an assumed adult survival rate of 90 percent. The quasi-extinction threshold (QET; the number of individuals at which the population will go extinct) was arbitrarily set at 50 percent of the initial population size.

More robust (and ultimately more accurate and reliable) PVA models include variables and parameters such as initial population size, age-specific birth rates, age-specific mortality rates, immigration rates, emigration rates, carrying capacity of the population, factors affecting birth/death rates including population density and other stochastic factors, and genetic variation. Most of these variables will not be known for a vast majority of threatened or endangered populations and will need to be assumed, predicted, or ignored. Variables not empirically derived would need separate validation. The QET for robust PVA modeling is estimated using demographic, survival, and genetic data so that a more precise estimate of the minimum viable population size is used in PVA analyses. If these factors have been experimentally assessed or can be reliably estimated there is an additional confounding factor for using PVA to provide insight into a consultation’s jeopardy analysis. The Hawaii shallow-set longline fishery consultation used a PVA based entirely on local population estimates, which provides risk associated with estimated take to that population only and not necessarily the ESA-listed entity, which may be a DPS or the worldwide population of the species. In these instances it might be necessary to perform two PVAs, one at the local level to determine extinction probability at that level, and another PVA at the DPS or global scale to determine the effects the extinction of that population would have on the ESA-listed entity. Considering the uncertainty involved with variables and parameters at the local level, there would be even more uncertainty at these extremely large scales.

In cases where there is not enough data to support a robust PVA, such as for the species considered in this Opinion, results of a PVA can be misinterpreted, misleading, contradictory, or completely wrong. Rather than perform an ill-advised PVA, which may provide a mathematical (and likely wrong) estimation of risk associated the HSTT activities, it is more appropriate to use simple analyses and qualitatively describe uncertainties surrounding the estimation of mortality and the effects it may have on the species. The two PVA models created for the Hawaii shallow-set longline fishery had completely contradictory results within and between the two turtle species analyzed. The classical model predicted a declining population trend for the leatherback sea turtle while the climate-based model predicted a stable or increasing population trend. The classical model predicted an increasing population trend for the loggerhead sea turtle while the climate-based model predicted a decreasing population trend. Van Houtan (2011) concluded the climate-based model provides a more robust analysis over one turtle generation but that both models are less reliable beyond that period. A single generation is not adequate to predict the risk of extinction. The author expressed that using the classical model to make decisions carries significant risk due to the limitations of the data and analytical approach, especially over extended periods of time more conducive to extinction risk analyses. As such, even in the case of the Hawaii shallow-set longline fishery consultation where mortality data was readily available, there is no basis for concluding that PVA analysis would have been a superior means of rendering a jeopardy or non-jeopardy opinion than a qualitative analysis of available data regarding local or species-wide extinction risk.

In sum, PVA analyses are not useful for the HSTT consultation due to a lack of data and the numerous estimations and assumptions required to perform the analyses. PVAs may be useful for consultations on smaller, well-defined, and well-studied populations or species. The most purposeful use of PVAs may be in determining minimum viable population estimates and risk extinction analyses for conservation and recovery planning, although data limitations would still need to be overcome.

3.4 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 their 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; as a result, 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 phenomema 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.

Cummulative 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.

4 STATUS OF LISTED RESOURCES

This section identifies the ESA-listed species that potentially occur within the Action Area that may be affected by U.S. Navy training and testing activities in the HSTT Action Area (Table 17). It then summarizes the biology and ecology of those species and what is known about their life histories in the Action Area. The species potentially occurring within the Action Area are listed in Table 17, along with their regulatory status.

4.1 ESA-listed Species and Designated Critical Habitat That May be Affected by the Action

Table 17. ESA-listed Species that May be Affected by U.S. Navy Training and Testing Activities

Species	ESA Status	Critical Habitat	Recovery Plan	Current Trend†
Marine Mammals – Cetaceans				
Blue Whale (*Balaenoptera musculus)*	E – 35 FR 18319	-- --	07/1998	?
Fin Whale (*Balaenoptera physalus)*	E – 35 FR 18319	-- --	75 FR 47538	?
Western North Pacific Gray Whale (*Eschrichtius robustus*)	E – 35 FR 18319	-- --		?
Humpback Whale (*Megaptera novaeangliae*)	E – 35 FR 18319	-- --	55 FR 29646	↑
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	-- --	76 FR 43985	?
Main Hawaiian Islands Insular False Killer Whale (*Pseudorca crassidens*)	E- 76 FR 70915	-- --	77 FR 71260	?
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	-- --	75 FR 81584	?
North Pacific Right Whale (*Eubalaena japonica*)	E – 35 FR 18319		78 FR 34347	?
Marine Mammals – Pinnipeds				
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	E – 41 FR 51611	53 FR 18988 Proposed revision - 76 FR 32026	72 FR 46966	↓
Guadalupe Fur Seal (*Arctocephalus townsendi*)	T - 50 FR 51252	-- --	-- --	↑
Sea Turtles				
Green Turtle (*Chelonia mydas*) – Mexico Breeding Population and all other areas	T/E – 43 FR 32800	63 FR 46693*	63 FR 28359	↓
Hawksbill Turtle (*Eretmochelys imbricata)*	E – 35 FR 8491	63 FR 46693*	63 FR 28359	↓
Olive Ridley Turtle (*Lepidochelys Oliacea*) – Eastern Pacific Population	E – 61 FR 17	-- --	63 FR 28359	↕
Leatherback Turtle (*Dermochelys coriacea*)	E – 61 FR 17	77 FR 4170	63 FR 28359	↓(Pacific)
Loggerhead Turtle (*Caretta caretta*) – North Pacific DPS	E – 76 FR 58868	Prop. 78 FR 43005	63 FR 28359	↓

Species	ESA Status	Critical Habitat	Recovery Plan	Current Trend†
Fish				
Steelhead Trout (*Oncorhynchus mykiss*) Southern California Distinct Population Segment (DPS)	E - 71 FR 834	70 FR 52488	01/2012	?
Scalloped Hammerhead Shark (*Sphyrna lewini)* – Eastern Pacific DPS	E – 79 FR 38213	-- --	-- --	?
Marine Invertebrates				
Black Abalone (*Haliotis cracherodii*)	E - 74 FR 1937	76 FR 66806	-- --	↓
White Abalone (*Haliotis sorenseni*)	E - 66 FR 29046	-- --	73 FR 62257	↓

***Critical habitat not present within the HSTT Action Area and not considered in this Opinion**

† Species status trends from the FY 2011 to 2012 Report to Congress (NMFS 2013): ↑ increasing, ↓ decreasing, ↕ mixed, ? unknown.

4.2 Species and Critical Habitat Not Considered Further

As described in the *Approach to the Assessment*, we use two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the various proposed activities. The first criterion is exposure or some reasonable expectation of a co-occurrence between one or more 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 listed species or designated 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 in Table 17; this subsection summarizes the results of those evaluations.

4.2.1 North Pacific Right Whale

The likelihood of a North Pacific right whale being present in the Action Area is extremely low as this species has only been observed rarely in the Bering Sea and Gulf of Alaska in recent years. The most recent estimated population for the North Pacific right whale is between 28 to 31 individuals and although this estimate may be reflective of a Bering Sea subpopulation, the total eastern North Pacific population is unlikely to be much larger (Wade et al. 2010). A right whale was last observed in the Maui Basin (Hawaiian waters) in April 1996 (Salden and Mickelsen 1999). Rare sightings of individual animals are typical of documented sightings, such as those of a single right whale on three occasions between March 25 and April 11, 1979, in Hawaiian waters (Herman et al. 1980, Rowntree et al. 1980). The only recorded sighting of a right whale in the SOCAL Range Complex area occurred in March 1992 approximately 43 miles (mi.) (70 kilometer [km]) off the southern end of San Clemente Island (Carretta et al. 1994). Sightings off

California are rare, and there is no evidence that the western coast of the United States was ever highly frequented habitat for this species (Brownell et al. 2001).

Individuals sighted near the Hawaiian Islands are considered "extralimital," as this region is not within the typical geographic range of this species (Reilly et al. 2008). Based on this information, it is highly unlikely for this species to be present in the Action Area. Therefore, we have determined that the likelihood of co-occurrence of Navy Training and Testing in SOCAL and Hawaii is suffiently low to be discountable and this species will not be considered further in this Opinion.

4.2.2 **Southern California Steelhead Trout and its Designated Critical Habitat**

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. They are a unique species; individuals develop differently depending on their environment. While all *O. mykiss* hatch in gravel-bottomed, fast-flowing, well-oxygenated rivers and streams, some stay in fresh water all their lives (commonly referred to as rainbow trout), whereas others migrate to the ocean after 1-3 years. 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. Steelhead tend to move immediately offshore on entering the marine environment, although, in general, steelhead tend to remain closer to shore than other Pacific salmon species (Beamish et al. 2005b). They generally remain within the coastal waters of the California Current (Beamish et al. 2005b). The ocean distributions for listed steelhead are not known in detail, but steelhead are caught only rarely in ocean salmon fisheries. Studies suggest that steelhead do not generally congregate in large schools as do other Pacific salmon species (Burgner et al. 1992a; Groot et al. 1991).

Trends in abundance and reproductive success of Pacific salmonids are typically observed through monitoring in the streams and rivers in which they spawn. (Boughton et al. 2005) assessed the occurrence of steelhead in southern California coastal watersheds in which the species occurred historically by conducting a combination of field reconnaissance and spot checks (snorkel surveys). Surveys indicated that between 38 percent and 45 percent of the streams surveyed in the range of the Southern California steelhead ESU contained the species, but that there were higher extirpation rates in the southern end of the range. Anthropogenic barriers appeared to be the factor most associated with extirpations. Of the 11 streams surveyed that drain into the HSTT Action Area, only San Mateo Creek contained steelhead. Though the authors expressed some uncertainty, NMFS (2005c) concluded that, with the exception of the small population in San Mateo Creek, the anadromous form of the species appears to be completely extirpated from all systems between the Santa Monica Mountains and the Mexican border. The San Mateo Creek population was formerly considered extirpated (Nehlsen et al. 1991), but California Department of Fish and Game documented presence of the species in 2003 NMFS (2005c). Many of the streams in this region contain resident populations of *O. mykiss*

((Boughton et al. 2005); NMFS (2005c)). However, fish from these populations in the watersheds that drain into the HSTT Action Area (e.g., San Diego River, Sweetwater River, Otay River) are not known to exhibit anadromy due to anthropogenic barriers to fish passage. The most recent monitoring data available for the Southern California steelhead ESU is from watersheds north of the HSTT Action Area (i.e., Santa Ynez River, Ventura River, Santa Clara River, Topanga Creek, Malibu Creek). Surveys indicated that very small (<10 fish), but consistent, runs of the species occur on an annual basis (Ford 2011). The most recent status review report for the Southern California steelhead ESU questioned how such small annual runs could persist, and speculated that the runs could be maintained either by strays from some another source population or by production of smolts from the resident population of rainbow trout (Ford 2011).

Popper (2003) and Hastings and Popper (2005) presented evidence that establishes that most fish only detect sounds within the 1 to 3 kHz range, which would make them sensitive to the lower end of the frequency range of mid-frequency active sonar. The U.S. Navy's Biological Evaluation for the Northwest Training Range Complex (Navy 2008b; Navy 2008c) provided a thorough review of the information available on the probable responses of endangered and threatened fish to active sonar. We have extracted most of the narratives that follow from that review, although we have made a few corrections and clarifications and supplemented the analyses with a few additional studies.

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.

Hastings et al. (1996) studied the effects of low frequency underwater sound on fish hearing. The authors exposed a teleost fish *Astronotus ocellatus* to sound of varying frequencies (60 or 300 Hz), duty cycles (20 percent or continuous), and intensity (100, 400, or 180dB re 1uPa). The only treatment where the authors observed some limited damage to sensory hair cells was with one hour of continuous exposure at 300 Hz and 180 dB, but this result was inconsistent. The authors recommended caution if attempting to extrapolate these result to other species or other sound sources, and also suggested that damage would be even more limited with shorter term stimulation or if fish were free to leave the site of stimulation. More recently, Popper et al. (2008; 2007; Popper and Hastings 2009) investigated the effects of exposing several fish species to the U.S. 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. Some fish experienced temporary losses in their hearing sensitivity but they recovered within several days of exposure.

Behavioral reactions of steelhead to non-impulsive acoustic sources could include temporary disruption or alteration of natural activities such as swimming, schooling, feeding, and migrating. Gearin et al. (2000) studied the effects of exposing fish to sounds produced by acoustic deterrent devices, which produce sounds in the mid frequency range. Adult sockeye salmon exhibited an initial startle response to the placement of inactive acoustic alarms but resumed their normal swimming pattern within 10 to 15 seconds. After 30 seconds, the fish approached the inactive alarm to within 30 cm (1 foot). When the experiment was conducted with an alarm active, the fish exhibited the same initial startle response from the insertion of the alarm into the tank; but were swimming within 30 cm of the active alarm within 30 seconds. After five minutes, the fish did not show any reaction or behavior change except for the initial startle response. In contrast, Doksaeter et al. (2009) observed no significant escape reactions from herring *Clupea harengus* in response to mid-frequency sonar transmissions. Based on the studies discussed above, if Southern California steelhead were exposed non-impulsive acoustic stressors (i.e., sonar), we would not expect this to result in direct mortality or injury. Behavioral disruptions could occur, but we would expect these impacts to be temporary and for individuals to resume normal activity shortly after exposure. We do not expect temporary behavioral reactions (e.g., cessation of feeding) to impact individual fitness as individuals would resume feeding upon cessation of the sound exposure and unconsumed prey would still be available in the environment.

Noise from Navy vessels is also not expected to impact Southern California steelhead as available evidence does not suggest that ship noise can injure or kill a fish (Popper et al. 2014). Further, we would expect the species to engage in avoidance behavior if vessels are moving in their direction. Misund (1997) found that fish ahead of a ship, that showed avoidance reactions, did so at ranges of 160 to 490 ft. (48.8–149.4 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. We do not expect temporary behavioral reactions (e.g., temporary cessation of feeding) to impact individual fitness as individuals will resume feeding upon cessation of the sound exposure and unconsumed prey will still be available in the environment.

The number of steelhead affected by an underwater explosion depends on the population density in the vicinity of the detonation, as well as the net explosive weight (NEW), depth of the explosion, and the size of the fish. Young (1991) provides equations that allow estimation of the potential effect of underwater explosions on fish. For example, it’s estimated that a one pound fish with a swim bladder (e.g., a steelhead) would need to be within 449 feet of the detonation of a 100 lb. NEW Maverick missile in order to be killed. A 30 pound fish with a swim bladder would need to be within 288 feet of the same detonation in order to be killed (Navy 2013c). Similarly, the number of steelhead affected by pile driving depends on the population density in the vicinity of the impact, the size of the fish, and the pile driving technique employed (i.e., impact or vibratory hammer). Pile driving is only anticipated to occur during the construction and removal phases of the elevated causeway training activities at the SSTC. Evidence suggests that fish within a few meters of the pile driving operation may be killed, but but the radius of impact is limited. For impact pile driving, the underwater noise threshold criteria for fish injury from a single pile strike occurs at a peak sound pressure level of 206 dB re 1 µPa. This sound level may be exceeded during impact pile driving within a circle centered at the location of the driven pile, out to a distance of approximately 60 ft. (18.3 m) (Navy 2013c). The radius of exposure from impulsive acoustic sources that may cause behavioral reactions (e.g., a startle response) is likely wider than these distances for both detonations and pile driving. However, we would expect these impacts to be temporary and for individuals to resume normal activity shortly after exposure. We do not expect temporary behavioral reactions (e.g., temporary disruption of feeding) to impact individual fitness as individuals will resume feeding upon cessation of the sound exposure and unconsumed prey will still be available in the environment.

In summary, the information available suggests extremely low abundance of Southern California steelhead in the Action Area. The only fish observed in a watershed that drains into the Action Area were in San Mateo Creek in 2002. Additionally, watersheds further north have very low documented abundance, with surveys indicating annual returns of less than 10 fish. Southern California steelhead eggs, fry, or juveniles still in freshwater habitats will not be exposed to Navy training and testing activities. Due to the low number of ESA-listed steelhead that are expected to occur over the relatively large Action Area, the infrequent and intermittent nature of Navy training and testing, and the proximity in which a steelhead in the marine environment would need to be located in order to be negatively effected by Navy training and testing activities, we conclude that HSTT exercises may affect, but are not likely to adversely affect endangered steelhead in the Southern California ESU because the probability of being exposed to stressors capable of eliciting a negative response is sufficiently low as to be discountable. Therefore, this species will not be considered in greater detail in the remainder of this Opinion.

NMFS designated critical habitat for Southern California steelhead on September 2, 2005 (70 FR 52488). A total of 708 miles of stream habitat was designated as critical habitat from the 32 watersheds within the range of this Evolutionary Significant Unit (ESU). Critical habitat for the Southern California Steelhead ESU includes most, but not all, occupied habitat from the Santa Maria River in southern San Luis Obispo County to San Mateo Creek in northern San Diego County, but excludes some occupied habitat based on economic considerations and all military lands with occupied habitat. Critical habitat was not designated for most of the watersheds south of Malibu Creek with the exception of San Juan Creek and San Mateo Creek.

These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this species identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. The critical habitat designation (70 FR 52488) contains additional description of the watersheds that are included as part of this designation, and any areas specifically excluded from the designation.

The critical habitat designation specifically excluded military areas for Southern California steelhead and the proposed training exercises and testing activities would not occur in freshwater areas or affect the primary constituent elements for steelhead. Therefore, the training exercises and testing activities the U.S. Navy proposes to conduct in the HSTT Action Area will not affect the designated critical habitat for Southern California steelhead. As a result, we will not consider Southern California steelhead critical habitat in the remainder of this Opinion.

4.2.3 Black Abalone and its Designated Critical Habitat

The black abalone (*Haliotis cracherodii*) is a large marine gastropod found in rocky intertidal and subtidal habitats. Both their "mantle" and "foot" are black. They have 5 to 9 open respiratory pores along the left sides of their shell and spiral growth lines on the rear. Black abalone are herbivores. During low tides, they can typically be found wedged into crevices, cracks, and holes of intertidal and shallow subtidal rocks, where they are fairly concealed (Leighton 2005). They generally occur in areas of moderate to high surf and range vertically from the high intertidal zone to a depth of about 20 ft (6 m) and are typically found in middle intertidal zones. However, variation in wave exposure and where drift kelp (an important food item for black abalone) accumulates may result in animals being distributed primarily in high or low intertidal zones depending on the local conditions at particular locations (see definition of intertidal zones in Ricketts et al. 1985). Black abalone can withstand extreme variation in temperature, salinity, moisture, and wave action. The species was listed as endangered on February 14, 2009 (74 FR 1937).

Black abalone historically occurred from Crescent City, California, USA, to southern Baja California, Mexico (Butler et al. 2009b), but today the species' constricted range occurs from Point Arena, California, USA, to Bahia Tortugas, Mexico, and it is rare north of San Francisco, California, USA (Butler et al. 2009b), and south of Punta Eugenia, Mexico (76 FR 66805).

Critical habitat was designated for black abalone on October 27, 2011 (76 FR 66805). Most of the designated critical habitat lies along the California coast north of the HSTT Action Area. Designated critical habitat includes rocky intertidal and subtidal habitats from the mean higher high water line to a depth of approximately 20 ft. (6 m), as well as the waters encompassed by these areas. Designated critical habitat extends from Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula. Within the Action Area, critical habitat occurs on Santa Catalina and Santa Barbara Islands. The specific areas proposed for designation off San Nicolas and San Clemente Islands were determined to be ineligible for designation because the Navy’s Integrated Natural Resources Management Plans provide benefits to black abalone in those areas. The critical habitat designation also identifies primary constituent elements, which are habitat elements essential for the conservation of the species. The primary constituent elements for black abalone are rocky substrate, food resources, juvenile settlement habitat, suitable water quality, and suitable nearshore circulation patterns.

The most important source of black abalone mortality is the disease known as withering syndrome caused by the bacterium *Candidatus Xenohaliotis californiensis*. Disease transmission and manifestation is intensified when local sea surface temperatures increase by as little as 2.5 °C above ambient sea surface temperatures and remain elevated over a prolonged period of time (i.e., a few months or more) (Ben-Horin et al. 2013; Friedman et al. 1997; Raimondi et al. 2002; Vilchis et al. 2005). Although there is no explicitly documented causal link between the persistence of withering syndrome and long-term climate change, patterns observed over the past three decades suggest that progression of ocean warming associated with large-scale climate change may facilitate further and more prolonged vulnerability of black abalone to the effects of withering syndrome.

Factors such as poaching, reduced genetic diversity, ocean acidification, non-anthropogenic predation (e.g., by octopuses, lobsters, sea stars, fishes, sea otters, and shorebirds) and competition (e.g., with sea urchins), food limitation, environmental pollutants and toxins, and substrate destruction may all impose mortality on black abalone at varying rates, but predicting the relative impacts of each of these factors on the long-term viability of black abalone is difficult without further study. In addition to the aforementioned present-day sources of mortality, commercial and recreational fisheries operating in California until 1993 likely contributed to the species' decline. For more information on historic and present-day factors leading to the decline of black abalone populations see Butler et al. (2009a).

Massive declines in black abalone began in 1986 that resulted in significant large-scale population reductions by the early 1990s (Lafferty and Kuris 1993). Evidence of population declines has also been observed in central California (Raimondi et al. 2002). The Black Abalone Status Review Team estimates that, unless effective measures are put in place to counter the population decline caused by withering syndrome and overfishing, the species will be extinct within 30 years (Butler et al. 2009a).

The black abalone population at one known location at San Nicolas Island may remain above a critical density threshold and is experiencing ongoing successful recruitment (VanBlaricom, unpublished data). The San Nicolas Island location is known to be characterized by small local sea surface temperature anomalies, with typical temperatures slightly lower (< 1 ºC on average) than at other monitored sites at the Island (VanBlaricom, unpublished data) (Butler et al. 2009a).

Black abalone could be exposed to underwater detonations associated with training exercies and testing activities; however, because the number of underwater detonations is small at 18 per year, the Navy has a practice of avoiding rocky habitat, and the density of black abalone is very low, the probability of black abalone being exposed to these activities is sufficiently small to be discountable. Similarly, the U.S. Navy has committed to restrict activities such as amphibious assaults, insertion and extraction, and Naval Fire Support to areas that would not support black abalone (Navy 2013e), so black abalone are not likely to be exposed to stressors associated with these activities. As a result, black abalone may be affected by the training exercises and testing activities activities the U.S. Navy proposes to conduct in the HSTT Action Area, but is not likely to be adversely affected by those activities. Therefore, this species will not be considered in greater detail in the remainder of this Opinion. Similarly, we do not expect primary constituent elements associated with black abalone critical habitat to be adversely affected and do not consider black abalone critical habitat further in this Opinion.

4.2.4 White Abalone

Historically, white abalone occurred from Point Conception, California to Punta Abreojos, Baja California, Mexico. They are the deepest-living of the west coast abalone species (Hobday and Tegner 2000): they had been caught at depths of 20 to 60 m (66 to 197 ft) but had been reported as having had the highest abundance at depths of 25 to 30 m (80 to 100 ft) (Cox 1960; Tutschulte 1976). At these depths, white abalone are found in open low relief rock or boulder habitat surrounded by sand (Davis et al. 1996; Tutschulte 1976).

Over the past 30 years, the white abalone populations have declined precipitously in abundance primarily as a result of exploitation. Surveys conducted at Tanner and Cortez Banks have yielded numbers of white abalone in the low hundreds (Butler et al. 2006). Surveys conducted off the western and south sides of San Clemente Island in 2004 yielded only 5 animals at 40 to 50 m depths (Stierhoff et al. 2014). A similar study in 2012 also found only five white abalone, although they were observed at 20 to 60 m depths (Stierhoff et al. 2014). Both studies located only adult white abalone (Stierhoff et al. 2014), suggesting recruitment may not be occurring at this location. The effects of activities associated with the Undersea Warfare Operations on invertebrates are not known, particularly the impacts of sound.

Other operations undertaken as part of Composite Training Unit or Joint Task Force Exercises, such as those involving underwater detonations, are not likely to adversely affect white abalone because the number of bottom-placed charges are few, these charges are not likely to adversely affect rocky habitat, and Sinking Exercises occur in at least 3,000 m of water, where white abalone are non-existent. Further, although Young (1991) did not study abalone, oysters (another mollusk) would be expected to exhibit 90 percent survival from the underwater explosion of a 20 pound explosive (a common explosive size for near-shore military underwater detonations) at a distance of 101 feet. Given the close proximity white abalone habitat would need to be to underwater detonations to be negatively affected in a significant manner, and the Navy's avoidance of these habitats, it is not reasonable to expect negative effects to occur.

We conclude that HSTT exercises may affect, but are not likely to adversely affect endangered white abalone because the probability of being exposed to stressors capable of eliciting a negative response is sufficiently low as to be discountable. Therefore, this species will not be considered in greater detail in the remainder of this Opinion.

4.2.5 **Critical Habitat for Leatherback Sea Turtle**

In 1979, NMFS designated critical habitat for leatherback turtles to include the coastal waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands (44 FR 17710).

In 2007, NMFS received a petition to revise the leatherback critical habitat designation to include waters off the U.S. West Coast. NMFS published a 90-day finding on the petition in December 2007. On January 26, 2012, we published a final rule (77 FR 4170). The revised critical habitat designated additional areas within the Pacific Ocean. This designation includes approximately 16,910 square miles (43,798 square km) stretching along the California coast from Point Arena to Point Arguello east of the 3,000 meter depth contour; and 25,004 square miles (64,760 square km) stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meter depth contour. The designated areas comprise approximately 41,914 square miles (108,558 square km) of marine habitat and include waters from the ocean surface down to a maximum depth of 262 feet (80 m). Other Pacific waters within the U.S. Exclusive Economic Zone (EEZ) were evaluated based on the geographical area occupied by the species, but we determined that they were not eligible for designation, as they do not contain the feature identified as essential to the conservation of the species.

The Critical Habitat Review Team (CHRT) identified two primary constituent elements (PCEs) essential for the conservation of leatherbacks in marine waters off the U.S. West Coast: (1)

Occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (*Chrysaora, Aurelia*, *Phacellophora*, and *Cyanea*) of sufficient condition, distribution, diversity, and abundance to support individual as well as population growth, reproduction, and development; and (2) Migratory pathway conditions to allow for safe and timely passage and access to/from/within high use foraging areas.

The training exercises and testing activities are not expected to alter or reduce the occurrence of prey species of the leatherback turtle and the CHRT determined that only permanent or long-term structures that alter the habitat would be considered as having effects on passage. Given this determination, the CHRT did not consider fishing gear or vessel traffic as threats to passage. Based on our analyses of the evidence available, the quantity, quality, or availability of the constituent elements or other physical, chemical, or biotic resources are not likely to decline as a result of being exposed to stressors associated with the activities the U.S. Navy proposes to conduct in the HSTT Action Area. Additionally, these stressors are not likely to exclude leatherback turtles from designated critical habitat or alter the primary constituent elements of the critical habitat. Therefore, the activities the U.S. Navy proposes to conduct in the HSTT Action Area may affect, but are not likely to adversely affect the designated critical habitat for leatherback turtles. As a result, we will not consider this critical habitat in greater detail in the remainder of this Opinion.

4.2.6 **Scalloped Hammerhead Shark – Eastern Pacific DPS**

On July 3, 2014, NMFS issued the final determination to list the Central and Southwest (SW) Atlantic Distinct Population Segment (DPS) and the Indo-West Pacific DPS of scalloped hammerhead shark as threatened species under the ESA. NMFS also issued a final determination to list the Eastern Atlantic DPS and Eastern Pacific DPS of scalloped hammerhead sharks as endangered species under the ESA. Unless otherwise noted, the information presented below was obtained from the Status Review Report for the Scalloped Hammerhead Shark (*Sphyrna lewini*) (Miller et al. 2014).

The scalloped hammerhead shark can be found in coastal warm temperate and tropical seas worldwide. In the eastern Pacific the scalloped hammerhead can be found from southern California to Peru, including the Gulf of California. The SOCAL Range Complex and Silver Strand Training Complex overlap with the extreme northern-most extent of the Eastern Pacific DPS of the scalloped hammerhead shark's range.

Scalloped hammerhead sharks are highly mobile and partly migratory. Migration is common along continental margins and between oceanic islands in tropical waters. Adult migratory movements are generally less than 200 km, but this species is also capable of moving much greater distances up to ~2,000 km. Juvenile movements are likely much shorter. Juveniles and adults occur as solitary individuals, pairs, or in schools, and there is evidence of site fidelity to known hot spots. A population of scalloped hammerhead sharks in the Gulf of California is known to school during the day and forage greater distances for food at night (Klimley and

Nelson 1984). Juvenile scalloped hammerhead sharks in Kāne'ohe Bay, Hawaii have similarly been known to school in shallow waters during the day and have increased movement rates and distances at night for foraging (Holland et al. 1993), suggesting that the near shore aggregation of scalloped hammerhead sharks during the day and greater offshore foraging at night is a general behavior of this species in the Pacific Ocean.

Scalloped hammerhead sharks primarily occur over continental and insular shelves and rarely in waters cooler than 22 °C. The species ranges from surface waters to depths of 512 meters (m), with occasional dives to deeper water up to 1000 m. It is also known to occur in bays and estuaries. Neonate and juvenile aggregations are more common in near shore nursery habitats because these habitats may provide valuable refuge from predation. Scalloped hammerhead sharks appear to prefer areas with stronger currents, greater turbidity, and higher sedimentation and nutrient flow. Based on the observation of 19 juveniles in 1997, it has been suggested the southern San Diego Bay may serve as a pupping ground and warm water refugium during warm water years (Lea and Rosenblatt 2000, Shane 2001), although NMFS would require further substantiation of this claim before identifying San Diego Bay as a pupping ground or nursery for this species.

Scalloped hammerhead sharks, like all fish, have an inner ear capable of detecting higher frequency sounds and a lateral line capable of detecting water motion caused by low frequencies (Hastings and Popper 2005; Popper and Schilt 2009). Data for cartilaginous fish suggest detection of sounds from 20 to 1000 Hz, with the highest sensitivity to sounds at the lower ranges (Casper et al. 2003; Casper and Mann 2006; Myrberg Jr. 2001). Scalloped hammerhead sharks lack a swim bladder which likely decreases their ability to detect or be affected by sound and explosive shockwaves (Young 1991).

The following Navy measures are likely to reduce the likelihood of effects to this species:

- Near shore underwater detonations are restricted to the boat lanes outside San Diego Bay associated with the Silver Strand Training Complex (SSTC) and immediately off San Clemente Island in a relatively small area. Ordnance use in deeper pelagic waters is typically restricted to daylight hours.

- Navy divers involved with underwater detonation in the SOCAL Range Complex will visually observe to the best extent practicable for hammerhead sharks prior to initiating detonation as part of the diver's normal underwater training procedures. If hammerhead sharks are observed within the immediate area, then detonation will be delayed until the shark is no longer observed in the immediate area.

- With the exception of the near shore underwater detonation site, a majority of the remaining explosives use is in deep, pelagic waters where scalloped hammerhead sharks are unlikely to occur.

- The use of items that could potentially entangle scalloped hammerhead sharks is restricted to deep water, pelagic areas where the occurrence of this species is unlikely. Fiber optic cables sink quickly, do not easily form loops, and break easily suggesting cable contact with any marine species is unlikely to have a significant impact on their individual fitness. Other potential entanglement stressors are often weighted and sink quickly, reducing the likelihood of fish entanglement.

The waters of the Pacific Ocean off the coast of southern California are relatively cold and rarely approach 22 °C, which is likely why the HSTT Action Area barely overlaps the known range for scalloped hammerhead sharks. Only 23 specimens have been verifiably recorded from southern California and 19 of those specimens were juveniles collected during a single extremely warm El Niño event in 1997 when sea water temperatures were 3 to 4 °C above normal. An analysis of hourly ocean temperature data (from the NOAA National Oceanic Data Center's La Jolla Station in southern California) averaged over each month from October 2013 through September 2014 indicates the months of June, July, August, and September had mean monthly temperatures of 20.2, 21.1, 21.8, and 20.8 °C, respectively. The calendar year 2014 was one of the warmest years on record globally, with temperatures along the western United States and eastern Pacific well above normal. Even with 2014 temperatures being above normal over the eastern Pacific Ocean and western United States, the average monthly summer near shore ocean temperatures did not exceed 22 °C along southern California.

It is expected that water temperatures within the SOCAL Range Complex and Silver Strand Training Complex will not exceed 22 °C except during extreme weather events such as the 1997 El Niño. Although climate change may lead to warmer Pacific temperatures off the coast of southern California, it is not expected to raise the temperature 3 to 4°C within the foreseeable future because the heat buffering capacity of water will likely cause ocean temperatures to rise at a slower pace than global air and land temperatures. Global air temperatures are expected to rise by less than 4.8 °C while global ocean temperatures within 100 m of the ocean's surface are expected to rise by less than 2.0 °C by the year 2100 (IPCC 2013).

The scalloped hammerhead shark is primarily a shallow water, coastal species and a majority of explosive ordnance use and all of activities with potential entanglement stressors will be performed 16 km or greater offshore over water depths greater than 512 m. Training also primarily occurs during the day when this species is more likely to be closer to shore. This suggests the co-occurrence of these activities with this species is unlikely even in the rare event water temperatures are sufficiently warm within the Action Area to support the species. Because these activities will occur where this species is not likely to exist, it is not reasonably expected that explosive ordnance use and entanglement stressors will coincide with the species. Therefore, effects from explosive ordnance and entanglement in deep, pelagic waters are discountable and NMFS has concluded these activities are not likely to adversely affect scalloped hammerhead sharks.

Only underwater detonations from Mine Neutralization training events will occur in near shore waters outside San Diego Bay within designated boat lanes and detonation sites. It is still not expected that scalloped hammerhead sharks will occur at this location except during extreme weather events causing elevated water temperatures. Underwater detonations at this location will utilize a maximum of a 20 pound explosive. Using the Young (1991) formula for effects of underwater detonations to fish without a swim bladder, it is estimated 90 percent of scalloped hammerhead sharks would survive at a range of 9.2 feet from a 20 pound explosive underwater detonation. Although sound thresholds inducing behavioral responses for scalloped hammerhead sharks have not been determined, it is expected they will be less susceptible than marine mammals and fish species with swim bladders.

For scalloped hammerhead sharks to be affected by the underwater detonations in the Silver Strand Training Complex, extreme weather events would be required to elevate water temperatures above 22 °C, the sharks would be required to migrate to the northern-most extent of their eastern Pacific range, and the sharks would need to be within close proximity to the infrequent and short duration detonations to be affected by them. These detonations would occur during the day when scalloped hammerhead sharks are less active (suggesting less likelihood of swimming into the detonation zone after diver observations) and divers make repeated observations prior to detonation. Therefore, it is not reasonable to expect near shore, underwater detonations will coincide with species occurrence or to expect the species to be affected. Therefore, effects from explosive ordnance in shallow, near shore waters is discountable and NMFS has concluded these activities are not likely to adversely affect scalloped hammerhead sharks.

We conclude that HSTT exercises may affect, but are not likely to adversely affect endangered scalloped hammerhead sharks in the eastern Pacific DPS because the probability of being exposed to stressors capable of eliciting a negative response is sufficiently low as to be discountable. Therefore, this species will not be considered in greater detail in the remainder of this Opinion.

4.3 Species Considered Further in this Opinion

Based on the anticipated exposure and response of species to stressors, we identified endangered and threatened species or critical habitat that are likely to be adversely affected by U.S. Navy training and testing activities in HSTT.

4.3.1 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 (108 ft) and can weigh more than 150,000 kg (330,700 lbs). 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.

4.3.1.1 ***Distribution***

Blue whales inhabit all oceans and typically occur near the coast, over the continental shelf, although they are also found in oceanic waters. Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999a; Reeves et al. 2004). Blue whales migrate toward the warmer waters of the subtropics in the fall to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a).

In the North Pacific Ocean, blue whales have been recorded off the island of Oahu in the main Hawaiian Islands and off Midway Island in the western edge of the Hawaiian Archipelago (Barlow 2006; Northrop et al. 1971; Thompson and Friedl 1982b), although blue whales are rarely sighted in Hawaiian waters and have not been reported to strand in the Hawaiian Islands.

The west coast of North America is known to be a feeding area for this species during summer and fall (Carretta et al. 2010b). This species has frequently been observed in waters off Southern California (Carretta et al. 2010b; Navy 2011a). Photographs of blue whales in California have been matched to individuals photographed off the Queen Charlotte Islands in northern British Columbia and the northern Gulf of Alaska (Calambokidis et al. 2009b). Three blue whale high-use areas occur during the summer along the coast of California near the western part of the Channel Islands, near the Gulf of Farallones, and at the northern part of Cape Mendocino (Irvine et al. 2014), although each of these areas are north of the HSTT Action Area. In the Southern California Bight, the highest densities of blue whales occurred along the 200-m isobath in waters with high surface chlorophyll concentrations (Redfern et al. 2013), although blue whale visual sightings and acoustic detections can occur across the entire Bight (Navy 2012b; Navy 2013a).

Blue whales observed in the spring, summer, and fall off California, Washington, and British Columbia are known to be part of a group that returns to feeding areas off British Columbia and Alaska (Calambokidis and Barlow 2004; Calambokidis et al. 2009c; Gregr et al. 2000; Mate et al. 1999; Moore et al. 2002; Stafford et al. 1999). These animals have shown site fidelity, returning to their mother's feeding grounds on their first migration (Calambokidis and Barlow 2004). Blue whales off the coast of California typically depart from near-shore U.S. Exclusive Economic Zone waters from mid-October to mid-November (Irvine et al. 2014). They are known to migrate to waters off Mexico and as far as the Costa Rica Dome (Calambokidis and Barlow 2004; Calambokidis et al. 2009c). Winter migration movements south along the Baja California, Mexico coast to the Costa Rica Dome indicate that the Costa Rica Dome may be a calving and breeding area (Mate et al. 1999). Blue whales belonging to the western Pacific stock may feed in summer, south of the Aleutians and in the Gulf of Alaska, and migrate to wintering grounds in lower latitudes in the western Pacific and central Pacific, including Hawaii (Stafford et al. 2004; Watkins et al. 2000a; Watkins et al. 2000b; Watkins et al. 2000c).

Blue whales in the eastern Pacific winter from California south (Carretta et al. 2013a); 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 2003). Fifteen blue whale sightings off British Columbia and in the Gulf of Alaska have been made since 1997 (Calambokidis et al. 2009b). Three of these photographically verified sightings were in the northern Gulf of Alaska within 71 nm of each other and were less than 100 nm offshore (Calambokidis et al. 2009b).

4.3.1.2 ***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 2003).

4.3.1.3 ***Natural Threats***

Natural causes of mortality in blue whales are largely unknown, but probably include predation and disease (not necessarily in 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 probably hunt blue whales as well (Perry et al. 1999a).

4.3.1.4 ***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 1982a). 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). Comparitively, (Monnahan et al. 2014b) estimates 3,411 (95 percent confidence interval (CI): 2,593 to 4,114) and 6,362 (95 percent CI: 5,659 to 7,180) blue whales were caught from 1905 to 1971 from the eastern North Pacific population and western North Pacific population, respectively. From 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984). Evidence of a population decline was seen in the catch data from Japan. In 1912, whalers captured 236 blue whales; in 1913, 58 blue whales; in 1914, 123 blue whales; from 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al. 1984). In the eastern North Pacific, whalers killed 239 blue whales off the California coast in 1926. And, in the late 1950s and early 1960s, Japanese whalers killed 70 blue whales per year off the Aleutian Islands (Mizroch et al. 1984).

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

Shipping is considered one of the primary threats to blue whales. 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). In 1980, 1986, 1987, and 1993, ship strikes were implicated in the deaths of blue whales off California (Barlow 1997b). 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. In 2007, National Oceanic and Atmospheric Administration declared an Unusual Mortality Event for endangered blue whales in Southern California as a result of commercial vessel ship strikes in that year. Available data from NMFS indicate that in waters off California between 1991 and 2010, there were 14 ship strikes involving blue whales (National Marine Fisheries Service Southwest Region Stranding Database 2011). Of these, three occurred in the same waters as HSTT, of which two were from commercial/research ships and one from Navy. Of the 14 blue whale strikes in California, 79 percent were in water north of the HSTT Action Area. With the exception of the one mentioned above, no identified blue whales have been struck by the Navy.

Although commercial fisheries using large gill nets or other large set gears poses some entanglement risk to marine mammals, there is little direct evidence of blue whale mortality from fishing gears. Therefore it is difficult to estimate the numbers of blue whales killed or injured by gear entanglements. The offshore drift gillnet fishery is the only fishery that is likely to take blue whales from this stock, but no fishery mortalities or serious injuries have been observed. In addition, the injury or mortality of large whales due to interactions or entanglements in fisheries may go unobserved because large whales swim away with a portion of the net or gear. Fishermen have reported that large whales tend to swim through their nets without becoming entangled and cause little damage to nets (Carretta et al. 2008).

4.3.1.5 ***Status and Trends***

Blue whales (including all subspecies) were originally listed as endangered in 1970 (35 FR 18319), and this status has continued since the inception of the ESA in 1973. Blue whales are also listed as endangered on the IUCN Red List of Threatened Animals (IUCN 2010) and are 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 numbered 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.

(Monnahan et al. 2014b) calculated a prewhaling abundance of 2,210 (95 percent CI: 1,823-3,721) for the eastern North Pacific population of blue whales. 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: 1,7774–2,584 (Monnahan et al. 2014b) (Fallis et al. 1983). Blue whale density estimates vary annually and by season. The average density of blue whales off the coast of southern California between 2004 and 2013 was 0.07, 0.07, 3.01, and 0.56 individuals per 1,000 square kilometres (km^2) in the winter, spring, summer, and fall, respectively (Campbell et al. 2014). The overall annual average density of blue whales off the coast of southern California over this time period was 0.91 individuals per km^2 (Campbell et al. 2014). There was a documented increase in the blue whale population size between 1979 and 1994, but there has not been evidence to suggest an increase in the population since then (Barlow 1994; Barlow and Taylor 2001; Carretta et al. 2010c) (Monnahan et al. 2014a). 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. 2009c). Given the current abundance of blue whales compared to prewhaling conditions, the cessation of whaling in the eastern North Pacific, and the minimal impact ship strikes are having on population growth/sustenance, Monnahan et al. (2014a) suggest the eastern North Pacific blue whale population is at carrying capacity and recovered[7]. Blue whales have not been observed during monitoring surveys in the Hawaii Range Complex, but have been observed during aerial monitoring in Southern California (Navy 2013a).

A vast majority of blue whales in the SOCAL training and testing area are from the eastern North Pacific population, while blue whales encountered around Hawaii could originate from either the eastern or western North Pacific populations (Monnahan et al. 2014b). To our knowledge, information is not available to estimate the current abundance of the western North Pacific blue whale population or to evaluate population trends.

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[7] Monnahan et al. (2014) defined recovery as when the population is above it's maximum net productivity, or 60% of it's carrying capacity.

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 (Monnahan et al. 2014a) (Monnahan et al. 2014b), Campbell et al. 2015). With the limited data available on other blue whale populations, 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).

4.3.1.6 ***Diving and Social Behavior***

Blue whales spend more than 94 percent of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5-20 times at 12-20 sec intervals before a deep dive of 3-30 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 (Corkeron et al. 1999; Fiedler et al. 1998; Schoenherr 1991; Shirihai 2002). Little is known of the mating behavior of blue whales. The primary and preferred diet of blue whales is krill (euphausiids).

Satellite tagging indicates that, for blue whales tagged off Southern California, movement is more linear and faster (3.7 km/h) while traveling versus while foraging (1.7 km/h)(Bailey et al. 2009). Residency times in what are likely prey patches averages 21 days and constituted 29 percent of an individual’s time overall, although foraging could apparently occur at any time of year for tagged individuals (Bailey et al. 2009). Broad scale movements also varied greatly, likely in response to oceanographic conditions influencing prey abundance and distribution (Bailey et al. 2009). Blue whales along Southern California were found to be traveling 85 percent of the time and milling 11 percent (Bacon et al. 2011). While feeding, blue whales show slowed and less obvious avoidance behavior then when not feeding (Sears et al. 1983 as cited in NMFS 2005a). In review of a 24-year blue whale sighting history, Sears et al. (Sears et al. 2013) documented a link between female blue whales sighted in the Gulf of California and the U.S. West Coast, although the authors suggest that only some of the U.S. West Coast blue whales migrate to the Gulf of California.

4.3.1.7 ***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 (1997) 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 of two types – songs and calls. Blue whale calls have high acoustic energy, with reports of 186 to 188 dB re 1 μPa-m (Cummings and Thompson 1971b; 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 1971b; McDonald et al. 2001b). The songs are divided into two components – 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. 2001); 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 2005).

In Southern Califronia, 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. 2007b). 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. 2007c).

Calling rates of blue whales tend to vary based on feeding behavior. Stafford et al. (2005) recorded the highest calling rates when blue whale prey was closest to the surface during its vertical migration. Wiggins et al. (2005) reported the same trend of reduced vocalization during daytime foraging followed by an increase at dusk as prey moved up into the water column and dispersed. Blue whales make seasonal migrations to areas of high productivity to feed, and vocalize less at the feeding grounds than during migration (Burtenshaw et al. 2004). Oleson et al. (2007c) 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 1997; Richardson et al. 1995c). Based on vocalizations and anatomy, blue whales are assumed to predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001c; Oleson et al. 2007c; Stafford and Moore 2005), although an audiogram of blue whale hearing thresholds suggests hearing may predominantly occur over a wider range of approximately 100 Hz to 10 kHz (Ketten 2014). 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. 2011b) and 13 in the Southern California-11 behavorial response study (Southall 2012). 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 2012). 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. 2013a). 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. Melcón et al. (2012a) 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.

4.3.1.8 ***Critical Habitat***

Blue whale critical habitat has not been designated.

4.3.2 **Fin Whale**

The fin whale, *Balaenoptera physalus* (Linnæus 1758), is a cosmopolitan species of baleen whale (Gambell 1985b). 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). Fin whales can be found in social groups of 2 to 7 whales.

4.3.2.1 ***Distribution***

Fin whales are distributed widely in every ocean except the Arctic Ocean. In the North Atlantic Ocean, fin whales occur in summer foraging areas from the coast of North America to the Arctic, around Greenland, Iceland, northern Norway, Jan Meyen, Spitzbergen, and the Barents Sea. In the western Atlantic, they winter from the edge of sea ice south to the Gulf of Mexico and the West Indies. In the eastern Atlantic, they winter from southern Norway, the Bay of Biscay, and Spain with some whales migrating into the Mediterranean Sea (Gambell 1985b).

In the Southern Hemisphere, fin whales are distributed broadly south of 50°S in the summer and migrate into the Atlantic, Indian, and Pacific Oceans in the winter, along the coast of South America (as far north as Peru and Brazil), Africa, and the islands in Oceania north of Australia and New Zealand (Gambell 1985b).

Fin whales undertake migrations from low-latitude winter grounds to high-latitude summer grounds and extensive longitudinal movements both within and between years (Mizroch et al. 1999a). Fin whales are sparsely distributed during November through April, from 60° N, south to the northern edge of the tropics, where mating and calving may take place (Mizroch et al. 1999a). However, fin whales have been sighted as far as 60° N throughout winter (Mizroch et al. 1999b). A resident fin whale population may exist in the Gulf of California (Tershy et al. 1993).

Fin whales are observed year-round off central and southern California with peak numbers in the summer and fall (Barlow 1997a; Dohl et al. 1983a; Forney et al. 1995a)(Campbell et al. 2015). Peak numbers are seen during the summer off Oregon, and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (Moore et al. 2000; Perry et al. 1999a). Fin whales are observed feeding in Hawaiian waters during mid-May, and their sounds have been recorded there during the autumn and winter (Balcomb 1987; Northrop et al. 1968b; Shallenberger 1981b; Thompson and Friedl 1982a). They have been recorded at Nihoa and other areas of the NWHI in the winter and spring months (Meigs et al. 2013). Fin whales in the western Pacific winter in the Sea of Japan, the East China, Yellow, and Philippine seas (Gambell 1985a). November 2008 surveys by the Marine Mammal Research Consultants within the Southern California portion of the Action Area resulted in the sighting of 22 fin whales. Navy sponsored monitoring in the SOCAL Range Complex in recent years also recorded the presence of fin whales (Navy 2011a; Navy 2012b; Navy 2013a). (Falcone and Schorr 2014) (2014) provide further evidence based on Southern California visual sighting records, photographic ID matches, and satellite tagging from 2006 to 2013 for a Southern California permanent or semi-permanent resident population of fin whales displaying seasonal distribution shifts within the region. Moore and Barlow (Moore and Barlow 2011) indicate that, since 1991, there is strong evidence of increasing fin whale abundance in the California Current area; they predict continued increases in fin whale numbers over the next decade, and that perhaps fin whale densities are reaching “current ecosystem limits.” Fin whales have been observed during aerial monitoring in Southern California (Navy 2010; Navy 2012b). Additionally, several fin whales were tagged during the Southern California (Southall et al. 2011a).

Open Ocean. The distribution of fin whales in the Pacific during the summer includes the northern area of the Hawaii portion of the Action Area to 32° N off the coast of California (Barlow 1995; Forney et al. 1995b). Fin whales are relatively abundant in north Pacific offshore waters, including the Hawaii portion of the Action Area (Berzin and Vladimirov 1981; Mizroch et al. 2009). Acoustic signals that may be attributed to the fin whale have also been detected in the Transit Corridor portion of the Action Area (Northrop et al. 1968a; Watkins et al. 2000b). Fin whales have been recorded in the eastern tropical Pacific (Ferguson 2005) and are frequently sighted there during offshore ship surveys. Fin whales were detected acoustically and visually sighted year-round within the SOCAL Range Complex from 2008 to 2013 (Navy 2011a; Navy 2012b; Navy 2013a). It is unclear if this represents a distinct fin whale sub-population with semipermenant status in the region, or is reflective of frequent transit by fin whales in general through and within Southern California (Navy SOCAL annual monitoring reports).

Fin whales were observed twice during a NMFS survey of waters within the Hawaiian Exclusive Economic Zone in 2010 (Bradford et al. 2013), five sightings were made in offshore waters during a NMFS 2002 survey in the same region, and a single sighting was made during aerial surveys conducted between 1993 to 1998 (Barlow 2006; Carretta et al. 2010b; Mobley Jr. et al. 2000b). There are other known sightings from Kaua‘i and Oahu, Hawaii, and a single stranding record from Maui, Hawaii (Mobley Jr. et al. 1996; Shallenberger 1981a); the most recent sighting was a single juvenile fin whale incidentally reported off Kauai in 2011 (Navy 2011a). Based on sighting data and acoustic recordings, fin whales are likely to occur in Hawaiian waters mainly in fall and winter (Barlow 2006). There was a fin whale sighted off of Lanai during Navy monitoring in December of 2012 as well as January 2015 (HRC 2013). Additionally, An incidental sighitng of a fin whale was made, and a photo was taken, off the Island of Hawaii by a commercial operator on January 8, 2015. The species idenfication was confirmed as a fin whale (pers. comm Robin Baird).

Locations of breeding and calving grounds for the fin whale are unknown, but it is known that the whales typically migrate seasonally to higher latitudes every year to feed and migrate to lower latitudes to breed (Kjeld et al. 2006; Macleod et al. 2006). The fin whale’s ability to adapt to areas of high productivity controls migratory patterns (Canese et al. 2006; Reeves et al. 2002). Fin whales are one of the fastest cetaceans, capable of attaining speeds of 25 mi. (40.2 km) per hour (Jefferson et al. 2008; Marini et al. 1996).

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 1985b). 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.

4.3.2.2 ***Population Structure***

Fin whales have two recognized subspecies: *Balaoptera physalus physalus* occurs in the North Atlantic Ocean while *B. p. quoyi* (Fischer 1829) occurs in the Southern Ocean. A third possible subspecies occurs off South America (Gray 1865; Van Waerebeek and Engblom 2007) (Archer et al. 2013). 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.

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, although 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. Naval-sponsored passive acoustic monitoring along Southern California found year-round vocalization detections from 2009 through 2012 (Navy 2013a). Fin whales along Southern California were found to be traveling 87 percent of the time and milling 5 percent in groups that averaged 1.7 individuals (Bacon et al. 2011).

4.3.2.3 ***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).

4.3.2.4 *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). The Japanese whalers planned to kill 50 whales per year starting in the 2007 to 2008 season and continuing for the next 12 years (IWC 2006; Nishiwaki et al. 2006).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (Carretta et al. 2007; Douglas et al. 2008; 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’s (2004) review of the NMFS’ 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/Hawaiʹi. Between 1999 and 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. 2008). 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 2004).

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 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).

4.3.2.5 ***Status and Trends***

Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status has continued 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 thoughout the rest of their range (NMFS 2011b).

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.

4.3.2.6 *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 1985b; 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 1981d). 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 (2003) reported mean group sizes of 1.1 to 4.0 during surveys off California, Oregon, and Washington.

4.3.2.7 *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 1981c; 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. (1995c) 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 (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 1981c). 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 (1997) 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 sensitivy 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 1997; Richardson et al. 1995c).

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. 1995c; Sirovic et al. 2007; Watkins 1981c; 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 1981c; 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 (1981c), 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 1997). 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). 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. 2007c).

4.3.2.8 *Critical Habitat*

Fin whale critical habitat has not been designated.

4.3.3 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 become sexually mature between 6 and 12 years, at an average of 8 years old. After 12 to 13 months of gestation, females give birth to a single calf. Newborn calves are approximately 14 to 16 ft (4.5 to 5 m) long, and weigh about 2,000 lb (920 kg). The average and maximum life span of gray whales is unknown, although one female was estimated at 75 to 80 years old after death (Jones and Swartz, 2002).

4.3.3.1 *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 1990a; 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).

4.3.3.2 ***Population Structure***

Gray whales occur in two genetically and spatially 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. 2010b; Leduc et al. 2002; Swartz et al. 2006; Weller et al. 2007; Weller et al. 2004b; Weller et al. 2006a).

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. 2008a; 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). Of those 23 individuals, only 15 were recorded from far enough south to have passed through the HSTT area. 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. 2004b; Weller et al. 2006a; Weller et al. 1999; Yakovlev and Tyurneva 2004).

4.3.3.3 ***Natural Threats***

Predation by killer whales is a significant threat to gray whales, with calves being particularly susceptible during their northward migration (Fay et al. 1978; Goley and Straley 1994; Poole

1984; Rice and Wolman 1971; Ternullo and Black 2002). However, not all attacks are fatal and many individuals escape with scars from the encounters. Killer whales are frequently observed in the primary western North Pacific gray whale foraging area and roughly one-third to one half of observed gray whales bear tooth marks from killer whales (30 percent of them during the course of the 10 year study)(Bradford et al. 2003; Bradford et al. 2006a; Weller et al. 2009a; Weller et al. 2002a). Vladimirov (2005) documented an attack on a mother-calf pair in shallow waters of the Piltun Bay foraging area. This rate is among the highest rate found amongst baleen whales. Researchers have also expressed significant concern about whales that appear "skinny"; the cause and consequences remain unknown (Bradford et al. 2007; Bradford et al. 2008a; Bradford et al. 2012b; Burdin et al. 2003; IWC 2003; Weller et al. 2008b; Weller et al. 2007; Weller et al. 2005; Weller et al. 2004a; Weller et al. 2004b; Weller et al. 2006a; Yakovlev and Tyurneva 2004). Lactating females appear to be in particularly poor body condition (Bradford et al. 2012b).

4.3.3.4 ***Anthropogenic Threats***

Western North Pacific gray whales experience many of the same human-induced threats as other baleen whales, including entanglement and ship strike. At least one fifth of individuals show evidence of entanglement in fishing gear (Bradford et al. 2006b) (Bradford et al. 2009). Four females were bycaught in fishing nets and died along Japan from 2005 through 2007 (Bradford et al. 2006b; Brownell Jr. 2007; Cooke et al. 2005; Cooke et al. 2008; Cooke et al. 2007; Kato et al. 2006; Kato et al. 2007; Weller et al. 2008b). Bradford et al. (2006b) and (2009) found that 1.8 to 2.0 percent of individuals showed scars consistent with ship strike. Another individual was found stranded in 1996 with several harpoons in it, likely from Japanese fishers (Brownell Jr. and Kasuya. 1999). Extensive oil and gas exploration and development is occurring near the summer foraging areas for western North Pacific gray whales, introducing noise, additional ship strike, pollutants, and the potential for oil spills to the region that causes concern for the recovery of western North Pacific gray whales (Anonymous 2009; Brownell 2004; Commission 2004; Donovan 2005; Gailey et al. 2007; IWC 2003; Johnson et al. 2007; Nowacek et al. 2012; Reeves et al. 2005; Reeves 2005; Webster 2003; Weller et al. 2008b; Weller et al. 2002b; Weller et al. 2006b; Weller et al. 2006c; Yazvenko et al. 2007a; Yazvenko et al. 2007b).

4.3.3.5 ***Status and Trends***

Gray whales throughout the North Pacific Ocean were originally listed on June 2, 1970 (35 FR 8495). On June 16, 1994 (59 FR 21094), the eastern North Pacific gray whales were delisted, but western North Pacific gray whales remain listed through the present as endangered.

Alter et al. (2007) concluded that eastern and western North Pacific gray whales historically numbered between 76,000 and 118,000 individuals combined prior to whaling; the proportion of individuals that was in each population is unknown. However, whaling dramatically reduced the population to a tiny fraction of its former abundance, with 100 to 130 non-calves remaining (Burdin et al. 2010; Cooke et al. 2005; Cooke et al. 2008; Reeves et al. 2008; Wade et al. 2003;

Weller et al. 2005; Weller et al. 2006a). The most recent abundance estimate of the western North Pacific gray whale population is 140 individuals (Carretta et al. 2015, unpublished 2014 stock assessment reports). The population was believed to be extinct in the 1970’s (Bradford et al. 2003). At least 1,700 to 2,000 individuals were commercially harvested from the late 1800’s to the mid 20th century (Commission 2004; IWC 2003). Findings that eastern North Pacific gray whales may be found within the range of western North Pacific gray whales may mean that even fewer individuals compose the western population, as individuals formerly believed to be western individuals may actually be part of the eastern population (Lang et al. 2010b).

Fortunately, the latest data on population growth indicates a positive trajectory for available data over 1994 through 2007 of roughly 2.5 to 3.2 percent growth per year (Bradford et al. 2008b; Cooke et al. 2008; Cooke et al. 2007; Cooke et al. 2006). However, the loss of a single adult female would strongly decrease this trajectory (Cooke et al. 2005). In 2009, Burdin et al. (2010) reported 26 mature females observed since 1995. Genetic findings have found that although genetic diversity is relatively high in western North Pacific gray whales considering their population size, significant portions of this diversity are retained in a few or single individuals (IWC 2003). The loss of one or a few of these individuals would greatly reduce the genetic diversity of the population as a whole. Also of significance is that only about half of males fathering offspring have been identified, supporting a larger population size than is currently known (Lang et al. 2010a; Lang et al. 2010b). There is a strong male bias in calf production of roughly 2:1 (Burdin et al. 2003; Cooke et al. 2008; Weller et al. 2009b; Weller et al. 2008b; Weller et al. 2004a; Weller et al. 2004b). Clapham et al. (1999b) conducted a review of western North Pacific gray whales, among other endangered whales, and found that this population matches in virtually all characteristics that would make a small population extinction-prone.

4.3.3.6 *Vocalizations and Hearing*

No data are available regarding western North Pacific gray whale hearing or communication. We assume that eastern North Pacific gray whale communication is representative of the western population and present information stemming from this population. Individuals produce broadband sounds within the 100 Hz to 12 kHz range (Dahlheim et al. 1984; Jones and Swartz 2002; Thompson et al. 1979). The most common sounds encountered are on feeding and breeding grounds, where “knocks” of roughly 142 dB re: 1 µPa at 1 m (source level) have been recorded (Cummings et al. 1968; Jones and Swartz 2002; Thomson and Richardson 1995). However, other sounds have also been recorded in Russian foraging areas, including rattles, clicks, chirps, squeaks, snorts, thumps, knocks, bellows, and sharp blasts at frequencies of 400 Hz to 5 kHz (Petrochenko et al. 1991). Estimated source levels for these sounds ranged from 167-188 dB re: 1 µPa at 1 m (Petrochenko et al. 1991). Low frequency (<1.5 kHz) “bangs” and “moans” are most often recorded during migration and during ice-entrapment (Carroll et al. 1989; Crane and Lashkari. 1996). Sounds vary by social context and may be associated with startle responses (Rohrkasse-Charles et al. 2011). Calves exhibit the greatest variation in frequency range used, while adults are narrowest; groups with calves were never silent while in calving grounds (Rohrkasse-Charles et al. 2011). Based upon a single captive calf, moans were more frequent when the calf was less than a year old, but after a year, croaks were the predominant call type (Wisdom et al. 1999).

Auditory structure suggests hearing is attuned to low frequencies (Ketten 1992a; Ketten 1992b). Responses of free-ranging and captive individuals to playbacks in the 160 Hz to 2 kHz range demonstrate the ability of individuals to hear within this range (Buck and Tyack 2000; Cummings and Thompson 1971a; Dahlheim and Ljungblad 1990; Moore and Clark 2002; Wisdom et al. 2001). Responses to low-frequency sounds stemming from oil and gas activities also support low-frequency hearing (Malme et al. 1986b; Moore and Clark 2002).

4.3.3.7 ***Critical Habitat***

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

4.3.4 **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 (6570 cm) to those of other baleen whales. Humpbacks in different geographical areas vary somewhat in body length, but maximum recorded size is 18m (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).

4.3.4.1 ***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 coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985).

In the eastern and central 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 calving grounds near Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter months.

In the Hawaii portion of the Action Area, 947 sightings of 1,594 individuals were made by Navy-funded surveys in 2005 through 2012 (HDR 2012a). Sixty-four of these sightings (11 percent of total, n=107) were observed within the boundaries of the Hawaiian Islands Humpback Whale National Marine Sanctuary, defined by the 100 fathom (183 m) isobaths off portions of the main Hawaiian Islands, and approximately half that number of sightings were observed outside the 1000 fathom (1830m) isobaths.

4.3.4.2 *Population Structure*

Though the ESA-listed entity is the worldwide population of humpback whales, some evidence suggests there may be multiple distinct populations within the North Pacific Ocean. 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, Hill and DeMaster (1998) recognized 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 1998b). 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 Hawaii, 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 Hawaii while the eastern North Pacific population (also called the California-Oregon-Washington 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 Hawaii 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 through March (Darling and Cerchio 1993; Eldredge 1991; Eldredge 2003; Fulling et al. 2011b; 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° and 48°N, with limited interchange with areas north of Washington State (Calambokidis et al. 2004; Calambokidis et al. 1996).

Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992a; Tynan et al. 2005).

4.3.4.3 ***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.

4.3.4.4 ***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. 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 2003b). 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.

4.3.4.5 ***Status and Trends***

Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. On August 29, 2013, NMFS announced a 90-day finding on a April 17, 2013, petition to identify the North Pacific humpback population as a Distinct Population Segment (DPS) (78 FR 53391).

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). The overall abundance of humpback whales in the north Pacific was recently estimated at 21,808 individuals (coefficient of variation = 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. 2011a). Data indicates the north Pacific population has been increasing at a rate of between 5.5 percent and 6.0 percent per year, therefore approximately doubling every 10 years (Calambokidis et al. 2008a). The current (year 2008) best estimate of abundance for the California, Oregon, and Washington stock is 1,918 (coefficient of variation = 0.03), with an estimated mimimum population size estimate of 1,876 individuals (Carretta et al. 2014c). Based on ship surveys conducted in the summer and fall from 1991 to 2005, it is estimated that 36 humpback whales (coefficient of variation = 0.51) occur off Southern California in the waters south of Point Conception (Barlow and Forney 2007). Humpback whale density estimates vary annually and by season. The average density of humpback whales off the coast of southern California between 2004 and 2013 was 1.07, 1.92, 1.03, and 0.82 individuals per 1,000 km^2 in the winter, spring, summer, and fall, respectively (Campbell et al. 2014). The overall annual average density of humpback whales off the coast of southern California over this time period was 1.17 individuals per km^2 (Campbell et al. 2014).The current (year 2006) best estimate of abundance for the Central North Pacific stock has been estimated at 10,103 individuals on wintering grounds throughout the main Hawaiian Islands (Allen and Angliss 2013a), with an estimated minimum population size of 5,833 individuals (Carretta et al. 2014c).

The Hawaiian Islands Humpback Whale National Marine Sanctuary reported in 2010 that as many as 12,000 humpback whales migrate to Hawaiian waters each year (Barlow et al. 2011b) (Calambokidis et al. 2008a). Mobley (2011) remarked on the doubling of this species' sighting rate during a 2011 aerial survey (0.04 sightings/km) when compared to the 2006 North Pacific Acoustic Laboratory aerial survey (0.02 sightings/km) (Mobley 2011). This sighting rate further increased to 0.065 sightings/km during the February 2012 SCC aerial survey (Mobley and Pacini 2012).

4.3.4.6 ***Diving and Social Behavior***

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

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

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

4.3.4.7 *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. 2006; Au et al. 2000b; Frazer and Mercado III 2000; Richardson et al. 1995c; 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. 1995c; 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. 1995c; Thompson et al. 1986). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995c).

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. (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. 2006; Winn et al. 1970).

Social calls range from 20 Hz to 10 kHz, with dominant frequencies below 3 kHz (D'Vincent et al. 1985; Dunlop et al. 2008; Silber 1986; Simao and Moreira 2005). Female vocalizations appear to be simple; Simão 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. 1985; Thompson et al. 1986). The fundamental frequency of feeding calls is approximately 500 Hz (D'Vincent et al. 1985) (D’Vincent et al. 1985; Thompson et al. 1986). The acoustics and dive profiles associated with humpback whale feeding behavior in the northwest Atlantic has been documented with Digital Acoustic Recording Tags (DTAGs[8]) (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, 2006) 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

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[8] *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 & Tyack 2003).*

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 (Maybaum 1990) reported that humpback whales showed a mild response to a handheld sonar marine mammal detection and location device with frequency of 3.3 kHz at 219 dB re 1μPa-m or frequency sweep of 3.1 kHz to 3.6 kHz (although it should be noted that this system is significantly different from the Navy's hull mounted sonar). In addition, the system 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.

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. 2007c).

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 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 et al. 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. 2013). 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 et al. 2013).

4.3.4.8 *Critical Habitat*

Humpback whale critical habitat has not been designated.

4.3.5 **Sei Whale**

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

Southern Hemisphere. These large animals can reach lengths of 40 to 60 ft (12 to 18 m) and weigh 100,000 lbs (45,000 kg). Females may be slightly longer than males. Sei whales have a long, sleek body that is dark bluish-gray to black in color and pale underneath. The body is often covered in oval-shaped scars (probably caused from cookie-cutter shark and lamprey bites) and sometimes has subtle "mottling".

The Sei is regarded as one of the fastest swimmers 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 become sexually mature at 6 to 12 years of age when they reach about 45 ft (13 m) in length, and generally mate and give birth during the winter in lower latitudes. Females breed every 2 to 3 years, with a gestation period of 11 to 13 months. Females give birth to a single calf that is about 15 ft (4.6 m) long and weighs about 1,500 lbs (680 kg). Calves are usually nursed for 6 to 9 months before being weaned on the preferred feeding grounds. Sei whales have an estimated lifespan of 50 to 70 years.

4.3.5.1 ***Distribution***

The sei whale occurs in all oceans of the world except the Arctic. The migratory pattern of this species is thought to encompass long distances from high-latitude feeding areas in summer to low-latitude breeding areas in winter; however, the location of winter areas remains largely unknown (Perry et al. 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 1985c).

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 1985c; Masaki 1977a). 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. Sei whales have been seen in monitoring efforts in Hawaii in 2007 and in 2010.

4.3.5.2 *Population Structure*

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

Mark-recapture, catch distribution, and morphological research indicate more than one population may exist in the North Pacific – one between 155° and 175° W, and another east of 155° W (Masaki 1976a; Masaki 1977a). 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. In Navy-funded surveys 2007 through 2012, there were three confirmed sighting of sei whales for a total of five individuals—all made from vessels (HDR 2012b). Two sightings were documented northeast of Oahu in 2007 (Smultea et al. 2007), while the third was encountered near Perret Seamount west of the Island of Hawaii in 2010 (HDR 2012b). Bottom depths for the sei whale sightings were from 3,100 to 4,500 m. Sightings were made during BSS 2-4. Smultea et al. (2010) noted that the lack of sightings of sei whales in the Hawaiian Islands may be due to misidentification and/or poor sighting conditions. Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998b). Whaling data suggest that sei whales do not venture north of about 55° N (Gregr et al. 2000). 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°. 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). Masaki (1977a) reported sei whales concentrating in the northern and western Bering Sea from July through September, although other researchers question these observations because no other surveys have reported sei whales in the northern and western Bering Sea.

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). Masaki (1977a) 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. Sei whales are distributed in offshore waters in the Southern California portion of the HSTT Action Area. There are records of sightings in California waters as early as May and June, but primarily are encountered there during July to September and leave California waters by mid-October. Aerial surveys conducted in October and November 2008 off the Southern California coast resulted in the sighting of one sei (or possibly fin) whale (Oleson and Hill 2009). On March 18, 2011 off Maui, the Hawaiian Islands Entanglement Response Network found a subadult sei whale entangled in rope and fishing gear. An attempt to disentangle the whale was unsuccessful although a telemetry buoy attached to the entangled gear was reported to be tracking the whale over 21 days as it moved north and over 250 nm from the Hawaiian Islands.

4.3.5.3 *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).

4.3.5.4 *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.

4.3.5.5 ***Status and Trends***

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

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). Between 1991 and 2001, during aerial surveys, there were two confirmed sightings of sei whales along the U.S. Pacific coast. The most recent abundance estimates for sei whales that we are aware of range from 7,260 to 12,620 in the North Pacific Ocean (Tillman et al. 1997) and 25,000 individuals worldwide (Braham 1991).

4.3.5.6 ***Diving and Social Behavior***

Generally, sei whales make 5 to 20 shallow dives of 20 to 30 second duration followed by a deep dive of up to 15 minutes (Gambell 1985c). 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 1985c).

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 through August is northern anchovy, while in September and 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.

4.3.5.7 *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. 1995c).

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

While no data on hearing ability for this species are available, Ketten (1997) hypothesized that mysticetes have acute infrasonic hearing. 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. 2007). There are no tests or modeling estimates of specific sei whale hearing ranges.

4.3.5.8 *Critical Habitat*

Sei whale critical habitat has not been designated.

4.3.6 Sperm Whale

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

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

4.3.6.1 *Distribution*

Sperm whales are distributed in all of the world’s oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the

Southern Ocean (Perry et al. 1999a; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50° 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.

4.3.6.2 ***Population Structure***

There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). The 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 Hawai′i; (Perry et al. 1999b; 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 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead 2008).

Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 40° 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. 1995c; Shallenberger 1981a). In the Hawaii portion of the HSTT Action Area there were 26 confirmed sperm whale sightings and a total of five individuals during Navy-funded surveys from 2006 to 2012 (HDR 2012b). Five sightings were noted off the west coast of Kaua‘i and another confirmed sperm whale sighting was in the channel between Oahu and Kaua‘i. The remaining sperm whale sightings were made off the west coast of the island of Hawaii. Bottom depth for this sighting was between 800 and 4,600 m. Cetos (2005) reported on an acoustic detection, and an additional sighting of a probable sperm whale was reported for the Alenuihaha Channel area by Cetos (Smultea et al. 2007). In a Navy-funded survey in May 2011, a pod of 20 sperm whales with at least two calves was seen within the SOCAL Range Complex, approximately 30 miles from San Diego.

Sperm whales are seen in every season except winter (December and February) in Washington and Oregon (Green et al. 1992a). In surveys of waters off Oregon and Washington conducted by Green et al. (1992a), no sperm whales were encountered in waters less than 200 meters deep, 12 percent of the sperm whales were encountered in waters 200 to 2000 meters deep (the continental slope), and the remaining 88 percent of the sperm whales were encountered in waters greater than 2,000 meters deep. 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 (Forney 2007). In May 2011, a pod of 20 sperm whales including two calves was sighted during aerial surveys approximately 44 km west of San Diego in waters 200 to 300 meters deep (Navy 2011).

Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly towards the middle of the tropical Pacific and northward towards the tip of Baja California (Carretta et al. 2006). Sperm whales occupying the California Current region are genetically distinct from those in the eastern tropical Pacific and Hawaiian waters (Mesnick et al. 2011). The discreteness of the later two areas remains uncertain (Mesnick et al. 2011).

Off California, sperm whales are present in offshore waters year-round, with peak abundance from April to mid-June and again from late August through November (Barlow 1997b; Dohl et al. 1981; Dohl et al. 1983b; Gosho et al. 1984). The majority of sightings off central and northern California were in waters deeper than 1,800 m (5,900 ft), but near the continental shelf edge (well offshore of the Action Area) (Dohl et al. 1983b). Navy surveys have found sperm whales in the Action Area (Navy 2012).

4.3.6.3 *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).

4.3.6.4 *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 through 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.

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). Following a moratorium on whaling by the International Whaling Commission, 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 2004). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006).

Based on reports from 2000 to 2010, a total of two sperm whales were entangled in fishing gear off California, both of which were reported within the Southern California Bight (Saez et al. 2013). Available data from NMFS indicate that in waters off California between 1991 and 2010, there was one ship strike involving a sperm whale (National Marine Fisheries Service Southwest Region Stranding Database 2011).

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 1998b; 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 cavitations (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.

4.3.6.5 *Status and Trends*

Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status has remained since 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).

The most comprehensive abundance estimate for sperm whales we are aware of is from Whitehead (2002a), who estimated that there are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii, and western North Pacific, and a worldwide population of 360,000 individuals. 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). Sperm whales within the northern-most portion of the HSTT Action Area are estimated at 26,300 (Barlow and Taylor 2005).

Sperm whale abundance varied off California between 1979/80 and 1991 (Barlow 1994) and between 1991 and 2008 (Barlow and Forney 2007). The estimate from 2008 is the lowest to date, in sharp contrast to the highest abundance estimates obtained from 2001 and 2005 surveys. There is no reason to believe that the population has declined; the most recent survey estimate likely reflects interannual variability in the Action Area.

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947-1987. 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.

4.3.6.6 *Diving and Social Behavior*

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 8-11 minute 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 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Unlike other cetaceans, there is a preponderance of dive information for this species, most likely because it is the deepest diver of all cetacean species and therefore 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).

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 1956; Rice 1989). Sperm whales have been observed near Long Island, New York, in water between 40 and 55 m deep (Scott and Sadove 1997).

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. (2005) reported increased sperm whales densities with strong turbulence associated topographic features along the continental slope near Heceta Bank. Two noteworthy strandings in the region include an infamous incident (well publicized by the media) of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. In addition, a mass stranding of 47 individuals in Oregon occurred during June 1979 (Norman et al. 2004a; Rice et al. 1986).

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

4.3.6.7 *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; Madsen et al.

2003; Weilgart and Whitehead 1997; Weilgart et al. 1993). 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 2006c; Weilgart et al. 1993). 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). These long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). However, clicks are also used in short patterns (codas) during social behavior and intra-group interactions (Weilgart et al. 1993). 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 1985; Watkins and Schevill 1975). 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 1999). Sperm whales typically produce short-duration repetitive broadband clicks with frequencies below 100 Hz to >30 kHz (Watkins 1977, see Thomson and Richardson 1995) 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 (Møhl 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 (Whitehead and Weilgart 1991; Miller et al. 2004). 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 (Miller et al. 2004; Laplanche et al. 2005).

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 (Weilgart and Whitehead 1997; Rendell and Whitehead 2004). 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 (Weilgart and Whitehead 1997; Pavan et al. 2000). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean and those in the Pacific (Weilgart and Whitehead 1997). 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 and Schevill 1975; Watkins et al. 1985). In the Caribbean, Watkins et al. (1985) 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,. 1985). 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. 2007).

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 (Southall et al. 2011; DoN 2011; Miller et al. 2012; Sivle et al. 2012).

4.3.6.8 *Critical Habitat*

Sperm whale critical habitat has not been designated.

4.3.7 Main Hawaiian Islands Insular False Killer Whale

Main Hawaiian Islands (MHI) insular false killer whales (*Pseudorca crassidens*) are large members of the dolphin family. Females reach lengths of 15 feet (4.5 m), while males are almost 20 feet (6 m). In adulthood, false killer whales can weigh approximately 1,500 pounds (700 kg). They have a small conical head without a beak. Their dorsal fin is tall and their flippers (pectoral fins) have a distinctive hump or bulge in the middle of the front edge. False killer whales have dark coloration except for some lighter patches near the throat and middle chest. Their body shape is more slender than other large delphinids.

4.3.7.1 *Distribution*

The MHI insular false killer whale distinct population segment occurs near the main Hawaiian Islands. Distribution of MHI insular false killer whales has been assessed using data from visual surveys and satellite tag data. Tagging data from seven groups of individuals tagged off the islands of Hawaii and Oahu indicate that the whales move rapidly and semi-regularly throughout the main Hawaiian Islands and have been documented as far as 112 km offshore over a total range of 82,800 km^2 (Baird et al. 2012a; Baird et al. 2012b). Three high-use areas were identified: (1) off the north half of Hawaii Island, (2) north of Maui and Molokai, and (3) southwest of Lanai (Baird et al. 2012a). However, note that limitations in the sampling, suggest the range of the population is likely underestimated, and there are probably other high-use areas that have not been identified. For example, a single satellite track suggests the potential for MHI insular false killer whales to use habitat around the Northwestern Hawaiian Islands, where a separate false killer whale DPS tends to occur (Baird et al. 2012a). Other MHI insular false killer whales tagged off of Kauai circumnavigated Niihau and returned to the northwest side of the island of Kauai.

Photo identification studies also document that the animals regularly use both leeward and windward sides of the islands (Baird et al. 2005; Baird et al. 2012a; Baird et al. 2010; Forney et al. 2010; Oleson et al. 2010). Some individual false killer whales tagged off the island of Hawaii have remained around that island for extended periods (days to weeks), but individuals from all tagged groups eventually were found broadly distributed throughout the main Hawaiian Islands (Baird 2009; Forney et al. 2010). Individuals utilize habitat over varying water depths <50 m to >4000 m (Baird et al. 2010). It has been hypothesized that interisland movements may depend on the density and movement patterns of their prey species (Baird 2009). Evidence from both tags and individual-identifying photography suggests that the area between Kauai and Niihau near the Pacific Missile Range Facility (PMRF) is an area of range overlap between two or three stocks of false killer whales, one of which is the MHI insular stock. It appears that these waters may be at the far northwestern limit of that of the MHI insular stock and the southeastern limit of the Northwestern Hawaiian Islands stock and therefore it is a question how much the animals utilize this distal area near PMRF in proportion to the rest of their larger home range (DoN 2013). Two sightings made near PMRF during Navy monitoring were not from the MHI insular stock. The first was made by Navy monitoring observers between Niihau and Kaula of an adult male, adult female, and calf (DoN 2010), and when photographs were examined, the individuals did not match any identities in the MHI insular population catalog (Baird, pers. comm.). The second was in June 2012, when three individuals from a group of false killer whales were satellite-tagged. The tag data showed that the group subsequently left the area of PMRF, Niihau, and Kauaʻi after few days and travelled to Nihoa, Necker, French Frigate Shoals, and Gardner Pinnacles in the NWHI (Baird 2012). These two sightings comprise a sparse body of evidence that suggests the area between Kauai and Niihau is visited by false killer whales from populations other than the MHI insular distinct population segment.

4.3.7.2 ***Population Structure***

NMFS currently recognizes three stocks of false killer whale in Hawaiian waters under the MMPA: the Main Hawaiian Islands Insular, Hawaii pelagic, and the Northwestern Hawaiian Islands stock (Bradford et al. 2012a; Carretta et al. 2011) (77 FR 70915). NMFS considers all false killer whales found within 40 km (22 nm) of the Main Hawaiian Islands as belonging to the Insular stock and all false killer whales beyond 140 km (76 nm) as belonging to the Pelagic Stock (77 FR 70915). The animals belonging to the Northwest Hawaiian Islands stock are Insular to the Northwest Hawaiian Islands (Bradford et al. 2012), however, this stock was identified by animals encountered off Kauai. It has been previously recognized that the ranges for the two stocks (pelagic and Insular) overlap by 100 km (Bradford et al. 2012a; Carretta et al. 2011), but there is in addition, also overlap between all three stocks given these presently identified ranges.

The MHI insular false killer whale DPS is considered resident to the main Hawaiian Islands and is genetically and behaviorally distinct compared to other stocks (77 FR 70915). Genetic data suggest little immigration into the MHI insular false killer whale population (Baird et al. 2012a). However, because data on ecological relationships among false killer whale groups in the region are uncertain,additional data are being collected to identify whether other false killer whale groups in the Hawaiian Islands should also be considered part of the MHI insular false killer whale DPS (77 FR 70915).

4.3.7.3 ***Natural Threats***

Reduced genetic diversity may be a natural, but partially anthropogenically induced factor leading to MHI insular false killer whale decline (Wearmouth and Sims 2008). Only a single instance of predation on false killer whales has been documented, where killer whales attacked, killed, and consumed a false killer whale calf off New Zealand (Heithaus 2001; Visser et al. 2010). Parasitic infections have risen to levels thought to contribute to the deaths of some false killer whales, but these were from stranded individuals and it is unknown whether other health issues allowed for unhealthy levels of parasitism to develop (Andrade et al. 2001; Hernandez-

Garcia 2002; Morimitsu et al. 1987; Odell et al. 1980; Sedlak-Weinstein 1991; Stacey et al. 1994; Zylber et al. 2002).

Various parasites have been documented in tissues of false killer whales, including nematodes (e.g., *Anisakis simplex, Stenurus globicephalae*), trematodes (e.g., *Nasitrema globicephalae*), acanthocephalans (e.g., *Bolbosoma capitatum*), amphipods (e.g., *Isocyamus delphinii, Syncyamus aequus, Syncyamus pseudorcae*) and crustaceans (e.g., *Xenobalanus globicipitus*) (Andrade et al. 2001; Hernandez-Garcia 2002; Sedlak-Weinstein 1991; Stacey et al. 1994; Zylber et al. 2002). In some cases, parasitic infections have been implicated as contributing to false killer whale strandings. For example, following two mass stranding events that occurred along the coasts of Japan and Florida, tissues of false killer whales were examined for parasites (Morimitsu et al. 1987; Odell et al. 1980). In both studies, evidence of parasitic infections (e.g., *Stenurus globicephalus, Nasitrema gondo*) were noted in the lungs, pterygoid sinus complexes, and tympanic cavities of the whales, and the authors postulated that these infections most likely contributed to the demise of the animals. Currently, no information is available on parasitism in MHI insular false killer whales.

4.3.7.4 ***Anthropogenic Threats***

Several threats have been identified that may have or continue to lead to the decline of MHI insular false killer whales. These include incidental interactions with commercial and recreational fisheries and aquaculture facilities, prey availability, vessel traffic, anthropogenic noise, small populations effects, disease and predation, parasitism, environmental contaminants, harmful algal blooms, and ocean acidification and climate change (Oleson et al. 2010). False killer whales in Hawaiian waters have been seen to take catches from longline and trolling lines (Nitta and Henderson 1993; Shallenberger et al. 1981). Interactions with longline and troll fishery operations appear to result in disfigurement to dorsal fins, with roughly 4 percent of the population showing this injury, as well as entanglement and hooking (Baird and Gorgone 2005; Forney and Kobayashi. 2007; McCracken and Forney 2010; Nitta and Henderson 1993; Shallenberger et al. 1981; Zimmerman 1983). Carretta et al. (2009) estimated that 7.4 individuals per year are killed or seriously injured during the course of fishing operations in the Hawaiian EEZ. In this area, false killer whales are the most frequently hooked or entangled cetacean species, with most interactions occurring in tuna-targeting longline operations (Forney and Kobayashi. 2007; McCracken and Forney 2010). In total, 31 observations of serious injury or mortality have been documented from 1994 through 2008, which has led to an estimated 13 false killer whales killed or seriously injured throughout the Hawaiian longline fishery (Forney and Kobayashi. 2007; McCracken and Forney 2010). It is noteworthy that most interactions occurred well beyond the range known for MHI Insular False killer whales (0.6 were estimated to have been killed or seriously injured from 2003 to 2008) (McCracken and Forney 2010). In addition, false killer whales depredate on catches from shortline fisheries at least off northern Maui, with deliberate shootings occurring in some cases (Nitta and Henderson 1993; NMFS 2009c; Schlais 1985; TEC 2009). In January 2010, NMFS established a Take Reduction Team to address the incidental mortality and serious injury of false killer whales in the Hawaii-based deep-set and shallow-set longline fisheries. In November 2012, NMFS published the False Killer Whale Take Reduction Plan, which included regulatory measures and non-regulatory measures and recommendations to reduce mortalities and serious injuries of false killer whales in Hawaii-based longline fisheries (77 FR 71259).

Overfishing of some pelagic fishes, including bigeye and yellowfin tuna, may be adversely affecting MHI insular false killer whales. Catch weights for mahimahi have also declined since 1987 (NMFS 2009e). These changes may limit the prey quantity or quality available for false killer whales.

Bioaccumulation particularly of organic contaminants may be more of a concern for false killer whales than for many other cetaceans due to the high trophic level at which false killer whales feed. The only available study of false killer whale contaminant burden found PCBs and DDT present, with adult females carrying lower burdens than subadults or adult males (likely due to contaminants being unloaded into fetuses and milk during lactation) (Aguilar and Borrell 1994; Krahn et al. 2009; Ylitalo et al. 2009). PCB levels were high enough that biological effects would be experienced in other mammals (Kannan et al. 2000). Persistent organic pollutant levels are similar between false killer whales sampled in Taiwan and Japan, but smaller (some much smaller) than samples from British Columbia (Chou et al. 2004; Haraguchi et al. 2006; Ylitalo et al. 2009). Although these pollutants are believed to typically be sequestered in blubber, individuals undergoing metabolic stress mobilize fat tissue, resulting in pollutants being mobilized into other body tissues (Aguilar et al. 1999). False killer whales from Australia and Japan have been found to have relatively high body burdens of mercury, lead and cadmium (Endo et al. 2010; Kemper et al. 1994).

4.3.7.5 ***Status and Trends***

NMFS listed the MHI Insular population of false killer whales as an endangered distinct population segment (DPS) under the ESA (77 FR 70915) in 2012.

Reeves et al. (Reeves et al. 2009a) summarized information on false killer whale sightings near Hawaii between 1989 and 2007, based on various survey methods, and suggested that the MHI stock of false killer whales may have declined during the last two decades. More recently, Baird (Baird 2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the main Hawaiian Islands between 1994 and 2003 (Mobley Jr 2001; Mobley Jr. 2003; Mobley Jr. 2004; Mobley Jr. 2005). Sighting rates during these surveys exhibited a statistically significant decline that could not be attributed to any weather or methodological changes. Reanalysis of previously published abundance estimates for the Insular stock has led to them generally being discounted (77 FR 70915).

The minimum population estimate for the Hawaii Islands stock complex of false killer whales is the number of distinct individuals identified during the 2008 to 2011 photo-identification studies, which is 129 false killer whales (Carretta et al. 2014c). The current (year 2011) best estimated population size is 151 indivudals, although this estimate is likely positively biased (Carretta et al. 2014c). No data are available on current or maximum net productivity rate for this stock.

4.3.7.6 *Diving and Social Behavior*

Diving is not well-known in false killer whales, but individuals are believed capable of reaching 500 m in depth and possibly 700 m (Cummings and Fish 1971; Wearmouth and Sims 2008). However, most dives are significantly shallower. False killer whales occasionally dive to 150 m (apart from the possible 700 m dive), with frequent dives to 5 to 20 m during daytime and 30 to 40 m during nighttime, with durations for nighttime dives running 6 to 7 minutes (Wearmouth and Sims 2008). Some prey, such as mahimahi, occur most prevalently in the top 100 m of the water column, while others, such as tuna and swordfish, may occur down to several hundred meters (Boggs 1992; Carey and Robinson 1981).

False killer whale group sizes can vary widely. False killer whale groups can consist of multiple dispersed subgroups, and total group size may be underestimated if encounter duration is insufficient (Baird et al. 2008). Larger dispersed aggregations of false killer whales have been noted during surveys (Baird 2009; Carretta et al. 2007; Reeves et al. 2009b; Wade and Gerrodette 1993) that can move in a coordinated fashion (Baird et al. 2008). Main Hawaiian Island insular false killer whales form strong long-term bonds (Baird et al. 2008).

4.3.7.7 *Vocalization and Hearing*

Functional hearing in mid-frequency cetaceans, including MHI insular false killer whales, is conservatively estimated to be between approximately 150 Hz and 160 kHz (Southall et al. 2007c). There are three categories of sounds that odontocetes make. The first includes echolocation sounds of high intensity, high frequency, high repetition rate, and very short duration (Au et al. 2000). The second category of odontocete sounds is comprised of pulsed sounds. Burst pulses are generally very complex and fast, with frequency components sometimes above 100 kHz and average repetition rates of 300 per second (Yuen et al. 2007).

The final category of odontocete sounds is the narrowband, low frequency, tonal whistles (Au et al. 2000b; Caldwell et al. 1990). With most of their energy below 20 kHz, whistles have been observed with an extensive variety of frequency patterns, durations, and source levels, each of which can be repeated or combined into more complex phrases (Tyack and Clark 2000; Yuen et al. 2007).

In general, odontocetes produce sounds across the widest band of frequencies. Their social vocalizations range from a few hundreds of hertz (Hz) to tens of kilohertz (kHz) (Southall et al. 2007c) with source levels in the range of 100 to 170 dB re 1 µPa (see (Richardson et al. 1995c). They also generate specialized clicks used in echolocation at frequencies above 100 kHz that are used to detect, localize and characterize underwater objects such as prey (Au et al. 1993).

Echolocation clicks have source levels that can be as high as 229 dB re 1 µPa peak-to-peak (Au et al. 1974).

Nachtigall and Supin (2008) investigated the signals from an echolocating false killer whale and found that the majority of clicks had a single-lobed structure with peak energy between 20 and 80 kHz false rather than dual-lobed clicks, as has been demonstrated in the bottlenose dolphin. Navy researchers measured the hearing of a false killer whale and demonstrated the ability of this species to change its hearing during echolocation (Nachtigall and Supin. 2008). They found that there are at least three mechanisms of automatic gain control in odontocete echolocation, suggesting that echolocation and hearing are a very dynamic process (Nachtigall and Supin. 2008). For instance, false killer whales change the focus of the echolocation beam based on the difficulty of the task and the distance to the target. The echo from an outgoing signal can change by as much as 40 dB, but the departing and returning signal are the same strength entering the brain (Nachtigall and Supin. 2008) . The Navy demonstrated that with a warning signal, the false killer whale can adjust hearing by 15 dB prior to sound exposure (Nachtigall and Supin. 2008).

4.3.7.8 *Critical Habitat*

NMFS has not designated critical habitat for MHI insular false killer whale.

4.3.8 **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).

4.3.8.1 *Distribution*

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. 2009). 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). All Guadalupe fur seals represent a single population.

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. No Guadalupe fur seals have been sighted during 2009 through 2013 Navy-funded surveys in the HSTT Action Area.

4.3.8.2 *Population Structure*

All Guadalupe fur seals represent a single population.

4.3.8.3 *Natural Threats*

Although currently protected from commercial harvest, natural genetic factors are seen as a significant threat to the continued survival of this species. Because few individuals remained after commercial hunting, relatively low genetic diversity means that remaining individuals tend to be more susceptible to disease and inbreeding effects over subsequent generations (Bernardi et al. 1998; Weber et al. 2004). Sharks are known to prey upon Guadalupe fur seals, although mortality level is unknown (Gallo-Reynosa 1992).

4.3.8.4 *Anthropogenic Threats*

Due to small population size, this species is highly susceptible to extinction risk by relatively small mortalities. Guadalupe fur seals have been found stranded with fish hooks and other evidence of fishing gear interaction along the California coast (Hanni et al. 1997a).

4.3.8.5 *Status and Trends*

Guadalupe fur seals were listed as threatened under the ESA on December 16, 1985 (50 FR 51252). Guadalupe fur seals were hunted to near extinction by the late 1800s, with pre-harvest population estimates of 20,000 to 100,000 individuals. By 1897, the Guadalupe fur seal was believed to be extinct until a small population was found on Guadalupe Island in 1926. The most recent estimate is 7,408 animals in 1993 (Carretta et al. 2014c), with a population growth rate of 13.7 percent per year (Carretta et al. 2002). The estimated minimum population size for Guadalupe sur seals in Mexico is 3,028 individuals based on actual counts of hauled out seals (Carretta et al. 2014c). The number of individuals on the San Benito Islands appear to be increasing rapidly, with over 2,000 individuals counted in 2008 and is undergoing an exponential increase in population sizes, likely due to immigration from Guadalupe Island (Aurioles-Gamboa et al. 2010).

4.3.8.6 *Diving and Social Behavior*

The mean dive depth of Guadalupe fur seal lactating females is 55 feet, with a mean dive duration of 2.6 minutes. Mean surface interval between dives was two minutes. Dives were organized as outings lasting 2.5 hours. Foraging occurred during the night and transit during the day, with a maximum of 168 dives per day. Generally diving occurred at night, between eight in the evening and five in the morning (Croll et al. 1999b). Little is known about Guadalupe fur seal behavior during non-breeding season. They appear to spend long periods foraging at shallow depths during this time, but little information is known on their distribution at sea (Belcher and T.E. Lee 2002). Guadalupe fur seals are solitary, non-social animals.

4.3.8.7 *Vocalization and Hearing*

Pinnipeds produce sounds both in air and water that range in frequency from approximately 100 Hz to several tens of kHz and it is believed that these sounds serve social functions such as mother-pup recognition and reproduction. Source levels for pinniped vocalizations range from approximately 95 to 190 dB re 1 μPa (see Richardson et al. 1995).

Underwater hearing in otariid seals is adapted to low frequency sound and less auditory bandwidth than phocid seals. Hearing in otariid seals has been tested in two species present in the Action Area: California sea lion (Kastak and Schusterman 1998) and northern fur seal (Babushina et al. 1991; Moore and Schusterman 1987). Based on these studies, Guadalupe fur seals would be expected to hear sounds within the ranges of 50 Hz to 75 kHz in air and 50 Hz to 50 kHz in water. Schusterman et al. (2000) reviewed available evidence on the potential for pinnipeds to echolocate and indicated that pinnipeds have not developed specialized sound production or reception systems required for echolocation. Instead, it appears pinnipeds have developed alternative sensory systems (e.g., visual, tactile) to effectively forage, navigate and avoid predators underwater.

4.3.8.8 *Critical Habitat*

NMFS has not designated critical habitat for Guadalupe fur seals.

4.3.9 **Hawaiian Monk Seal**

The Hawaiian monk seal has a silvery-grey colored back with lighter creamy coloration on the underside; newborns are black. Additional light patches and red and green tinged coloration from attached algae are common. The back of the animals may become darker with age, especially in males. Adults generally range in size from 375 lbs to 450 lbs (170 to 205 kg); females are slightly larger than males; pups are 35 lbs (16 kg) at birth. Monk seals grow to 7.0 to 7.5 feet (2.1 to 2.3 m) in length with females being slightly larger than males; pups are 3 feet (1 m) at birth. The lifespan is estimated at 25 to 30 years.

4.3.9.1 *Distribution*

The Hawaiian monk seal is found primarily on the Northwestern Hawaiian Islands, especially Nihoa, Necker, French Frigate Shoals, Pearl and Hermes Reef, Kure Atoll, Laysan, and Lisianski. Sightings on the main Hawaiian Islands have become more common in the past 15 years and monk seals have been born on the Islands of Kauai, Molokai, Niihau, and Oahu (Carretta et al. 2005; Johanos and Baker. 2004; Kenyon 1981). Midway was an important breeding rookery, but is now used by a small number of monk seals (Reeves et al. 1992). Hawaiian monk seals breed primarily at Laysan Island, Lisianski Island, and Pearl and Hermes Reefs (Tomich 1986). Monk seals have been reported on at least three occasions at Johnston Island over the past 30 years (not counting nine adult males that were translocated there from Laysan Island in 1984).

During Navy-funded marine mammal surveys 2007 through 2012, there were 41 sightings of Hawaiian monk seals for a total of 58 individuals on (or near) Kauai, Kaula, Niihau, Oahu, and Molokai (HDR 2012b). Forty-seven (81 percent) individuals were seen during aerial surveys, and eleven (19 percent) during vessel surveys. Monk seals were most frequently observed at

Niihau. Fifty-two (88 percent) individual seals were observed hauled out, and six (10 percent) were in the water as deep as 800 m. In addition, seven seals were observed in August 2013, eleven in April 2013, and five in 2014 on t he ledges of Kaula Island (Normandeau Associates 2014).

The distribution, destinations, routes, food sources, and causes of monk seal movements when they are not traveling between islands are not well known (Johnson and Johnson 1979), but recent tagging studies have shown individuals sometimes travel between the breeding populations in the Northwest Hawaiian Islands. Based on one study, on average, 10 to 15 percent of the monk seals migrate among the northwestern Hawaiian Islands and the main Hawaiian Islands (Carretta et al. 2010b). Another source suggests that 35.6 percent of the main Hawaiian Island seals travel between islands throughout the year (Littnan 2011).

Navy-funded tagging studies in the main Hawaiian Islands demonstrate that mean foraging trip distance and duration, as well as maximum dive depth are similar between seals (Littnan and Wilson 2011). However, there were multiple outlying data points for all seals which varied by individual home ranges. Excluding one seal (R012) extended pelagic foraging trip, none of the seals traveled more than 300 km per trip and most traveled less than 50 km and remained within the 600 m depth contour near the MHI. The mean dive depth was 27.03 ± 44.97 m with a maximum of 529.4 m and a median depth of 14.4 m. The average dive duration was 5.006 ± 3.10 min with a median of 5.07 min with 28 percent of the time between dives spent at the surface. Although foraging trip distances and durations were similar among seals, there were high levels of individual variation in where the seals travelled (Wilson and D'Amico 2012). Based on the activity budgets obtained by the tags, more accurate information is now known for modeling and estimation purposes for the percentage of time seals spend in the water and what the likely proportion of the monk seal is in the water at any given time.

4.3.9.2 *Population Structure*

Hawaiian monk seals appear to exist as a single population that occurs in the Northwest Hawaiian Islands and main Hawaiian Islands. However, groups of individuals that occupy specific islands or atolls in the Hawaiian Archipelago are treated as sub-populations for the purposes of research and management activity.

Pearl and Hermes Reef, the Midway Islands, and Kure Atoll form the three westernmost sub-populations of Hawaiian monk seals. There is a higher degree of migration among these sub-populations than among the sub-populations that occupy Laysan, Lisianski and French Frigate Shoals, which are more isolated. As a result, population growth in the westernmost sub-populations can be influenced more by immigration than by intrinsic growth. Several recent cohorts (groups of individuals born in the same year) at all three sites indicate that survival of juveniles has declined.

4.3.9.3 *Natural Threats*

Monk seals are threatened by natural predation, disease outbreaks, biotoxins, and agonistic behavior by male monk seals (NMFS 2011c). Monk seals, particularly pups, are also subjected to extensive predation by sharks, which appear to be a particular problem for the monk seals occupying French Frigate Shoals in the Northwest Hawaiian Islands Islands (Antonelis et al. 2006). Monk seal remains have been found in the stomachs of both tiger and Galapagos sharks. Sharks predation has increased significantly in the Northwest Hawaiian Islands, particularly French Frigate Shoals. Over the past decade, there have been several attempts to combat or mitigate the effects of shark predation on Hawaiian monk seals. From 2000 through 2003, sharks were removed (through hazing or targeted fishing) at Trig Island, which was followed by declines in the number of monk seal pups killed at the island. These effects were only successful temporarily and, in 2002 and 2003, hazing was discontinued because it made the sharks wary and difficult to catch.

Hawaiian monk seals appear to be threatened by the spread of infectious diseases, including leptospirosis, toxoplasmosis, and West Nile virus, although domestic animals and humans may be vectors for these diseases (which would make them anthropogenic rather than natural threats). The absence of antibodies to these diseases in monk seals would make them extremely vulnerable to infection. Biotoxins such as ciguatera can cause mortality in phocids, but its role in mortality of monk seals was implicated and not confirmed, remaining unclear due to the lack of assays for testing tissues and the lack of epidemiological data on the distribution of toxin in monk seal prey.

The primary cause of adult female mortality affecting the recovery in the monk seal population during the 1980s and early 1990s was injury and death of female monk seals caused by “mobbing” attacks initiated by male monk seals. Although NMFS has developed and implemented measures to mitigate the effects of mobbing attacks, they are still considered a serious threat to Hawaiian monk seals. In recent years, low juvenile survival, in part due to food limitation, has been evident at all subpopulations of Hawaiian monk seals in the Northwest Hawaiian Islands. Nevertheless, the death of adult and immature females that resulted from this behavior would reduce the total number of breeding females and the recruitment of immature females into the adult, breeding population. Fewer breeding adults would produce fewer pups which, by itself, would increase the population’s rate of decline; when coupled with reductions in the survival probability of pups, it would create a feedback loop that would tend to cause the population to decline.

There have been several attempts to balance sex ratios at Laysan Island by removing problem males. In 1984, a group of ten adult males that had been observed attacking females, or whose behavior profile was similar to those that attacked females, were captured on Laysan and transported to Johnston Atoll. One of the ten died prior to release, and of the remaining nine, most were not seen after a few months. The last male was not observed until after a period of 16 months. Another group of five problem males was removed from Laysan and entered into captivity in 1987 for studies identified in the plan. Males in the 1987 group were used to define the testosterone cycle in males and to evaluate a drug to suppress testosterone for possible field application to reduce aggressive behavior. The captive trials proved effective at suppressing testosterone levels in the male seals (Atkinson et al. 1993) and a pilot field trial was performed (Atkinson et al. 1998). However, severe limitations in this approach (each male had to be captured and injected a number of times over the course of the breeding season; these repeated captures would have resulted in extensive disturbance to most seals on the island during the breeding season) caused it to be terminated.

One of the most substantial threats to Hawaiian monk seals results from dramatic declines in the survival of juveniles and appears to be related to significantly reduced body sizes in pup and juvenile seals. These declines in body size appear to be evidence of chronic or episodic limitations in available prey. In recent years, low juvenile survival, in part due to food limitation, has been evident at all subpopulations of monk seals in the Northwest Hawaiian Islands. The mean age-specific birth rates of adult female Hawaiian monk seals, which are low relative to other phocid seals, could also be evidence of food limitation (NMFS 2007a).

4.3.9.4 ***Anthropogenic Threats***

Several human activities are known to threaten Hawaiian monk seals: commercial and subsistence hunting, intentional harassment, competition with commercial fisheries, entanglement in fishing gear, habitat destruction on breeding beaches, pollution, and unintentional human disturbance (Kenyon 1981; Reeves et al. 1992; Riedman 1990). The revised recovery plan for Hawaiian monk seals identifies food limitation, entanglements, and shark predation as crucial threats to the continued existence of this species (NMFS 2007a).

Entangled monk seals were first observed in 1974 (Henderson 1984). Historically, monk seals have become entangled in net, line (including monofilament nylon line), net and line combinations, straps, rings (including hagfish or eel traps), and other random items such as discarded lifejackets, buckets (portion of rims), bicycle tires, rubber hoses, etc. (Henderson 1990b). Monk seal pups (including newly weaned pups) are entangled at higher rates than other age classes (Henderson 1985; Henderson 1990b; Henderson 2001). Between 1982 and 1988, pups comprised 11 percent of the population, but represent about 42 percent of observed entanglements (for comparison, adults represented about 49 percent of the population but only 16 percent of entanglements)(Henderson 1990b). Collectively immature monk seals were involved in almost 80 percent of all observed entanglements, even though they represented only 46 percent of the population (Henderson 2001).

Between 1982 and 2006, a total of 268 entanglements of monk seals were documented, including 118 in fishing gear. There were 57 serious injuries (including 32 from fishing gear) and eight mortalities (including seven from fishery items). From 1982 to 2000, there was an estimated minimum rate of 2.3 serious injuries or deaths per year attributable to fishery related marine debris (NMFS 2007a). In recent years, there have been several shooting deaths of monk seals in the MHI (Carretta et al. 2013a) ("Who Would Kill a Monk Seal" NY Times Magazine, May 8, 2013). There is also a multiagency marine debris working group that was established in 1998 to remove derelict fishing gear, which has been identified as a top threat to this species, from the Northwestern Hawaiian Islands (Donohue and Foley 2007). Agencies involved in these efforts include The Ocean Conservancy, the City and County of Honolulu, the Coast Guard, the Fish and Wildlife Service, the Hawaii Wildlife Fund, the Hawaii Sea Grant Program, the National Fish and Wildlife Foundation, the Navy, the University of Alaska Marine Advisory Program, and numerous other state and private agencies and groups (MMC 2002).

4.3.9.5 ***Status and Trends***

Hawaiian monk seals were listed as endangered under the Endangered Species Act of 1973 on November 23, 1976 (41 FR 51611). A 5-year status review completed in 2007 recommended retaining monk seals as an endangered species (72 FR 46966, August 22, 2007). Critical habitat was originally designated for Hawaiian monk seals on April 30, 1986 (51 FR 16047), and was extended on May 26, 1988 (53 FR 18988; CFR 226.201).

Monk seals are considered one of the most endangered groups of pinnipeds on the planet because all of their populations are either extinct (for example, the Caribbean monk seal) or exist at numbers that are precariously close to extinction (Mediterranean and Hawaiian monk seals). Two periods of decline have been reported for Hawaiian monk seals. The first decline occurred in the 1800s when sealers, crews of wrecked vessels, and guano and feather hunters nearly hunted the population to extinction (Dill and Bryan 1912; Kenyon and Rice 1959). Following the collapse of this population, expeditions to the Northwest Hawaiian Islands reported increasing numbers of seals (Bailey 1952). A survey in 1958 suggested that the population had partially recovered from its initial collapse. In June 2006, the Papahanaumokuakea Marine National Monument (71 FR 36443, June 26, 2006) was established in the Northwest Hawaiian Islands. The boundary of the Monument includes about 140,000 square miles of emergent and submerged lands and waters of the northwest Hawaiian Islands, and some activities such as fishing that pose risks to the marine habitat of Hawaiian monk seals are regulated.

Consistent declines in the monk seal population trends have been recorded since range-wide surveys began in the late 1950s (survey results that were reported by Kenyon and Rice (1959) and Rice (1960). Rice (1960) conducted additional counts at Midway Islands in 1956 and 1958 and Wirtz (1968) conducted counts at Kure Atoll in 1963 to 1965. Between the late 1950s and 1980s, counts at the atolls, islands, and reefs in the Northwest Hawaiian Islands suggested a 50 percent decline in this population (Johnson et al. 1982). The total population for the five major breeding locations plus Necker Island for 1987 was estimated to be 1,718 seals including 202 pups of the year (Gilmartin 1988). This compares with 1,488 animals estimated for 1983 (Gerrodette 1985). In 1992 the Hawaiian monk seal population was estimated to be 1,580 (standard error = 147) (Ragen 1993). The best estimate of total abundance for 1993 was 1,406 (standard error = 131, assuming a constant coefficient of variation).

Beach counts of juveniles, sub-adults, and adults declined by about 5 percent per year from 1985 through 1993, and then became relatively stable until the current decline began in 2001 (NMFS 2007a). Between 1958 and 1993, mean beach counts declined by 60 percent and included declines in the number of monk seals at French Frigate Shoals, which once accounted for more than 50 percent of the total non-pup beach counts among the six primary Northwest Hawaiian Island sub-populations. Between the years 1958 and 2006, beach counts of juveniles, sub-adults and adults declined by 66 percent; the total abundance of monk seals at the six primary subpopulations in the Northwest Hawaiian Islands is declining at an annual rate of 3.9 percent (95 percent CI = -4.8 to -3.0 percent)(NMFS 2007a).

Sightings and births are increasing in the MHI, although systematic surveys were not conducted before 2000, and counts do not represent total abundance, as they do not account for seals in the water, and not every seal on land is detected. In 2000, the count in the MHI was 45 seals, and in 2001, 52 were counted. In 2005, the total number of unique seals identified was 77, based on non-systematic sightings. Annual births have increased since the mid-1990s. Although this could be a positive indication for the survival of the species, the increased chance of contraction of diseases such as leptospirosis and toxoplasmosis from wild and domestic animals, and increased interactions with humans, including fishermen, boaters, and divers raise conservation concerns which do not apply to the NWHI (NMFS 2007b). The only available estimate of abundance in the MHI is 152 individuals, with an annual population growth rate of 7 percent (Baker et al. 2011). Survival to one year of age is 77 percent in the MHI, versus 42 to 57 percent in the NWHI (Baker et al. 2011).

The best published estimate of the total Hawaiian monk seal population size is 1,209 individuals (Carretta et al. 2014). A more recent unpublished stock assessment report estimates the total population of Hawaiian monk seals to be 1,153 individuals, with an estimated minimum population size of 1,118 individuals (Carretta et al. 2015, unpublished 2014 stock assessment reports). Population dynamics at the different locations in the northwestern Hawaiian Islands and the main Hawaiian Islands has varied considerably (Antonelis et al. 2006). The overall trend has been a steady decline, with the total number of Hawaiian monk seals decreasing from a 2007 estimate of 1,146 individuals (Littnan 2011). In the northwestern Hawaiian Islands, where most seals reside, the decline in abundance is approximately 4 percent per year. While this decline has been occurring in the northwestern Hawaiian Islands, the number of documented sightings and annual births in the main Hawaiian Islands (where Navy training and testing occurs) has increased since the mid-1990s (Baker and Johanos 2004). In the main Hawaiian Islands, a minimum abundance of 45 seals was found in 2000, and this increased to 52 in 2001 (Baker and Johanos 2004). In 2009, 113 individual seals were identified in the main Hawaiian Islands based on flipper tag ID numbers or unique natural markings. The total number in the main Hawaiian

Islands is estimated to be around 138 animals (Baker and Johanos 2004; Carretta et al. 2013a)(Carretta et al. 2015, unpublished 2014 stock assessment reports).

Hawaiian monk seal pups weaned in the MHI exhibit higher girths and lengths compared to pups from the NWHI, as a result of pre- and post-partum maternal investment—a partial reflection of prey availability—contradicting the studies that indicate better foraging conditions in the NWHI. Suggested explanations for this include a higher per capita availability of prey in the MHI, similar absolute preferred prey densities when apex predators are not included in the biomass, and increased prey availability due to reduced competition from apex predators (Baker and Johanos 2004).

Based on the evidence available, Hawaiian monk seals exist as a population that is subject to the dynamics of “small” populations. That is, they 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. For example, Hawaiian monk seals have very low juvenile and sub-adult survival rates (due to starvation which is believed to be caused by limitations in the food base), low juvenile survival has led to low juvenile recruitment into the adult population, and the adult population increasingly consists of ageing females whose reproductive success is expected to decline (if it has not already declined) in the reasonably foreseeable future. A positive feedback loop between reduced reproductive success of adult females and reduced recruitment into the adult population (which reduces the number of adult females) is the kind of demographic pattern that is likely to increase the monk seal’s decline toward extinction. As a result, we assume that Hawaiian monk seals have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities (primarily reductions in prey base due to competition with commercial and subsistence hunting, entanglement in fishing gear, and habitat destruction), natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate), and endogenous threats caused by the small size of their population.

4.3.9.6 ***Diving and Social Behavior***

Several recent studies of the foraging patterns of Hawaiian monk seals near rookeries in the northwestern Hawaiian Islands provide insight into their diving behavior. Monk seals spend most of their time at sea in nearshore, shallow marine habitats (Littnan et al. 2007; Littnan and Stewart. 2007). Dive depths appear to differ slightly between rookeries as well as between age classes and genders. At Pearl and Hermes Reef, most dives were from 8 to 40 m with a second much smaller node at 100 to 120 m (Stewart 2004; Stewart et al. 2006).

At Kure Atoll, most dives were shallower than 40 m, with males tending to dive deeper than females (Stewart and Yochem. 2004). At Laysan Island, a similar dive pattern was recorded with most dives shallower than 40 m, but at that location females tended to dive deeper than males (250 to 350 m) (Stewart and Yochem 2004). Parrish et al. (2002) noted a tendency towards night diving at French Frigate Shoals, with dives to ~80 to 90 m. Based on these data, the following are rough order estimates of time at depth: 90 percent at 0 to 40 m; 9 percent at 40 to 120 m; 1 percent at >120 m. In Hawaii, overall results showed most foraging trips to last from a few days to 1 to 2 weeks, with seals remaining within the 650 ft. (200 m) isobaths surrounding the main Hawaiian Islands and nearby banks (Littnan et al. 2007).

4.3.9.7 *Vocalizations and Hearing*

The information on the hearing capabilities of endangered Hawaiian monk seals is somewhat limited, but they appear to have their most sensitive hearing at 12 kHz to 28 kHz. Below 8 kHz, their hearing is less sensitive than that of other pinnipeds. Their sensitivity to high frequency sound drops off sharply above 30 kHz (Richardson et al. 1995a; Richardson et al. 1995c; Thomas et al. 1990b). An underwater audiogram for Hawaiian monk seal, based on a single animal whose hearing may have been affected by disease or age, was best at 12 kHz to 28 kHz and 60 kHz to 70 kHz (Thomas et al. 1990b). The hearing showed relatively poor hearing sensitivity, as well as a narrow range of best sensitivity and a relatively low upper frequency limit (Thomas et al. 1990b). Schusterman et al. (2000) reviewed available evidence on the potential for pinnipeds to echolocate and indicated that pinnipeds have not developed specialized sound production or reception systems required for echolocation. Instead, it appears pinnipeds have developed alternative sensory systems (e.g., visual, tactile) to effectively forage, navigate and avoid predators underwater.

4.3.9.8 *Critical habitat*

Critical habitat was originally designated on April 30, 1986 (51 FR 16047), and was extended on May 26, 1988 (53 FR 18988; CFR 226.201). The critical habitat includes all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms (37 m) around the following: Kure Atoll (28°24′ N, 178°20′ W), Midway Islands, except Sand Island and its harbor (28°14′ N, 177°22′ W), Pearl and Hermes Reef (27°55′ N, 175° W), Lisianski Island (26°46′ N, 173°58′ W), Laysan Island (25°46′ N, 171°44′ W), Maro Reef (25°25′ N, 170°35′ W), Gardner Pinnacles (25°00′ N, 168°00′ W), French Frigate Shoals (23°45′ N, 166°00′ W), Necker Island (23°34′ N, 164°42′ W), Nihoa Island (23°03.5′ N, 161°55.5′ W). The marine component of this habitat was designated primarily as feeding areas for Hawaiian monk seals, while terrestrial habitat serves as pupping and nursing habitat for mothers and pups. Both components are currently under significant degradation pressure. Because the marine critical habitat is in the Papahānaumokuākea Marine National Monument, fishing is forbidden in the critical habitat. A result of this is the establishment of large predatory fishes (sharks and jacks) that compete with Hawaiian monk seals for prey resources. This may be a cause of seal malnourishment seen throughout many islands in the region.

On June 2, 2011, NMFS proposed to extend critical habitat in the NWHI to Sand Island (Midway) and ocean waters out to a depth of 500 (76 FR 27988). The area around the MHI (Kaula Island, Niihau, Kauai, Oahu, Maui Nui, and Hawaii) to a depth of 500 m and inland to a distance of 5 m from shore was also proposed. NMFS proposed national security exclusions for some Navy training areas including Puuloa Training Range and the Naval Defensive Sea Area.

4.3.10 Green Sea Turtle

Green sea turtles (*Chelonia mydas*) are the largest of all the hard-shelled sea turtles, but have a comparatively small head. While hatchlings are just 2 inches (50 mm) long, adults can grow to more than 3 feet (0.91 m) long and weigh 300 to 350 pounds (136 to 159 kg).

Adult green turtles are unique among sea turtles in that they are herbivorous, feeding primarily on sea grasses and algae. This diet is thought to give them greenish colored fat, from which they take their name. A green turtle's carapace (top shell) is smooth and can be shades of black, gray, green, brown, and yellow. Their plastron (bottom shell) is yellowish white.

Scientists estimate green turtles reach sexual maturity anywhere between 20 and 50 years, at which time females begin returning to their natal beaches (i.e., the same beaches where they were born) every two to four years to lay eggs.

The only nesting population in the Action Area is in Hawaii, with approximately 3,846 breeding females nesting and with approximately 3,710 of those nesting annually at French Frigate Shoals (NMFS 2015). Nesting in the Hawiian Islands occurs from May to September, peaking in early June, with turtles laying an average of two but up to six nests per season with a mean of 104 eggs per clutch (Balazs 1979). Four other populations are located in the eastern Pacific Ocean, south of the Action Area, with nesting occurring along the western Mexico coast, as well as within the Gulf of California (NMFS and USFWS 2007a).

4.3.10.1 *Distribution*

Green turtles are found in the Pacific Ocean, Atlantic Ocean, Indian Ocean, Carribean Sea, and Mediterranean Sea, primarily in tropical or, to a lesser extent, subtropical waters. These regions can be further divided into nesting aggregations within the eastern, central, and western Pacific Ocean; the western, northern, and eastern Indian Ocean; Mediterranean Sea; and eastern, southern, and western Atlantic Ocean, including the Carribean Sea.

Green turtles appear to prefer waters that usually remain around 20 °C in the coldest month. During warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution. Stinson (1984a) found green turtles to appear most frequently in U.S. coastal waters with temperatures exceeding 18 °C. Further, green turtles seem to occur preferentially in drift lines or surface current convergences, probably because of the prevalence of cover and higher densities of their food items associated with these oceanic phenomena. Underwater resting sites include coral recesses, the underside of ledges, and sand bottom areas that are relatively free of strong currents and disturbance from natural predators and humans. Available information indicates that green turtle resting areas are in proximity to their feeding areas (NMFS and USFWS 1998b).

The green turtle is the most common sea turtle species in the Hawaii region of the Action Area, occurring in the coastal waters of the main Hawaiian Islands throughout the year and commonly migrating seasonally to the Northwestern Hawaiian Islands to reproduce. Green sea turtles are found in inshore waters around all of the main Hawaiian Islands and Nihoa Island, where reefs, their preferred habitats for feeding and resting, are most abundant. They are also common in an oceanic zone surrounding the Hawaiian Islands. This area is frequently inhabited by adults migrating to the Northwestern Hawaiian Islands to reproduce during the summer and by ocean-dwelling individuals that have yet to settle into coastal feeding grounds of the main Hawaiian Islands. Farther offshore, green turtles occur in much lower numbers and densities.

Green sea turtles have been sighted in Pearl Harbor, but do not nest in the harbor; they are routinely seen in the outer reaches of the entrance channel (Navy 2012a). The number of resident turtles at the entrance channel is estimated at 30 to 40, with the largest number occurring at Tripod Reef and the Outfall Extension Pipe. They are also found beneath the outfall pipe of the Fort Kamehameha wastewater treatment plant, at depths of approximately 65 ft. (20 m) (Smith 2010). Green sea turtles are also regularly seen in West Loch (Smith et al. 2006). In the spring of 2010, two green turtles nested at Pacific Missile Range Facility for the first time in more than a decade, with successful hatching in August 2010 (O'Malley 2010, 2011, 2012). Green sea turtles are also common at all three landing beaches of U.S. Marine Corps Base Hawaii in Kaneohe Bay, where they forage in the shallow water seagrass beds.

The only nesting population in the Action Area is in Hawaii, with approximately 3,846 breeding females nesting and with approximately 3,710 of those nesting annually at French Frigate Shoals (NMFS 2015). A large foraging population resides in and returns to the shallow waters surrounding the main Hawaiian Islands (especially around Maui and Kauai), where they are known to come ashore at several locations on all eight of the main Hawaiian Islands for basking or nesting.

Green sea turtles are widely distributed in the subtropical coastal waters of southern Baja California, Mexico, and Central America, several hundred kilometers (km) south of the Action Area (NMFS and USFWS 1998b). The main group of eastern Pacific Ocean green turtles is found on the breeding grounds of Michoacán, Mexico, from August through January and year-round in the feeding areas, such as those on the western coast of Baja California, along the coast of Oaxaca, and in the Gulf of California (the Sea of Cortez) (NMFS and USFWS 1998b). Bahía de Los Angeles in the Gulf of California has been identified as an important foraging area for green turtles (Seminoff et al. 2003). Eastern Pacific Ocean green turtles have been reported as far north as British Columbia (48.15° N) (Eckert 1993b; NMFS and USFWS 1998b). The western coasts of Central America, Mexico, and the United States constitute a shared habitat for this population (NMFS and USFWS 1998b). The green turtle is not known to nest on Southern California beaches.

In general, turtle sightings in the eastern Pacific Ocean increase during summer as warm water moves northward along the coast (NMFS and USFWS 1998b). Sightings may also be more numerous in warmer years compared to colder years. In waters south of Point Conception, Stinson (Stinson 1984a) found this seasonal sighting pattern to be independent of interyear temperature fluctuations. More sightings occurred during warmer years north of Point Conception. Stinson also reported that more than 60 percent of eastern Pacific Ocean green turtles observed in California were in areas where the water was less than 165 ft. (50 m) deep, often observed along shore in areas of eelgrass.

San Diego Bay is home to a resident sub-population of green turtles (Dutton and McDonald 1990; Stinson 1984a). A 20-year monitoring program of these turtles indicates an annual abundance of between 16 and 61 turtles (Eguchi et al. 2010). Eelgrass beds and marine algae are particularly abundant in the southern half of the bay, and green turtles are frequently observed foraging on these items (Dutton et al. 2002). Until December 2010, the southern part of San Diego Bay was warmed by the effluent from the Duke Energy power plant, a fossil fuel power generation facility in operation since 1960. Green sea turtles are known to congregate in this area. The closure of the power plant may impact these resident turtles and alter movement patterns. Ultrasonic tracking studies have shown that green turtles in southern San Diego Bay have relatively small home ranges (Dutton et al. 2002). Between 2009 and 2011, MacDonald et al. (MacDonald et al. 2012) used acoustic telemetry to track 25 green turtles in San Diego Bay. The results of the study suggest that resident turtles likely do not spend much, if any, time foraging in central or northern San Diego Bay, where human activities are greatest (including Navy activities). A few sea turtles have been observed in northern San Diego Bay, but these are likely transient green turtles that enter the bay in warmer months (MacDonald et al. 2012). Another green turtle population resides in Long Beach, California, although less is known about this population (Eguchi et al. 2010).

Ocean waters off Southern California and northern Baja California are also designated as areas of occurrence because of the presence of rocky ridges and channels and floating kelp habitats suitable for green turtle foraging and resting (Stinson 1984a); however, these waters are often at temperatures below the thermal preferences of this primarily tropical species.

4.3.10.2 *Population Structure*

The population dynamics of green turtles and all of the other sea turtles we consider in this Opinion are usually described based on the distribution and habit of nesting females, rather than their male counterparts. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we describe sea turtle populations based on the nesting beaches that female sea turtles return to when they mature. Because the patterns of increase or decrease in the abundance of sea turtle nests over time are determined by internal dynamics rather than external dynamics, we make inferences about the growth or decline of sea turtle populations based on the status and trend of their nests.

Primary nesting aggregations of green turtles (i.e. sites with greater than 500 nesting females per year) include: Ascension Island (south Atlantic Ocean), Australia, Brazil, Comoros Islands, Costa Rica, Ecuador (Galapagos Archipelago), Equatorial Guinea (Bioko Island), Guinea-Gissau (Bijagos Archipelago), Iles Eparses Islands (Tromelin Island, Europa Island), Indonesia, Malaysia, Myanmar, Oman, Philippines, Saudi Arabia, Seychelles Islands, Suriname, and United States (Florida; NMFS and USFWS 1998c; Seminoff et al. 2002).

Smaller nesting aggregations include: Angola, Bangladesh, Bikar Atoll, Brazil, Chagos Archipelago, China, Costa Rica, Cuba, Cyprus, Democratic Republic of Yemen, Dominican Republic, d’Entrecasteaux Reef, French Guiana, Ghana, Guyana, India, Iran, Japan, Kenya, Madagascar, Maldives Islands, Mayotte Archipelago, Mexico, Micronesia, Pakistan, Palmerston Atoll, Papua New Guinea, Primieras Islands, Sao Tome é Principe, Sierra Leone, Solomon Islands, Somalia, Sri Lanka, Taiwan, Tanzania, Thailand, Turkey, Scilly Atoll, United States (Hawaii), Venezuela, and Vietnam.

Molecular genetic techniques have helped researchers gain insight into the distribution and ecology of migrating and nesting green turtles. In the Pacific Ocean, green turtles group into two distinct regional clades: (1) western Pacific and South Pacific islands, and (2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawaii. In the eastern Pacific, greens forage coastally from San Diego Bay, California in the north to Mejillones, Chile in the South. Based on mtDNA analyses, green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton et al. 2003).

4.3.10.3 *Natural Threats*

The various habitat types green turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which green 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 including herons, gulls, dogfish, and sharks. Larger green turtles, including adults, are also killed by sharks and other large, marine predators.

Green turtles in the Pacific are afflicted with a tumor disease, fibropapillomatosis, which is of an unknown etiology and often fatal, as well as spirochidiasis, both of which are the major causes of strandings of this species (Chaloupka et al. 2008b; Reséndiz et al. 2012; Work and Balazs 1999; Work et al. 2009). The presence of fibropapillomatosis among stranded turtles has increased significantly over the past 17 years, ranging from 47 to 69 percent during the past decade (Murakawa et al. 2000). Preliminary evidence suggests an association between the distribution of fibropapillomatosis in the Hawaiian Islands and the distribution of toxic benthic dinoflagellates (*Prorocentrum* spp.) known to produce a tumor promoter, okadaic acid (Landsberg et al. 1999).

4.3.10.4 *Anthropogenic Threats*

Three human activities are known to threaten green turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of green turtles populations were the number of eggs and adults captured and killed on nesting beaches in combination with the number of juveniles and adults captured and killed in coastal feeding areas. Some populations of green turtles still lose a large number of eggs, juveniles, and adults to subsistence hunters, local communities that have a tradition of harvesting sea turtles, and poachers in search of turtle eggs and meat.

Directed harvests of eggs and other life stages of green turtles were identified as a “major problem” in American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated States of Micronesia, Republic of the Marshall Islands, and the Unincorporated Islands (Wake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway). Green sea turtles are also captured and killed in commercial fisheries. Gillnets account for the highest number of green turtles that are captured and killed, but they are also captured and killed in trawls, traps and pots, longlines, and dredges.

Green sea turtles are also threatened by domestic or domesticated animals which 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 2010). 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 2010). 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 2010). 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 2010).

From 1985 to 2012 the likelihood of green sea turtles ingesting oceanic debris increased from 30 percent to nearly 50 percent (Schuyler et al. 2013). Ingestion of debris occurred primarily in pelagic areas and floating plastics was the most commonly ingested material. These results suggest oceanic juveniles may be more susceptible to ingesting oceanic debris than older benthic-feeding individuals (Schuyler et al. 2013).

4.3.10.5 *Status and Trends*

Green turtles are listed as threatened under the ESA, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. The green sea turtles that use waters off southern California for feeding originate from the endangered breeding population along the Pacific coast of Mexico, while turtles in Hawaii are part of the rangewide threatened population. Using a precautionary approach, Seminoff (2002) estimates that the global green turtle population has declined by 34 percent to 58 percent over the last three generations (approximately 150 years). Causes for this decline include harvest of eggs, subadults, and adults; incidental capture by fisheries; loss of habitat; and disease.

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, as a direct consequence of a historical combination of overexploitation and habitat loss (Eckert 1993a; Seminoff 2004). The East Island nesting beach in Hawaii is showing a 4.8 percent annual growth rate over the last 40 years (NMFS 2015). In the Eastern Pacific, mitochondrial DNA analysis has indicated three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a) and available evidence suggests an increasing population trend at the Michoacán nesting site (NMFS 2015). However, historically, >20,000 females per year are believed to have nested in Michoacán alone (Clifton et al. 1982; NMFS and USFWS 2007a). Thus, the current number of nesting females is still far below historical levels. The species is also thought to be declining in the Atlantic Ocean. However, like several of the species we have already discussed, the information available on the status and trend of green turtles do not allow us to make a definitive statement about the global extinction risks facing these sea turtles or risks facing particular populations (nesting aggregations) of these turtles. With the limited data available on green turtles, we do not know whether green turtles 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 green turtles are threatened more by exogenous threats such as anthropogenic activities (entanglement, habitat loss, overharvests, etc.) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate).

A recovery plan for the U.S Population of Atlantic Green Turtles was written in 1991 (NMFS and USFWS 1991a). A recovery plan for the U.S. Pacific Populations of the Green Turtle was written in 1998 (NMFS and USFWS 1998c).

4.3.10.6 *Diving and Social Behavior*

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS and USFWS 1998c). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson 1967b; Lutcavage and Lutz 1997), while subadults routinely dive 20 meters for 9 to 23 minutes, with a maximum recorded dive of 66 minutes (Brill et al. 1995 in Lutcavage and Lutz 1997).

4.3.10.7 *Vocalizations and Hearing*

Sea turtles do not appear to use sound for communication, and there are no published recordings of green turtle vocalizations. Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 Hz to 2,000 Hz, with a range of maximum sensitivity between 100 Hz and 800 Hz (Bartol et al. 1999c; Lenhardt 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Several studies have been conducted to measure green turtle hearing sensitivity, each using a slightly different methodology. Ridgway et al. (1969) studied the auditory evoked potentials of three green turtles (in air and through direct mechanical stimulation of the inner ear) and concluded that their maximum sensitivity occurred from 300 Hz to 400 Hz with rapid declines for tones at lower and higher frequencies. They reported an upper limit for cochlear potentials without injury of 2000 Hz and a practical limit of about 1000 Hz.

Bartol and Ketten (2006) measured auditory brainstem responses (short latency auditory evoked potentials) to aerial tones in partially submerged green turtles and documented hearing between 100 Hz and 800 Hz, with maximum sensitivity between 600 Hz and 700 Hz in Atlantic juvenile greens, and100 Hz and 500 Hz with maximum sensitivity between 200 Hz and 400 Hz in Pacific subadult greens (Moein Bartol and Ketten 2006).

Dow Piniak et al. (2012b) recorded auditory evoked potential in response to both aerial and underwater acoustic stimuli. Green turtles detected acoustic stimuli in both media, responding to underwater signals between 50 and 1,600 Hz (turtles completely submerged) and aerial signals between 50 and 800 Hz, with maximum sensitivity between 200 and 400 Hz underwater and 300 and 400 Hz in air (Piniak et al. 2012b). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a).

4.3.10.8 *Critical Habitat*

On September 2, 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey. This critical habitat does not occur within the HSTT Action Area.

4.3.11 Hawksbill Turtle

The hawksbill turtle (*Eretmochelys imbricata*) is a small to medium-sized sea turtle; adults typically range between 65 and 90 cm (26 to 35 in) in carapace length and weigh around 80 kg (176 lb) (Witzell 1983). Hawksbills are distinguished from other sea turtles by their hawk-like beaks, posteriorly overlapping carapace scutes, and two pairs of claws on their flippers (NMFS and USFWS 1993). The carapace of this species is often brown or amber with irregularly radiating streaks of yellow, orange, black, and reddish-brown.

4.3.11.1 *Distribution*

Hawksbill sea turtles occur in tropical and subtropical seas of the Atlantic, Pacific and Indian Oceans. Hawksbill sea turtles occupy different habitats depending on their life history stage. After entering the sea, hawksbill turtles occupy pelagic waters and occupy weed-lines that accumulate at convergence points. When they grow to about 20 to 25 cm carapace length, hawksbill turtles re-enter coastal waters where they inhabit and forage in coral reefs as juveniles, sub-adults and adults. Hawksbill sea turtles also occur around rocky outcrops and high energy shoals, where sponges grow and provide forage, and they are known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent.

Hawksbills are considered the most coastal of the sea turtles that inhabit the Action Area, with juveniles and adults preferring coral reef habitats (National Marine Fisheries Service 2010b). Reefs provide shelter for resting hawksbills day and night, and they are known to visit the same resting spot repeatedly. Hawksbills are also found around rocky outcrops and high-energy shoals—optimum sites for sponge growth—as well as in mangrove-lined bays and estuaries (NMFS and USFWS 2013).

Hatchling and early juvenile hawksbills have also been found in the open ocean, in floating mats of seaweed (Musick and Limpus 1997). Although information about foraging areas is largely unavailable due to research limitations, juvenile and adult hawksbills may also be present in open ocean environments (NMFS and USFWS 2007a). Very little is known about the open ocean habitat and distribution of hawksbills in the Transit Corridor.

Hawksbills are mostly found in the coastal waters of the eight main islands of the Hawaiian Island chain. Stranded or injured hawksbills are occasionally found in the Northwestern Hawaiian Islands (Parker et al. 2009). Hawksbills are the second-most-common species in the offshore waters of the Hawaiian Islands, yet they are far less abundant than green turtles (Chaloupka et al. 2008b). The lack of hawksbill sightings during aerial and shipboard surveys likely reflects the species' small size and difficulty in identifying them from a distance.

Hawksbills have been captured in Kiholo Bay and Kau (Hawaii), Palaau (Moloka'i), and Makaha (Oahu). Strandings have been reported in Kaneohe and Kahana Bays (Oahu) and throughout the main Hawaiian Islands (Eckert 1993b; NMFS and USFWS 1998b). Hawksbills primarily nest on the southeastern beaches of the Island of Hawaii. Since 1991, 81 nesting female hawksbills have been tagged on the Island of Hawaii at various locations. This number does not include nesting females from Maui or Moloka‘i, which would add a small number to the total. Post-nesting hawksbills have been tracked moving between Hawaii and Maui over the deep waters of the Alenuihaha Channel (Parker et al. 2009). Only two hawksbills have ever been sighted in the Pearl Harbor entrance channel, and none have been sighted inside the harbor (Smith 2010).

Water temperature in the Southern California region of the Action Area is generally too low for hawksbills, and they are rare in this region. Nesting is rare in the eastern Pacific Ocean region, and does not occur along the U.S. west coast (NMFS and USFWS 1998b; Witzell 1983). Stinson (1984b) did not mention the hawksbill turtle in her summary of sea turtle occurrences in eastern north Pacific waters from Baja California to the Gulf of Alaska, and no hawksbill sightings have been confirmed along the U.S. west coast in recent history (Eckert 1993b; NMFS and USFWS 2007a). If hawksbills were to occur in the Southern California region of the Action Area, it would most likely be during an El Niño event, when waters along the California current are unusually warm.

Hawksbills were once thought to be a nonmigratory species because of the proximity of suitable nesting beaches to coral reef feeding habitats and the high rates of marked turtles recaptured in these areas; however, tagging studies have shown otherwise. For example, a post-nesting female traveled 995 miles (1,601 km) from the Solomon Islands to Papua New Guinea (Meylan 1995), indicating that adult hawksbills can migrate distances comparable to those of green and loggerhead turtles.

Research suggests that movements of Hawaiian hawksbills are relatively short, with individuals generally migrating through shallow coastal waters and few deepwater transits between the islands. Nine hawksbill turtles were tracked within the Hawaiian Islands using satellite telemetry. Turtles traveled from 55 to 215 mi. (89 to 346 km) and took between 5 and 18 days to complete the trip from nesting to foraging areas (Parker et al. 2009).

4.3.11.2 *Population Structure*

Hawksbill sea turtles, like other sea turtles, are divided into regional groupings that represent major oceans or seas: the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea. In these regions, the population structure of hawksbill turtles are usually based on the distribution of their nesting aggregations.

4.3.11.3 *Natural Threats*

The various habitat types hawksbill turtles occupy throughout their lives expose these sea turtles to a wide variety of natural threats. The beaches on which hawksbill 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. Adult hawksbill turtles are also killed by sharks and other large, marine predators.

4.3.11.4 *Anthropogenic Threats*

Three human activities are known to threaten hawksbill turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of hawksbill turtle populations was overharvests by humans for subsistence and commercial purposes. In the Atlantic, hawksbill turtles are still captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines (Brautigam and Eckert 2006).

For centuries, hawksbill turtles have been captured for their shells, which have commercial value, rather than food (the meat of hawksbill turtles is considered to have a bad taste and can be toxic to humans) (NMFS and USFWS 1998d). Until recently, tens of thousands of hawksbills were captured and killed each year to meet demand for jewellery, ornamentation, and whole stuffed turtles (Eckert 1993a; Milliken and Tokunaga 1987). In 1988, Japan’s imports from Jamaica, Haiti and Cuba represented some 13,383 hawksbills: it is extremely unlikely that this volume could have originated solely from local waters (Greenpeace 1989 cited in Eckert 1993a). Although Japan banned the importation of turtle shell in 1994, domestic harvests of eggs and turtles continue in the United States, its territories, and dependencies, particularly in the Caribbean and Pacific Island territories. Large numbers of nesting and foraging hawksbill turtles are captured and killed for trade in Micronesia, the Mexican Pacific coast, southeast Asia and Indonesia (NMFS and USFWS 1998d). In addition to the demand for the hawksbill’s shell, there is a demand for other products including leather, oil, perfume, and cosmetics. Before the U.S. certified Japan under the Pelly Amendment, Japan had been importing about 20 metric tons of hawksbill shell per year, representing approximately 19,000 turtles.

The second most important threat to hawksbill turtles is the loss of nesting habitat caused by the expansion of resident human populations in coastal areas of the world and increased destruction or modification of coastal ecosystems to support tourism. Hawksbill sea turtles are also captured and killed in commercial fisheries. Like green turtles, hawksbill 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. Hawksbill sea turtles are the most likely species to ingest marine debris and the debris most commonly ingested is floating plastics (Schuyler et al. 2013). This study also suggests oceanic juvenile sea turtles may be more susceptible to ingesting oceanic debris than older benthic-feeding individuals (Schuyler et al. 2013).

4.3.11.5 *Status and Trends*

Hawksbill turtles were listed as endangered throughout their range under the ESA in 1970. Critical habitat for hawksbill turtles was designated in 1998 to include the coastal waters surrounding Mona and Monito Islands, Puerto Rico. Under the Convention on International Trade in Endangered Species of Wild Fauna and Flora, hawksbill turtles are identified as "critically endangered" (IUCN 2010).

The Hawksbill Sea Turtle (*Eretmochelys imbricata*) 5-year Review: Summary and Evaluation (NMFS and USFWS 2013) assessed nesting abundance and nesting trends in all regions that the hawksbill turtles inhabit. Where possible, historical population trends were determined, and most showed declines for the 20 to 100 year period of evaluation. Recent trends for 42 of the sites indicated that 69 percent were decreasing, seven percent were stable, and that 24 percent were increasing. Seven of the 83 sites occur in the central Pacific Ocean and one occurs in the eastern Pacific Ocean (Baja California, Mexico), all with decreasing long-term population trends; only the Hawaii site has a recent increasing trend. Hawksbills in the eastern Pacific Ocean are probably the most endangered sea turtle population in the world (Gaos et al. 2008). Hawksbills sometimes nest in the southern part of the Baja Peninsula, while juveniles and subadults are seen foraging in coastal waters regularly. No nesting occurs on the western coast of the United States. Hawksbills in the U.S. Pacific region nest only on eastern beaches of the Island of Hawaii (5 to 10 nesting females annually, although 13 were reported in 2011 (Rivers 2011), as well as in the Northwestern Hawaiian Islands (NMFS and USFWS 2013).

Hawksbill turtles, like green turtles, are thought to be declining globally as a direct consequence of a historical combination of overexploitation and habitat loss. However, like several of the species we have already discussed, the information available on the status and trend of hawksbill turtles do not allow us to make definitive statements about the global extinction risks facing these sea turtles or the risks facing particular populations (nesting aggregations) of these turtles. The limited data available suggests that several hawksbill turtles populations exist at sizes small enough to be classified as "small" populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) while others are large enough to avoid these problems. Exogenous threats such as overharvests and entanglement in fishing gear only increase their probabilities of becoming extinct in the reasonably foreseeable future.

4.3.11.6 *Diving and Social Behavior*

The duration of foraging dives in hawksbill turtles commonly depends on the size of the turtle: larger turtles diving deeper and longer. Shorter and more active foraging dives occur predominantly during the day, while longer resting dives occur at night (Blumenthal et al. 2009; Storch et al. 2005). Lutcavage and Lutz (1997) cited a maximum dive duration of 73.5 minutes for a female hawksbill in the U.S. Virgin Islands. Van Dam and Diez (Diez and van Dam 2002) reported that foraging dives at a study site in the northern Caribbean ranged from 19 to 26 minutes at depths of 26 to 33 ft. (8 to 10 m), with resting night dives from 35 to 47 minutes. Foraging dives of immature hawksbills are shorter, ranging from 8.6 to 14 minutes, with a mean and maximum depth of 16.4 and 65.6 ft. (5 and 20 m), respectively (van Dam and Diez 1996).

Blumenthal et al. (2009) reported consistent diving characteristics for juvenile hawksbill in the Cayman Islands, with an average daytime dive depth of 25 ft. (8 m), a maximum depth of 140 ft. (43 m), and a mean nighttime dive depth of 15 ft. (5 m). A change in water temperature affects dive duration; cooler water temperatures in the winter result in increased nighttime dive durations (Storch et al. 2005).

4.3.11.7 *Vocalizations and Hearing*

Sea turtles do not appear to use sound for communication, and there are no published recordings of hawksbill turtle vocalizations. Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 Hz to 2,000 Hz, with a range of maximum sensitivity between 100 Hz and 800 Hz (Bartol et al. 1999c; Lenhardt 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Recent research measuring hatchling hawksbill turtle auditory evoked potentials has shown that aerial and underwater acoustic stimuli elicited auditory evoked potential responses between 50 Hz and 1,600 Hz (underwater fully submerged and in air), with maximum sensitivity between 200 Hz and 400 Hz in hatchling hawksbills (Dow Piniak et al. 2011).

4.3.11.8 *Critical Habitat*

On September 2, 1998, NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey. This critical habitat does not occur within the HSTT Action Area.

4.3.12 **Leatherback Turtle**

The leatherback turtle (*Dermochelys coriacea*) is the largest turtle and the largest living reptile in the world. Mature males and females can be as long as six and a half feet (2 m) and weigh almost 2000 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 leatherbacks 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).

Leatherbacks 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.

4.3.12.1 *Distribution*

Leatherback 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 turtles are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Eckert and Eckert 1988; Eckert 1999a; Morreale et al. 1994). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert 1998). In the North Atlantic Ocean, leatherback turtles regularly occur in deep waters (>328 ft), and an aerial survey study in the north Atlantic sighted leatherback turtles in water depths ranging from 3 to 13,618 ft, with a median sighting depth of 131.6 ft (CETAP 1982). This same study found leatherbacks in waters ranging from 7 to 27.2 °C. 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. Males are rarely observed near nesting areas, and it has been hypothesized that leatherback turtles probably mate outside of tropical waters, before females swim to their nesting beaches (Eckert and Eckert 1988).

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 2007b). 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.

The primary data available for leatherbacks in the North Pacific Transition Zone come from longline fishing bycatch reports, as well as several satellite telemetry data sets (Benson et al. 2007a). 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 through the North Pacific Transition Zone to summer foraging grounds off the coast of northern California (Benson et al. 2007a). Genetic sampling of 18 leatherback turtles caught in the Hawaiian longline fishery indicated that about 94 percent originated from western Pacific Ocean nesting beaches (NMFS and USFWS 2007b). The remaining six percent of the leatherback turtles found in the open ocean waters north and south of the Hawaiian Islands represent nesting groups from the eastern tropical Pacific Ocean.

Leatherback turtles are regularly sighted by fishermen in offshore waters surrounding the Hawaiian Islands, generally beyond the 3,800 ft. (1,158 m) contour, and especially at the southeastern end of the island chain and off the northern coast of Oahu (Balazs 1995a). Leatherbacks encountered in these waters, including those caught accidentally in fishing operations, may be migrating through the Insular Pacific-Hawaiian Large Marine Ecosystem (NMFS and USFWS 1998a). Sightings and reported interactions with the Hawaii longline fishery commonly occur around seamount habitats above the Northwestern Hawaiian Islands (from 35° N to 45° N and 175° W to 180° W) (Skillman and Balazs 1992; Skillman and Kleiber 1998).

The leatherback turtle occurs within the entire Insular Pacific-Hawaiian Large Marine Ecosystem beyond the 330 ft. (101 m) isobath; inshore of this isobath is the area of rare leatherback occurrence. Incidental captures of leatherbacks have also occurred at several offshore locations around the main Hawaiian Islands (McCracken 2000). Although leatherback bycatches are common off the island chain, leatherback-stranding events on Hawaiian beaches are uncommon. Since 1982, only five leatherbacks have stranded in the Hawaiian Islands (Chaloupka et al. 2008a). Leatherbacks were not sighted during any of the aerial surveys, all of which took place over waters lying close to the Hawaiian shoreline. Leatherbacks were also not sighted during any of the NMFS shipboard surveys; their deep diving capabilities and long submergence times reduce the probability that observers could spot them during marine surveys. One leatherback turtle was observed along the Hawaiian shoreline during monitoring surveys in 2006 (Rivers 2011).

In the eastern North Pacific Ocean, leatherback turtles are broadly distributed from the tropics to as far north as Alaska, where 19 occurrences were documented between 1960 and 2001 (Eckert 1993a; Eckert 1993b; Hodge and Wing 2000). Stinson (Stinson 1984a) concluded that the leatherback was the most common sea turtle in U.S. waters north of Mexico. Aerial surveys off California, Oregon, and Washington indicate that most leatherbacks occur in waters over the continental slope, with a few beyond the continental shelf (Eckert 1993a). While the leatherback is known to occur throughout the California Current Large Marine Ecosystem, it is not known to nest anywhere along the U.S. Pacific Ocean coast. In general, turtle sightings increase during summer, as warm water moves northward along the coast (Stinson 1984a). Sightings may also be more numerous in warm years than in cold years.

Leatherback turtles are regularly seen off the western coast of the United States, with the greatest densities found off central California. Off central California, sea surface temperatures are highest during the summer and fall, and oceanographic conditions create favorable habitat for leatherback turtle prey (jellyfish). Satellite telemetry data indicate that these animals are within the California Current Large Marine Ecosystem, as well as that portion of the Action Area that is included within it (Benson et al. 2007a). There is some evidence that they follow the 61 °F (16 C) isotherm into Monterey Bay, and the length of their stay apparently depends on prey availability (Starbird et al. 1993). Satellite telemetry studies link leatherback turtles off the U.S. west coast to one of the two largest remaining Pacific Ocean breeding populations in Jamursba Medi, Indonesia. Thus, nearshore waters off central California represent an important foraging region for the critically endangered Pacific Ocean leatherback turtle. There were 96 sightings of leatherbacks within 50 km of Monterey Bay from 1986 to 1991, mostly by recreational boaters (Starbird et al. 1993).

4.3.12.2 *Population Structure*

Leatherbacks break into four nesting aggregations: Pacific, Atlantic, and Indian oceans, and the Caribbean Sea. Detailed population structure is unknown, but is likely dependent upon nesting beach location.

4.3.12.3 *Natural Threats*

The various habitat types leatherback turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which leatherback 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 (Caut et al. 2009). Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Larger leatherback turtles, including adults, are also killed by sharks and other large, marine predators (Pitman and Dutton 2004).

4.3.12.4 *Anthropogenic 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, and ingestion of marine debris (NMFS and USFWS 1998e).

The foremost threat is the number of leatherback turtles killed or injured in fisheries. Spotila (2000) 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 per year. 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 (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 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 Hawaii are estimated to have captured and killed several hundred leatherback 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 1 or 2 leatherback turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii are estimated to have captured about 19 leatherback 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 reasonably foreseeable future (NMFS 2008a). 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 fisheries in the Gulf of Mexico capture the largest number of leatherback turtles: each year, they have been estimated to capture about 1,393 leatherback turtles with 144 of those sea turtles dying as a result. Along the Atlantic coast of the U.S., NMFS estimated that about 800 leatherback 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 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 turtles, leatherback 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 2010).

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 2010).

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 2010). 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 2010).

The likelihood of leatherback sea turtles ingesting oceanic debris increased significantly over time, although the likelihood plateaued in the mid 1980s suggesting leatherback feeding habitat had become saturated with debris at that point (Schuyler et al. 2013). Ingestion of debris occurred primarily in pelagic areas and floating plastics was the most commonly ingested material.

4.3.12.5 *Status and Trends*

The leatherback turtles are listed as endangered under the ESA throughout the species' global range. Increases in the number of nesting females have been noted at some sites in the Atlantic Ocean, but these are far outweighed by local extinctions, especially of island populations, and the demise of populations throughout the Pacific, such as in Malaysia and Mexico. Spotila et al. (1996) estimated the global population of female leatherback turtles to be only 34,500 (confidence limits: 26,200 to 42,900) nesting females; however, the eastern Pacific population has continued to decline since that estimate, leading some researchers to conclude that the leatherback is now on the verge of extinction in the Pacific Ocean (e.g., Spotila et al. 1996; Spotila et al. 2000).

Globally, leatherback turtle populations have been decimated. In 1980, the global leatherback population was estimated at approximately 115,000 adult females (Pritchard 1982). By 1995, this global population (of adult females) is estimated to have declined to 34,500 (Spotila et al. 1996). Throughout the Pacific, leatherbacks are seriously declining at all major nesting beaches (NMFS USFWS 2013).

Leatherback sea turtles appear to be in a critical state of decline in the North Pacific Ocean. The leatherback population that nests along the Pacific Ocean was estimated to be over 91,000 adults in 1980 (Spotila et al. 1996) or greater than 39,000 nests (NMFS and USFWS 2013), but is now estimated to number 3,172 total nests (NMFS and USFWS 2013). Leatherback turtles have experienced major declines at all major Pacific basin rookeries.

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (NMFS and USFWS 1998a; Spotila et al. 1996; Spotila et al. 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing (Eckert and Sarti 1997; Sarti et al. 1996).

For several years, NMFS' biological opinions have established that leatherback populations currently face high probabilities of extinction as a result of both environmental and demographic stochasticity. Demographic stochasticity, which is chance variation in the birth or death of an individual of the population, is facilitated by the increases in mortality rates of leatherback populations resulting from the premature deaths of individual sea turtles associated with human activities (either removal of eggs or adult females that are killed on nesting beaches or that die as a result of being captured in fisheries) or incidental capture and mortality of individuals in various fisheries.

In the Pacific Ocean, leatherback turtles are endangered as a direct consequence of a historical combination of overexploitation and habitat loss. The information available suggests that leatherback turtles have high probabilities of becoming extinct in the Pacific Ocean unless they are protected from the combined threats of entanglements in fishing gear, overharvests, and loss of their nesting habitat. The limited data available suggests that leatherback turtles exist at population sizes small enough to be classified as "small" populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) as evidenced by biases in the male to female ratios in the Pacific. The status of leatherback turtles in the Atlantic Ocean appears to be stable (NMFS and USFWS 2013).

4.3.12.6 *Diving and Social Behavior*

The maximum dive depths for leatherbacks have been recorded at over 1,000 m (Doyle et al. 2008), with routine dives recorded between 50 and 84 m. The maximum dive length recorded for such female leatherback turtles was 86.5 minutes (Lopez-Mendilahars et al 2008), while routine dives ranged from 4 to 14.5 minutes (in Lutcavage and Lutz 1997). Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert et al. 1989).

A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their inter-nesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57 to 68 percent of their time submerged. Mean dive depth was 19 ± 1 m and the mean dive duration was 7.4 ± 0.6 minutes (Southwood et al. 1999). Similarly, Eckert (1999) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 m in depth, although maximum depths ranged from 132 m to over 750 m. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert (1999) speculates that these short duration dives most likely represent just surfacing activity after each dive. Excluding these short dives, five of the turtles had dive durations greater than 24 minutes, while three others had dive durations between 12 to 16 minutes.

Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora et al. 1984, cited in Southwood et al. 1999). Based on depth profiles of four leatherbacks tagged and tracked from Monterey Bay, California in 2000 and 2001, using satellite-linked dive recorders, most of the dives were to depths of less than 100 meters and most of the time was spent shallower than 80 meters. Based on preliminary analyses of the data, 75 to 90 percent of the time the leatherback turtles were at depths less than 80 m.

4.3.12.7 *Vocalizations and Hearing*

Sea turtles do not appear to use sound for communication. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook and Forrest 2005; Mrosovsky 1972). However these sounds appeared to be associated with breathing (Cook and Forrest 2005; Mrosovsky 1972).

Recent research measuring hatchling leatherback turtle auditory evoked potentials has shown that hatchling leatherbacks respond to tonal stimuli between 50 and 1,200 underwater (maximum sensitivity: 100 to 400 Hz) and 50 and 1,600 in air (maximum sensitivity: 50 to 400Hz) (Dow Piniak et al. 2012a).

4.3.12.8 *Critical Habitat*

On March 23, 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, U.S.V.I. from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity. On January 26, 2012, the NMFS designated critical habitat for leatherback sea turtles in waters along Washington State and Oregon (Cape Flattery to Cape Blanco; 64,760 km^2) and California (Point Arena to Point Arguello; 43,798 km^2). The primary constituent element of these areas includes (1.) the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (*Chrysaora, Aurelia, Phacellophora,* and *Cyanea*), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

Leatherback sea turtle critical habitat was addressed in Section 4.2.5 and will not be discussed further.

4.3.13 **Loggerhead Turtle**

Loggerhead turtles (*Caretta caretta*) were named for their relatively large heads, which support powerful jaws and enable them to feed on hard-shelled prey, such as whelks and conch. The carapace (top shell) is slightly heart-shaped and reddish-brown in adults and sub-adults, while the plastron (bottom shell) is generally a pale yellowish color. The neck and flippers are usually dull brown to reddish brown on top and medium to pale yellow on the sides and bottom. Mean straight carapace length of adults in the southeastern U.S. is approximately 36 in (92 cm); corresponding weight is about 250 lbs (113 kg).

Loggerheads reach sexual maturity at around 35 years of age. In the southeastern U.S., mating occurs in late March to early June and females lay eggs between late April and early September. Females lay three to five nests, and sometimes more, during a single nesting season. The eggs incubate approximately two months before hatching sometime between late June and mid-November.

Hatchlings vary from light to dark brown to dark gray dorsally and lack the reddish-brown coloration of adults and juveniles. Flippers are dark gray to brown above with white to white-gray margins. The coloration of the plastron is generally yellowish to tan. At emergence, hatchlings average 1.8 in (45 mm) in length and weigh approximately 0.04 lbs (20 g).

4.3.13.1 *Distribution*

Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons in temperate, subtropical, and tropical waters. Major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics (NMFS and USFWS 1998e). The majority of loggerhead nesting is at the western rims of the Atlantic and Indian Oceans. Nesting aggregations occur in the eastern Atlantic at Cape Verde, Greece, Libya, Turkey and along the West African Coast. The western Atlantic and Caribbean hosts nesting aggregations along the U.S. east coast from Virginia through the Florida peninsula, the Dry Tortugas and Northern Gulf of Mexico, the Bahamas, the Yucatan Peninsula, Central America and the Caribbean and into South America. Within the Indian Ocean, nesting aggregations occur at Oman, Yemen, Sri Lanka and Madagascar and South Africa. Pacific Ocean nesting sites include western and eastern Australia and Japan.

Adult loggerheads are known to make considerable migrations from nesting beaches to foraging grounds (TEWG 2009); and evidence indicates turtles entering the benthic environment undertake routine migrations along the coast that are limited by seasonal water temperatures. Small juveniles are found in pelagic waters (e.g., of the North Atlantic and the Mediterranean Sea); and the transition from oceanic to neritic juvenile stages can involve trans-oceanic migrations (Bowen et al. 2004). Loggerhead nesting is confined to lower latitudes, concentrated in temperate zones and subtropics; the species generally does not nest in tropical areas (NMFS and USFWS 1991b; NRC 1990; Witherington et al. 2006). Loggerhead turtles travel to northern waters during spring and summer as water temperatures warm, and southward and offshore toward warmer waters in fall and winter; loggerheads are noted to occur year round in offshore waters of sufficient temperature.

4.3.13.2 *Population Structure*

Under the ESA, loggerhead sea turtles are divided into nine DPS: the threatened Northwest Atlantic Ocean DPS, South Atlantic Ocean DPS, Southeast Indo-Pacific Ocean DPS, and Southwest Indian Ocean DPS, and the endangered Northeast Atlantic Ocean DPS, Mediterranean Sea DPS, North Indian Ocean DPS, North Pacific Ocean DPS, and South Pacific Ocean DPS. Only individuals from the North Pacific Ocean DPS are likely to occur in the HSTT Action Area. Within the DPS, the population structure of loggerhead turtles is usually based on the distribution of their nesting aggregations. In the Pacific Ocean, loggerhead turtles have nesting aggregations in Japan (Hatase et al. 2002), Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. One of the largest loggerhead nesting aggregations in the world is found in Oman, in the Indian Ocean.

4.3.13.3 *Natural Threats*

The various habitat types loggerhead turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural and anthropogenic threats. The beaches on which loggerhead 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. For example, in

1992, all of the eggs over a 90-mile length of coastal Florida were destroyed by storm surges on beaches that were closest to the eye of Hurricane Andrew (Milton et al. 1994). Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Adult loggerhead turtles are also killed by sharks and other large, marine predators. Loggerhead sea turtles are also killed by cold stunning, exposure to biotoxins, sharks and other large, marine predators.

4.3.13.4 ***Anthropogenic Threats***

A wide variety of human activities adversely affect hatchlings and adult female turtles when they are on land, including beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; human presence on nesting beaches; beach driving; coastal construction and fishing piers that alter patterns of erosion and accretion on nesting beaches; exotic dune and beach vegetation; and poaching. As the size of the human population in coastal areas increases, that population brings with it secondary threats such as exotic fire ants, feral hogs, dogs, and the increase of native species that tolerate human presence (*e.g.*, raccoons, armadillos, and opossums) and which feed on turtle eggs.

When they are in coastal or marine waters, loggerhead turtles are affected by a completely different set of human activities that include discharges of toxic chemicals and other pollutants into the marine ecosystem; underwater explosions; hopper dredging, offshore artificial lighting; entrainment or impingement in power plants; entanglement in marine debris; ingestion of marine debris; boat collisions; poaching, and interactions with commercial fisheries. Of these, interactions with fisheries represent a primary threat because of the number of individuals that are captured and killed in fishing gear each year.

Loggerhead turtles are captured and killed in commercial fisheries. In the Pacific Ocean, between 2,600 and 6,000 loggerhead turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004). Shallow-set Hawai‘i based longline fisheries are estimated to have captured and killed several hundred loggerhead 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 fewer than 5 loggerhead turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii are estimated to have captured 45 loggerhead turtles, killing about 10 of these sea turtles. A recent biological opinion on these fisheries expected this rate of interaction and deaths to continue into the reasonably foreseeable future (NMFS and USFWS 2008). Loggerhead 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 fisheries account for the highest number of loggerhead turtles that are captured and killed, but they are also captured and killed in trawls, traps and pots, longlines, and dredges. Along the Atlantic coast of the U.S., NMFS estimated that almost 81,620 loggerhead turtles are captured in shrimp fisheries each year in the Gulf of Mexico, with 7,701 of those sea turtles dying as a result of their capture. Each year, several hundred loggerhead turtles are also captured in herring fisheries; mackerel, squid, and butterfish fisheries; monkfish fisheries; pound net fisheries, summer flounder and scup fisheries; Atlantic pelagic longline fisheries; and gillnet fisheries in Pamlico Sound. Although most of these turtles are released alive, these fisheries are combined to capture about 2,000 loggerhead turtles each year, killing almost 700; the health effects of being captured on the sea turtles that survive remain unknown.

In the pelagic environment, loggerhead turtles are exposed to a series of longline fisheries that include the U.S. Atlantic tuna and swordfish longline fisheries, an Azorean longline Fleet, a Spanish longline Fleet, and various Fleets in the Mediterranean Sea (Aguilar et al. 1995; Bolten et al. 2002). In the benthic environment in waters off the coastal U.S., loggerheads are exposed to a suite of fisheries in Federal and state waters including trawl, purse seine, hook and line, gillnet, pound net, longline, dredge, and trap fisheries.

Like all of the other sea turtles we have discussed, loggerhead 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.

4.3.13.5 ***Status and Trends***

The North Pacific DPS of loggerhead sea turtles was listed as endangered in 2011 (76 FR 58868). The global abundance of nesting female loggerhead turtles is estimated at 43,320 to 44,560 (Spotila 2004). All loggerheads inhabiting the North Pacific Ocean are derived primarily, if not entirely, from Japanese beaches (although low level nesting may occur in areas around the South China Sea) (Chan et al. 2007). Along the Japanese coast, nine major nesting beaches (greater than 100 nests per season) and six "submajor" beaches (10 to 100 nests per season) were identified. Using information collected from these nine beaches Kamezaki et al. (Kamezaki et al. 2003) found a substantial decline (50 to 90 percent) in the size of the annual loggerhead nesting population over the last half of the 20th century. Also, nest count data for the last two decades suggests that the North Pacific population is "small" and lacks a robust gene pool when compared to the larger northwest Atlantic and north Indian Ocean loggerhead populations. Small populations are more susceptible to demographic variability which increases their probability of extinction. Available evidence indicates that due to loss of adult and juvenile mortalities from fishery bycatch and, to a lesser degree the loss of nesting habitat, the North Pacific loggerhead population is declining.

Snover (2008) combined nesting data from the Sea Turtle Association of Japan and data from Kamezaki et al. (2002) to analyze an 18-year time series of nesting data from 1990 through 2007. Nesting declined from an initial peak of approximately 6,638 nests in 1990 and 1991, followed by a steep decline to a low of 2,064 nests in 1997. During the past decade, nesting increased gradually to 5,167 nests in 2005, declined and then rose again to a high of just under 11,000 nests in 2008. Estimated nest numbers for 2009 were on the order of 7,000 to 8,000 nests. While nesting numbers have gradually increased in recent years and the number for 2009 was similar to the start of the time series in 1990, historical evidence from Kamouda Beach (census data dates back to the 1950s) indicates that there has been a substantial decline over the last half of the 20th century (Kamezaki et al. 2003) and that current nesting represents a fraction of historical nesting levels.

In addition, loggerheads uncommonly occur in U.S. Pacific waters, and there were no documented strandings of loggerheads on the Hawaiian Islands in nearly 20 years (1982 to 1999 stranding data). There are very few records of loggerheads nesting on any of the many islands of the central Pacific, and the species is considered rare or vagrant in this region (USFWS 1998). Overall, Gilman (2009) estimated that the number of loggerheads nesting the Pacific has declined by 80 percent in the past 20 years.

4.3.13.6 *Diving and Social Behavior*

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). The maximum recorded dive depth for a post-nesting female was 211 to 233 m, while mean dive depths for both a post-nesting female and a subadult were 9 to 22 m. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamot et al. 1990 cited in Lutcavage and Lutz 1997).Two loggerheads tagged by Hawaii-based longline observers in the North Pacific and attached with satellite-linked dive recorders were tracked for about 5 months. Analyses of the dive data indicate that most of the dives were very shallow with 70 percent of the dives no deeper than 5 m. In addition, the loggerheads spent approximately 40 percent of their time in the top meter and nearly all of their time at depths shallower than 100 m. On 5 percent of the days, the turtles dove deeper than 100 m; the deepest daily dive recorded was 178 m (Polovina et al. 2003).

Polovina et al. (2004) reported that tagged turtles spent 40 percent of their time at the surface and 90 percent of their time at depths shallower than 40 m. On only five percent of recorded dive days loggerheads dove to depths greater than 100 m at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 m. There were also several strong surface temperature fronts the turtles were associated with, one of 20 °C at 28° N latitude and another of 17 °C at 32° N latitude.

4.3.13.7 *Vocalizations and Hearing*

Two studies have been conducted to measure loggerhead turtle hearing sensitivity, each using a slightly different methodology. Vibratory stimuli delivered directly to the tympanun produced auditory brainstem responses in loggerheads between 250 Hz and 750 Hz (Bartol et al. 1999b). Underwater tones elicited behavioral responses to frequencies between 50 and 800 Hz and auditory evoked potential responses between 100 Hz and 1,131 Hz in one adult loggerhead (Martin et al. 2012). The lowest threshold recorded in this study was 98 dB re: 1 µPa at 100 Hz.

Lavender et al. (2014) found post-hatchling loggerheads responded to sounds in the range of 50 Hz to 800 Hz while juveniles responded to sounds in the range of 50 Hz to 1,000 Hz. Post-hatchlings had the greatest sensitivity to sounds at 200 Hz while juveniles had the greatest sensitivity at 800 Hz (Lavender et al. 2014).

4.3.13.8 *Critical Habitat*

On July 18, 2013, NMFS proposed critical habitat for loggerhead sea turtles along the U.S. Atlantic and Gulf of Mexico coasts from North Carolina to Mississippi (78 FR 43005).

4.3.14 Olive Ridley Turtle

The olive ridley turtle (*Lepidochelys olivacea*) is a small to medium-sized sea turtle; adults typically range between 55 and 80 cm (22 to 31 in) in carapace length and weigh around 45 kg (100 lb). They are olive/grayish-green (darker in the Atlantic than in the Pacific) with a heart-shaped top shell (carapace) with 5 to 9 pairs of costal "scutes" with 1 to 2 claws on their flippers; hatchlings emerge mostly black with a greenish hue on the sides.

4.3.14.1 *Distribution*

Olive ridley turtles occur in the tropical waters of the Pacific and Indian Oceans from Micronesia, Japan, India, and Arabia south to northern Australia and southern Africa. In the Atlantic Ocean, they occur off the western coast of Africa and the coasts of northern Brazil, French Guiana, Surinam, Guyana, and Venezuela in South America, and occasionally in the Caribbean Sea as far north as Puerto Rico. In the eastern Pacific Ocean, olive ridley turtles are found from the Galapagos Islands north to California. While Pacific ridley turtles have a generally tropical to subtropical range, individual turtles have been reported as far as the Gulf of Alaska (Hodge and Wing 2000).

Olive ridley turtles nest along continental margins and oceanic islands. The largest nesting aggregation in the world occurs in the Indian Ocean along the northeast coast of India where more than 600,000 olive ridley turtles nested in a single week in 1991 (Mrosovsky 1993). The second most important nesting area occurs in the eastern Pacific along the west coast of Mexico and Central America. Olive ridley turtles also nest along the Atlantic coast of South America, western Africa, and the western Pacific (Groombridge 1982; Sternberg and Pritchard 1981).

In the eastern Pacific, olive ridley turtles nest along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Few turtles nest as far north as southern Baja California, Mexico (Fritts et al. 1982) or as far south as Peru (Brown and Brown 1982). The post-nesting migration routes of olive ridleys traversed thousands of kilometers of deep oceanic waters, ranging from Mexico to Peru, and more than 3,000 kilometers out into the central Pacific (Plotkin 2007). Although they are the most abundant north Pacific sea turtle, surprisingly little is known of the oceanic distribution and critical foraging areas of Pacific ridley turtles.

Most records of olive ridley turtles are from protected, relative shallow marine waters. Nevertheless, olive ridley turtles have also been observed in the open ocean. Since olive ridley turtles throughout the eastern Pacific Ocean depend on rich upwelling areas off South America for food, Pacific ridley turtles sighted offshore may have been foraging.

4.3.14.2 *Population Structure*

Olive ridley sea turtles exist as two separate populations: one that occurs in the western Pacific and Indian Ocean (northern Australia, Malaysia, Thailand, and the State of Orissa in India) and another that occurs along the Pacific coast of the Americas from Mexico to Colombia (Chaloupka et al. 2004b). The eastern Pacific Ocean population is the population that overlaps with the HSTT Action Area.

4.3.14.3 *Natural Threats*

The various habitat types olive ridley turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which olive ridley 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. Adult olive ridley turtles are also killed by sharks and other large, marine predators.

4.3.14.4 *Anthropogenic Threats*

In India, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large scale mortality of adult olive ridley turtles during the last two decades. Since 1993, more than 50,000 Olive ridleys have stranded along the coast, at least partially because of near-shore shrimp fishing (Shanker and Mohanty 1999). Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997-1998 season and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets despite mandatory requirements passed in 1997 (Pandav and Choudhury 1999).

Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (NMFS and USFWS 1998f). However, human-induced mortality caused this population to decline. From the 1960s to the 1970s, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan (NMFS and USFWS 1998f). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather (Green and Ortiz-Crespo 1982).

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Arenas et al. 2000). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still seriously decremented and is threatened with extinction (Silva-Batiz et al. 1996). Nevertheless, some authors have suggested that olive ridley turtles in Mexico should be considered recovered (Arenas et al. 2000).

The main threats to turtles in Thailand include egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e. carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi et al. 1999). During the 1996 to 1997 survey, only six olive ridley nests were recorded, and of these, half were poached, and one was predated by feral dogs. During the 1997 to 1998 survey, only three nests were recorded.

Olive ridley nests in Indonesia are subject to extensive hunting and egg collection. In combination with rapid rural and urban development, these activities have reduced the size of the nesting population in the region as well as their nesting success.

4.3.14.5 ***Status and Trends***

Olive ridley turtle populations on the Pacific coast of Mexico were listed as endangered under the ESA in 1978 (61 FR 17) ; all other populations are listed as threatened. The International Union for Conservation of Nature and Natural Resources has classified the olive ridley turtle as “endangered” (IUCN 2010).

Where population densities are high enough, nesting takes place in synchronized aggregations known as arribadas. The largest known arribadas in the eastern Pacific are off the coast of Costa Rica (~475,000 to 650,000 females estimated nesting annually) and in southern Mexico (~800,000 nests per year at La Escobilla, in Oaxaca, Mexico). In Costa Rica, 25,000 to 50,000 olive ridleys nest at Playa Nancite and 450,000 to 600,000 turtles nest at Playa Ostional each year (NMFS and USFWS 1998f). In an 11-year review of the nesting at Playa Ostional, Ballestero (Ballestero et al. 2000) reported that the data on numbers of nests deposited was too limited for a statistically valid determination of a trend; although the number of nesting turtles appeared to decline over a six-year period.

At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single arribada produced hatchlings (NMFS and USFWS 1998f). In addition, some female olive ridleys nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre et al. 1999). At Playa La Flor, the second most important nesting beach for Pacific ridleys on Nicaragua, Ruiz (Ruiz 1994) documented 6 arribadas (defined as 50 or more females resting simultaneously). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*).

In the western Pacific, information on the size of olive ridley nesting aggregations are limited although they do not appear to be recovering (with the exception of the nesting aggregation at

Orissa, India). There are a few sightings of Olive ridleys from Japan, but no reports of egg-laying. Similarly, there are no nesting records from China, Korea, the Philippines, Taiwan, Viet Nam, or Kampuchea, and nesting records in Indonesia are not sufficient to assess population trends (Eckert 1993a; Suwelo 1999). In Thailand, olive ridleys occur along the southwest coast, on the Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles has declined markedly from 1979 to 1990.

Olive ridley turtles have been observed in Indonesia and surrounding waters, and some olive ridley turtles have been documented as nesting in this region recently. On Jamursba-Medi beach, on the northern coast of Irian Jaya, 77 olive ridley nests were documented from May to October, 1999 (Teguh 2000 in (Putrawidjaja 2000).

Olive ridley turtles nest on the eastern and western coasts of peninsular Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 eggs (2,400 nests, with approximately 100 eggs per nest (see Siow and Moll 1982, in Eckert 1993a), while only 187 nests were reported from the area in 1990 (Eckert 1993a). In eastern Malaysia, olive ridleys nest very rarely in Sabah and only a few records are available from Sarak (Eckert 1993a).

Olive ridleys are the most common species found along the east coast of India, migrating every winter to nest en-masse at three major rookeries in the state of Orissa, Gahirmatha, Robert Island, and Rushikulya (Pandav and Choudhury 1999). According to Pandav and Choudhury (1999), the number of nesting females at Gahirmatha has declined in recent years, although after three years of low nestings, the 1998 to 1999 season showed an increasing trend (Noronha Environmental News Service, April 14, 1999), and the 1999 to 2000 season had the largest recorded number of Pacific ridleys nesting in 15 years (The Hindu, March 27, 2000; The Times of India, November 15, 2000). During the 1996 to 1997 and 1997 to 1998 seasons, there were no mass nestings of olive ridleys. During the 1998 to 1999 nesting season, around 230,000 females nested during the first arribada, lasting approximately a week (Pandav and Kar 2000); unfortunately, 80 percent of the eggs were lost due to inundation and erosion (Shanker and Mohanty 1999). During 1999 to 2000, over 700,000 olive ridleys nested at Nasi Islands and Babubali Island, in the Gahirmatha coast.

4.3.14.6 *Diving and Social Behavior*

Although olive ridley turtles are probably surface feeders, they have been caught in trawls at depths of 80 to 110 m (NMFS and USFWS 1998f), and a post-nesting female reportedly dove to a maximum depth of 290 m. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin 1994, in Lutcavage and Lutz 1997).

4.3.14.7 *Vocalizations and Hearing*

Sea turtles do not appear to use sound for communication, and there are no published recordings of olive ridley turtle vocalizations. There is no information on olive ridley turtle hearing.

However, we assume that their hearing sensitivities will be similar to those of green, hawksbill, leatherback and loggerhead turtles: their best hearing sensitivity will be in the low frequency range, with maximum sensivitiy below 400 Hz and an upper hearing range not likely to exceed 2,000 Hz.

4.3.14.8 *Critical Habitat*

Olive ridley sea turtle critical habitat has not been designated.

5 ENVIRONMENTAL BASELINE

By regulation, environmental baselines for biological opinions include the past and present impacts of all state, Federal, or private actions and other human activities in the Action Area, the anticipated impacts of all proposed Federal projects in the Action Area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR § 402.02). The environmental baseline for this Opinion includes the effects of several activities that affect the survival and recovery of ESA-listed resources in the Action Area.

The following information summarizes the principal natural and human-caused phenomena in the HSTT Action Area believed to affect the survival and recovery of these species in the wild.

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. 2012). 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. 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. 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. 2012). 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.

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. 1999a; Payne et al. 1986; Payne et al. 1990). 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, Guadalupe fur seal). 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. 2009a; Fuentes et al. 2010; Fuentes et al. 2009b) (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. Baker et al. (2006) predicted sea level rise may result in significant levels of terrestrial habitat loss in the Northwest Hawaiian Islands for Hawaiian monk seals and green sea turtles. Hawaiian monk seals are predicted to experience more crowding and competition for landing sites when islands shrink. Similarly, green sea turtles were predicted to experience more crowding at suitable nesting beaches. Poloczanska et al. (Poloczanska et al. 2009) 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 (Veit et al 1996; Sydeman et al 2009; Doney et al 2012). 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). Similarly, the most severe coral bleaching events observed to date have typically been accompanied by ocean warming events such as the El Niño-Southern Oscillation (Glynn 2001). Bleaching episodes result in substantial loss of coral cover, and result in the loss of important habitat for associated reef fishes and other biota (e.g., sea turtles).

This is not an exhaustive review of all available literature regarding the potential impacts of climate change to the species considered in this Opinion. However, this review provides some examples of impacts that may occur. While it is difficult to accurately predict the consequences of climate change to the species considered in this Opinion, a range of consequences are expected, ranging from beneficial to catastrophic. Given a lack of available information within the context of the temporal scale of the action, specific climate change related impacts on the species evaluated in this Opinion are speculative, cannot be meaningfully assessed, and will not be considered further.

5.2 Vessel Strike

The following sections describe the *Environmental Baseline* for vessel strikes of cetaceans, pinnipeds, and sea turtles in the Action Area.

5.2.1 Cetaceans

Vessel strike is a threat to large whale species. This threat may be increasing with increases in commercial shipping because shipping lanes cross important large whale breeding and feeding habitats or migratory routes. Large whale ship strike data collected by NMFS from 1988 through January 2015 is summarized in the following sections.

5.2.1.1 ***SOCAL Range Complex Vessel Strikes***

Over the 10-year period from 2005 through 2014, there were 55 reported whale strikes in all of California. Of all California strikes, 94.5 percent were from civilian vessels. This figure includes the 58.2 percent of strikes where the source was unknown and assumed to be civilian (given that the Navy reports 100 percent of its strikes). The annual average of all whale strikes in California over 10 years is 5.50 whales per year (Figure 5). Gray and fin whales account for the majority (51 percent) of all California strikes (gray: 29 percent, fin: 22 percent; Figure 6).

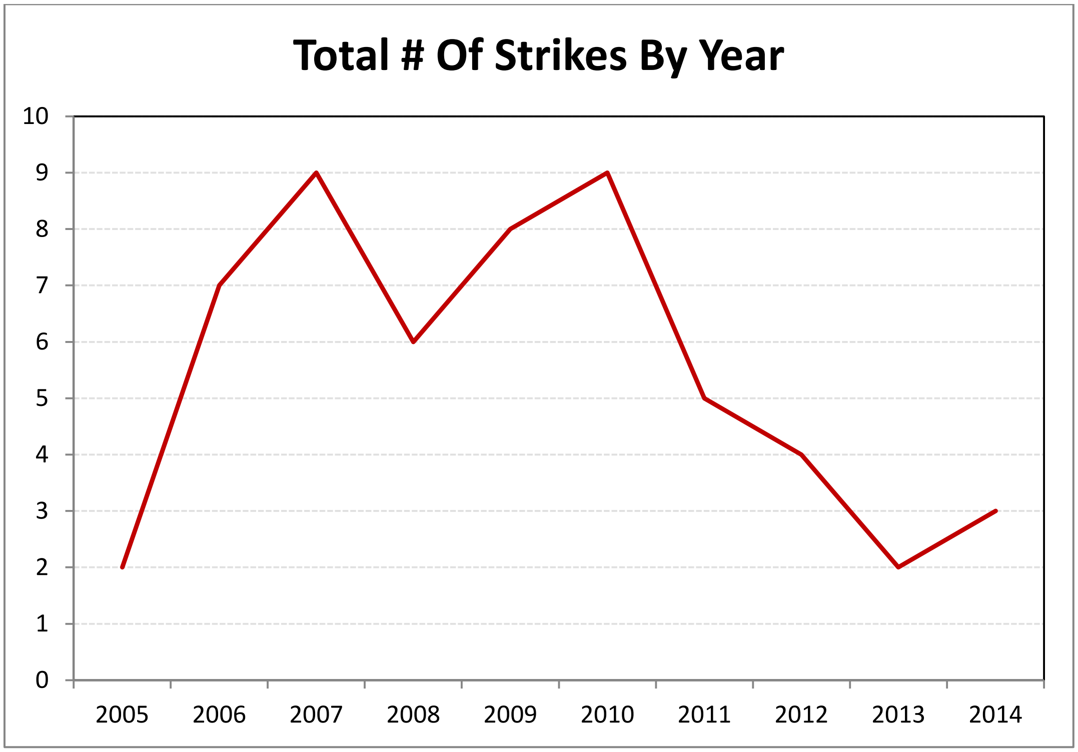

Figure 5. Total large whale vessel strikes per year in California from 2005 through 2014

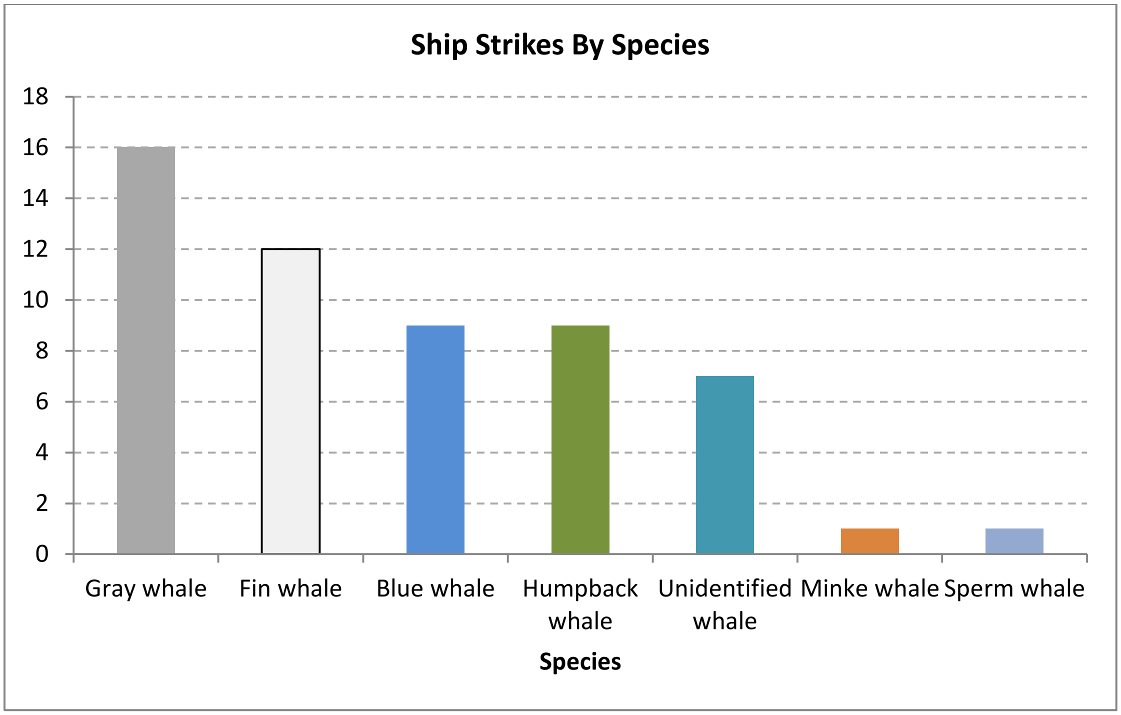

Figure 6. Large whale species composition of ship strikes in California from 2005 through 2014

Of all California strikes over the 10-year period 2005 through 2014, the Navy accounted for 5.45 percent (n = 3) of the strikes. Seventy-eight percent all California ship strikes occurred from non-Navy vessels outside of the SOCAL Range Complex. Of the 55 total California ship strikes over this period, only 12 occurred within the geographic area overlapping the Navy's SOCAL portion of HSTT. Forty-five percent of all California ship strikes occurred in association with the more northern portion of Southern California from approximately portions of Orange County to Point Conception. All of these areas are north of and outside of the Navy's SOCAL Range Complex. This region is also where the Los Angeles/Long Beach port complex is located.

There were 12 whale strikes (including both Navy and non-Navy strikes) in the SOCAL Range Complex from 2005 through 2014. Forty-two percent of the strikes were to gray whales, and 33 percent were to fin whales. There were no reported strikes to blue, Bryde's, humpback, minke, sperm, or sei whales in SOCAL over the 10-year period.

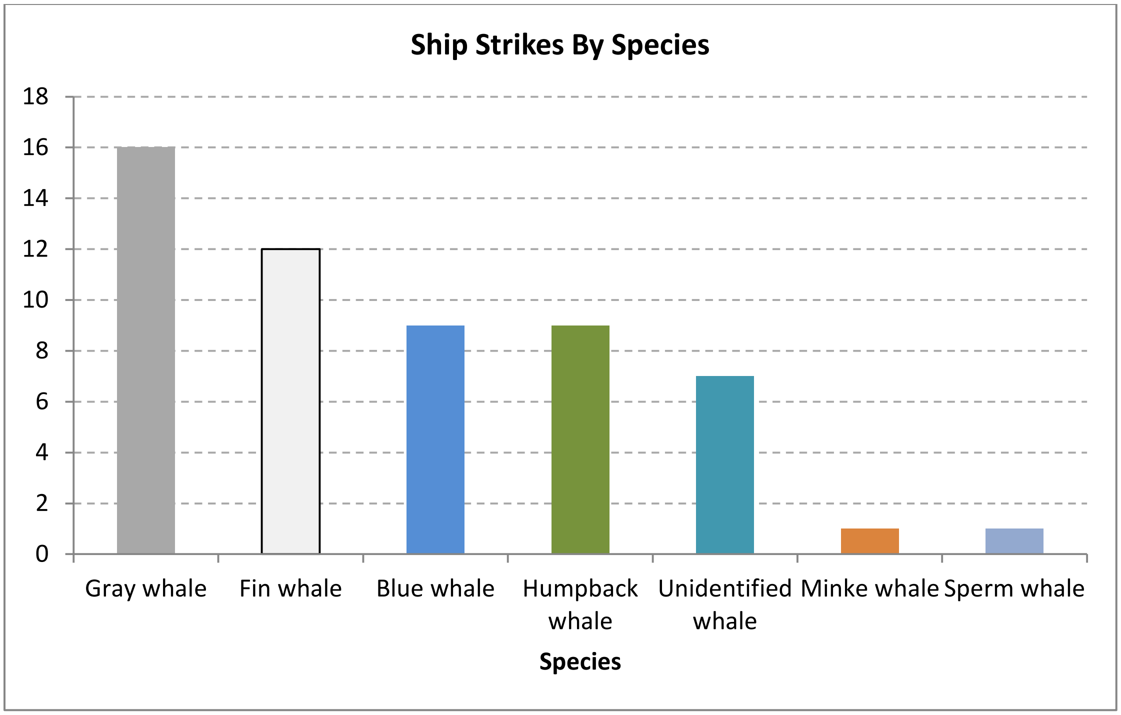

Figure 7. Large whale species composition of ship strikes in California from 2005 through 2014

Navy strikes accounted for 25 percent of the SOCAL total (n =3 of the 12). The SOCAL-only ship strike (all sources) annual average over the 10-year period is 1.2 per year, although there have been no reported SOCAL strikes since April 2012.

Table 18. Large whale strikes in California by vessel type (Source: NMFS West Coast Regional Office)

Vessel Type	# of ship strikes	% of total
Unknown	32	58%
Commercial ships	4	7%
Recreational (powerboat, sailboat)	4	7%
Ferry	3	5%
Navy vessels	3	5%
Coast Guard vessel/boat	3	5%
Cruise ship	2	4%
Commercial- Tug Boat	1	2%
Research Boat	1	2%
Commercial- Whale watching boat	1	2%
Commercial- Fishing boat	1	2%

5.2.1.2 *Hawaii Range Complex Vessel Strikes*

Ship strike data from Hawaii is limited, compared with ship strike data from SOCAL. The number of physical injuries to humpback whales as a result of ship collisions has increased in Hawaiian waters (Glockner-Ferrari et al. 1987; Lammers et al. 2007), possibly stemming from humpback whale population growth. In 2011, a Military Sealift Command vessel struck an unidentified whale while transiting between Guam and Oregon. This vessel was not associated with Pacific Fleet training or testing activities at the time, and therefore is not included in the risk assessment for vessel strike from training and testing activities.

5.2.1.3 *Impact of vessel strike on cetacean populations*

The significance of vessel strike in the overall survival and recovery of large whale species is difficult to quantify. This is particularly true given the lack of information on interactions between whales and vessels outside of U.S. waters in the North Pacific Ocean. With the information available, we know interactions occur, but available literature does not allow us to estimate their significance to most whale species in the Action Area.

The one exception is the blue whale (Monnahan et al. 2014a). Monnahan et al. (2014) used a population dynamics model to assess the trends and status of Eastern North Pacific blue whales, and the effects of ship strikes. The authors estimate the ENP blue whale population is currently at 97 percent carrying capacity, and that it would take an 11-fold increase in the current number of vessels for the population to have a 50 percent chance of being below its maximum level of productivity, and thus depleted due to ship strike.

5.2.2 **Pinnipeds**

Due to their agility, vessel strike is not known to be a primary threat to Guadalupe fur and Hawaiian monk seals. However, strikes may occur and lead to serious injury or mortality. Since 1986, two Hawaiian monk seals with possible vessel collision injuries (non-fatal) have been reported (NMFS 2009d). Additionally, close encounters with vessels not resulting in strike may result in behavioral responses such as startle response, diving, and evasive behaviors until the vessel is no longer present. Given Guadalupe fur and Hawaiian monk seals' mobility and the lack of information indicating vessels frequently strike these species, we do not believe vessel strike is a significant threat to the survival and recovery of ESA-listed pinnipeds in the Action Area.

5.2.3 **Sea Turtles**

Vessel strike of sea turtles is poorly studied, but has the potential to be highly significant (Work et al. 2010). Sea turtles must surface to breath and several species are known to bask at the surface for long periods. Sea turtles are not able to move out of the way of vessels moving at more than 4 km/hr; most vessels move far faster than this in open water (Hazel et al. 2007; Work et al. 2010).

Not all struck sea turtles are likely to strand (NMFS 2008b). Based on an observed annual average of 8 green sea turtles stranded in the Main Hawaiian Islands between 1982 and 2007 (as compiled from the Hawaii Sea Turtle Stranding Database), and after applying a correction factor for those that do not strand, NMFS estimates 25 to 50 green sea turtles are killed by vessel strike annually in the Main Hawaiian Islands (NMFS 2008b). A total of two hawksbill sea turtles were observed stranded with obvious boat strike injuries in the Main Hawaiian Islands between 1982 and 2008 (PIFSC 2008). The majority of strandings are likely the result of strikes with relatively small, but high speed fishing boats making thousands of trips through Hawaiian nearshore waters annually. The frequency of vessel strike in open ocean waters surrounding Hawaii is much less clear. It is assumed if an animal is struck in waters further from shore, it is less likely to strand and be documented. There has been one recent report of a stranded turtle in Hawaii that appeared as though it may have been struck by a large propeller (such as those used by some Navy vessels) (D. Hubner, personal communication, December 4, 2014). However, it is more likely turtles struck by large propellers would not strand (D. Hubner, personal communication, December 4, 2014) because the damage to the carcass would be so extensive as to facilitate sinking or consumption by scavengers.

There is not a high level of sea turtle stranding data on the U.S. West Coast (R. LeRoux, personal communication, December 5, 2014). This does not necessarily indicate vessel strike is less common off the U.S. West Coast versus Hawaii. Ocean currents, vessel sizes, or other factors may simply affect the likelihood a struck turtle will strand. Regardless, this lack of stranding data makes estimating the frequency of sea turtle vessel strike off the U.S. West Coast difficult. The only turtles observed stranded in the Action Area since 1990 occurred within San Diego Bay, where a population of green sea turtles resides. Between 1990 and 2014, 10 green sea turtle strandings were observed with evidence of boat collision (R. LeRoux, personal communication, December 5, 2014). No other sea turtle species have stranded near or in the SOCAL Range Complex that have had evidence of boat strike (R. LeRoux, personal communication, December 5, 2014).

5.2.3.1 *Impact of vessel strike on sea turtle populations*

Similar to the impact of vessel strike on large whales, the significance of vessel strike in the overall survival and recovery of sea turtles species is difficult to quantify. With the information available, we know interactions occur, but available literature does not allow us to estimate their significance to sea turtle species in the Action Area.

5.3 Ambient and Anthropogenic Noise

Due to their use of sound in navigating, locating prey, mating, and communicating, marine organisms may be expected to choose their locations and modify their behavior based, in part, on natural and anthropogenic background noise. Noise in the ocean is the result of both natural and anthropogenic sources. Natural sources of noise include processes such as earthquakes, wind-driven waves, rainfall, bio-acoustic sound generation, and thermal agitation of the seawater. Anthropogenic noise is generated by a variety of activities, including shipping; oil and gas exploration, development, and production (e.g., air-guns, ships, oil drilling); naval operations (e.g., military sonars, communications, and explosions); fishing (e.g., commercial/civilian sonars, acoustic deterrent, and harassment devices); research (e.g., air-guns, sonars, telemetry, communication, and navigation); and other activities such as construction, icebreaking, and recreational boating. Sources of anthropogenic noise are becoming more pervasive, increasing oceanic background noise levels as well as peak sound intensity levels. Many anthropogenic sources of noise are located along shipping routes and encompass coastal and continental shelf waters, areas that represent important marine habitat.

5.3.1 Deep Water Ambient Noise

Urick (1983) provided a discussion of the ambient noise spectrum expected in the deep ocean (e.g., offshore habitats below the surface). Shipping, seismic activity, and weather are primary causes of deep-water ambient noise. Noise levels between 20 Hz and 500 Hz appear to be dominated by distant shipping noise. Above 300 Hz, the level of wind-related noise occasionally exceeds shipping noise. Wind, wave, and precipitation noise originating close to the point of measurement dominate frequencies from 500 Hz to 50,000 Hz. The frequency spectrum and level of ambient noise 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 Hz 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. Underwater ambient noise within the SOCAL Range Complex has higher than average noise (78 to 86 dB) at low frequencies (100 Hz to 500 Hz), owing to the dominance of ship noise at frequencies below 100 Hz and local wind and waves above 100 Hz (Hildebrand et al. 2012). In addition, there is monthly variation including noise peaks at 15 Hz to 30 Hz and also 47 Hz related to the presence of blue and fin whale calls (Hildebrand et al. 2012).

5.3.2 Shallow Water Ambient Noise

In contrast to deep water, ambient noise levels in shallow waters (i.e., surface waters, 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 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.

5.3.3 Anthropogenic Sources

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 2003; Richardson and Wursig 1995). McDonald observed an increase in low-frequency noise of 10 to 12 dB over 39 years at a site off the southern California coast (2006a), (MMC 2007). A variety of anthropogenic noise sources have been identified in the Action Area, including vessel noise from shipping and other activities, military training and testing (addressed in section 5.9 below), and seismic survey work associated with research and the oil and gas industry.

5.3.3.1 *Vessel noise and commercial shipping*

Much of the increase in noise in the ocean environment is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003) (Hildebrand 2009) (McKenna et al. 2012). Shipping constitutes a major source of low-frequency noise in the ocean, particularly in the Northern Hemisphere where the majority of ship traffic occurs (Figure 8). At frequencies below 300 Hz, ambient noise levels are elevated by 15 to 20 dB when exposed to sounds from ships at a distance (McKenna et al. 2013). Analysis of noise from ships revealed that their propulsion systems are a dominant source of radiated underwater noise at frequencies <200 Hz (Ross 1976). Additional sources of ship noise include rotational and reciprocating machinery that produces tones and pulses at a constant rate.

Individual vessels produce unique acoustic signatures, although these signatures may change with ship speed, vessel load, and activities that may be taking place on the vessel. Peak spectral levels for individual commercial ships are in the frequency band of 10 Hz to 50 Hz and range from 195 dB re $\mu Pa^2/Hz$ at 1 m for fast-moving (>20 knots) supertankers to 140 dB re $\mu Pa^2/Hz$ at 1 m for small fishing vessels (NRC 2003). Small boats with outboard or inboard engines produce sound that is generally highest in the mid-frequency (1 kHz to 5 kHz) range and at moderate (150 to 180 dB re 1 μPa @ 1 m) source levels (Erbe 2002b) (Gabriele et al. 2003) (Kipple and Gabriele 2004). On average, noise levels are higher for the larger vessels, and increased vessel speeds resulted in higher noise levels.

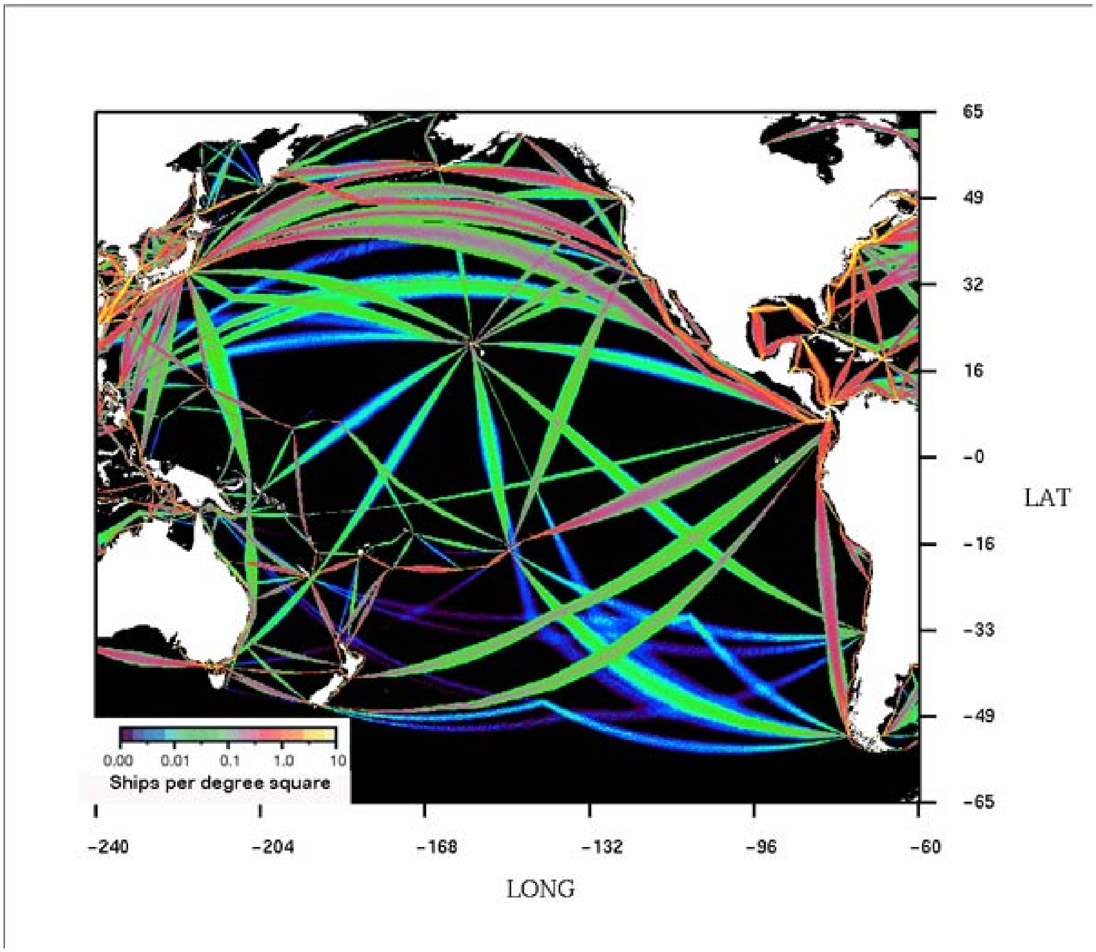

Figure 8. Estimated number of ships per 1 degree latitude by 1 degree longitude for the Pacific Ocean. The estimate is derived from the Historical Temporal Shipping (HITS–IV) distribution, derived from a United States Naval Oceanographic Office shipping traffic database. Courtesy of R. Heitmeyer (U.S. Naval Research Laboratory). (Adapted from (Hildebrand 2009).

Measurements made over the period 1950 through1970 indicated low-frequency (50 Hz) ship traffic noise in the eastern North Pacific and western North Atlantic Oceans was increasing by 0.55 dB per year.

Comparisons of data from 1963 to 1965 and 1994 to 2007 indicate low-frequency ship traffic noise (25 Hz to 50 Hz) at sites in Southern California increased by 8 to 10 dB (about 0.3 dB per year) (Chapman and Price 2011). Data obtained in the northeast Pacific from 1978 to 1986 suggest the 0.55 dB/year increase seen in the early data continued to around 1980, but then slowed to about 0.2 dB/year (Chapman and Price 2011).

The scientific community recognizes the addition of anthropogenic sound to the marine environment as a stressor that could possibly harm marine animals or significantly interfere with their normal activities (NRC 2005). The species considered in this Opinion may be impacted by noise in various ways. Once detected, some sounds may produce a behavioral response, including but not limited to, changes in habitat to avoid areas of higher noise levels, changes in diving behavior, or changes in vocalization (MMC 2007).

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 and construction (Richardson et al. 1995). Most observations have been limited to short term behavioral responses, which included temporary cessation of feeding, resting, or social interactions, however, habitat abandonment can lead to more long-term effects which may have implications at the population level. Masking may also occur, in which an animal may not be able to detect, interpret, and/or respond to biologically relevant sounds. Masking can reduce the range of communication, particularly long-range communication, such as that for blue and fin whales. This could have a variety of implications for an animal's fitness including, but not limited to, predator avoidance and the ability to reproduce successfully (MMC 2007). Recent scientific evidence suggests that marine mammals, including blue and fin whales, compensate for masking by changing the frequency, source level, redundancy, or timing of their signals, but the long-term implications of these adjustments are currently unknown (McDonald et al. 2006c) (Parks 2003) (Parks 2009b). Physical injury could also occur if an animal is exposed to high intensity sound of relatively short duration (e.g., exposure to airguns).

5.3.3.2 ***Seismic Surveys***

Seismic surveys are the primary means for finding and monitoring fossil fuel reserves and are also used by the scientific community (MMC 2007). Seismic surveys use an array of airguns which emit low-frequency sound into the marine environment (Hildebrand 2005) (Dragoset 2000). Numerous seismic surveys have been undertaken through the Action Area and its surrounding region over the past 35 years (Figure 9). ESA-listed marine mammals and sea turtles are long-lived individuals whose experience likely includes prior exposure to seismic sound sources. Prior exposure could lead to habituation, sensitization, or other changes to future exposure based upon prior experience. Significant attention has been paid to the potential impact of seismic airguns on ESA-listed species.

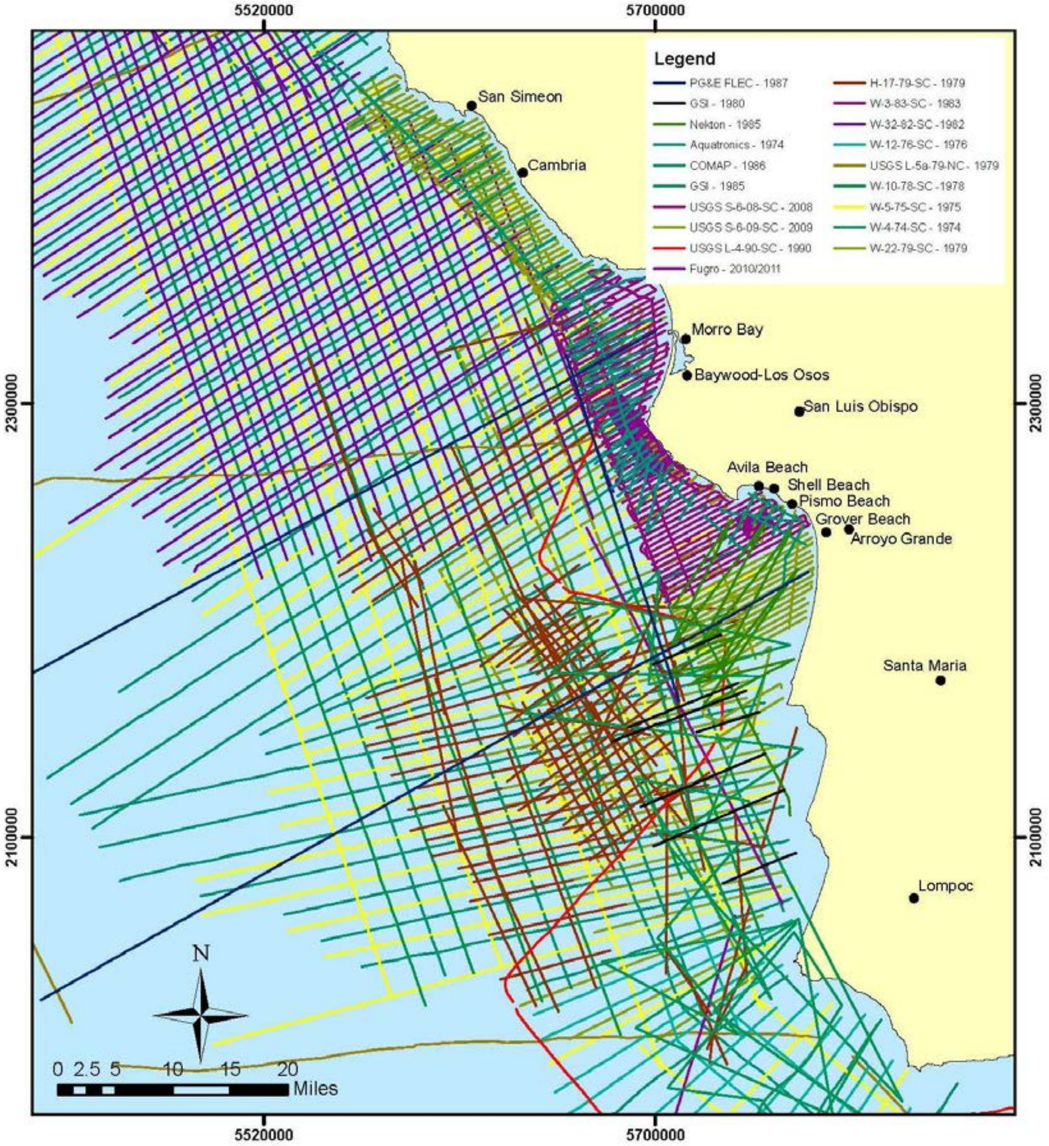

Figure 9. Seismic surveys north of the Navy's Southern California operating area.

Cetaceans

(Gordon et al. 2004) found that marine mammals can be impacted by the intense, broadband pulses produced by seismic airguns through: hearing impairment (TTS or PTS); physiological changes such as stress responses; impacting their prey; behavioral alterations such as avoidance responses, displacement, or a change in vocalizations; or through masking (obscuring sounds of interest). Responses can differ according to context, sex, age class, or species.

Generally, only the area within 500 m of the seismic vessel is observed, yet high noise levels can occur at much greater distances. (Madsen et al. 2006) discovered that in the Gulf of Mexico received levels can be as high at a distance of 12 km from a seismic survey as they are at 2 km (in both cases >160 dB peak-to-peak). Received levels, as determined from acoustic tags on sperm whales, generally fell at distances up to 8 km from the seismic survey, only to increase again at greater distances (Madsen et al. 2006).

Pinnipeds

When exposed to a single airgun or small airgun array, gray seals showed avoidance and switched from foraging to transiting behavior. They also began hauling out, possibly to escape the noise. Harbor seals exhibited a slowing of their heart rate together with dramatic avoidance behavior and stopped feeding (Thompson et al. 1998).

Sea Turtles

Marine turtles show a strong initial avoidance response to air-gun arrays at a strength of 175 dB re 1 μPa rms or greater (O'Hara and Wilcox 1990) (McCauley et al. 2000) (Lenhardt 2002). Turtles respond progressively less to successive airgun shots which may indicate reduced hearing sensitivity (TTS). One turtle experienced a TTS of 15 dB, recovering two weeks later (Lenhardt 2002). (McCauley et al. 2000) estimated that a typical airgun array operating in 100 to 120 m water depth could impact behavior at a distance of about 2 km and cause avoidance at around 1 km for marine turtles. A recent study found that 51 percent of turtles dived at or before their closest point of approach to an airgun array (DeRuiter and Doukara 2010).

5.3.3.3 ***Potential population-level impacts of anthropogenic noise***

It is clear that impacts may result from increased levels of anthropogenic-induced background noise or high intensity, short-term anthropogenic sounds. The majority of impacts will likely be short-term behavioral responses, although more serious impacts are possible. Despite the potential for these impacts to affect individual animals, information is not currently available to determine the potential population level effect of anthropogenic sound levels in the marine environment (MMC 2007) on ESA-listed marine mammals and sea turtles. More information would be required including, but not limited to, empirical data on how sound impacts an individual's growth and vital rates, how these changes impact that individual's ability to reproduce successfully, and then the relative influence of that individual's reproductive success on the population being considered. As a result, the consequences of anthropogenic sound on threatened and endangered marine mammal and sea turtles at the population or species scale remain uncertain.

5.4 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, most large whale species had been significantly depleted. Table 19 lists the reported catches of all whale species considered in this Opinion and the year in which the International Whaling Commission (IWC) issued a moratorium on harvest of that species.

Table 19. Reported Catch of Endangered Whales Considered in This Opinion, in the North Pacific Ocean

Species	Estimated total catch	Data years	Source	IWC moratorium
Blue whale	9,500 whales	1910 - 1965	(Ohsumi and Wada 1972)	1966
Fin whale	46,000 whales	1919 - 1945	(C. Allison, IWC, pers. comm.; cited in : Caretta et al. 2014)	1976
Humpback whale	15,000 whales	1919 - 1987	(Tonnessen and Johnsen 1982b); C. Allison, IWC unpubl. Data; cited in: Caretta et al. 2014)	1966
North Pacific right whale	37,000 whales	1839 - 1909	(Scarff 2001)	1946
	1,031 whales	1900 - current	(Allen and Angliss 2013c)	
Sei whale	61,500 whales	1947 - 1987	(C Allison, IWC, pers. comm.(Allison 2007)	1976
Sperm whale	258,000 whales	1947 - 1987	(C. Allison, IWC, pers. comm.; cited in: Caretta et al. 2014)	1988

These whaling numbers represent minimum catches, as illegal or underreported catches are not included. For example, recently uncovered Union of Soviet Socialist Republics catch records indicate extensive illegal whaling activity between 1948 and 1979, with a harvest totalling 157,680 sperm whales in the North Pacific Ocean (Ivashchenko et al. 2014). Of these, only 132,505 were reported by the USSR to the Bureau of International Whaling Statistics. Additionally, despite the moratorium on large-scale commercial whaling, catch of some of these species still occurs in the Pacific Ocean whether it be under objection of the IWC, for aboriginal subsistence purposes, or under IWC special permit (Table 20). Although these fisheries operate outside of the Action Area, some of the whales killed in these fisheries are likely part of the same populations of whales occurring within the Action Area for this consultation.

Table 20. Catches taken in the Pacific Ocean by commercial, aboriginal, and scientific permit whaling since 1985. Note that the large majority of these catches were taken in the Northwest Pacific Ocean by either Japan or Russia (USSR prior to 1992). Data compiled from the International Whaling Commission website (iwc.int/home; accessed on January 24, 2015).

Year	Sperm whale	Gray whale	Sei whale
1985	0	170	0
1986	200	171	0
1987	188	159	0
1988	0	151	0
1989	0	180	0

Year	Sperm whale	Gray whale	Sei whale
1990	0	162	0
1991	0	169	0
1992	0	0	0
1993	0	0	0
1994	0	44	0
1995	0	92	0
1996	0	43	0
1997	0	79	0
1998	0	125	0
1999	0	124	0
2000	5	115	0
2001	8	112	1
2002	5	131	40
2003	10	128	50
2004	3	111	100
2005	5	124	100
2006	6	134	101
2007	3	132	100
2008	2	130	100
2009	1	116	101
2010	3	118	100
2011	1	128	96
2012	3	143	100
2013	1	127	100
Totals	444	3418	1089

Historically, commercial whaling caused all of the large whale species to decline to the point where they faced extinction risks high enough to list them as endangered species. Since the end of large-scale commercial whaling, the primary threat to these species has been eliminated. However, as described in greater detail in the *Status of Listed Resources* section of this Opinion, all whale species have not recovered from those historic declines. Scientists cannot determine if those initial declines continue to influence current populations of most large whale species in the North Pacific. For example, the North Pacific right and Western North Pacific gray whales have not recovered from the effects of commercial whaling and continue to face very high risks of extinction because of their small population sizes and low population growth rates. In contrast, species such as humpack and blue whale have increased substantially from post-whaling population levels and appear to be recovering despite the impacts of ship strikes, interactions with fishing gear, and increased levels of ambient sound in the Pacific Ocean.

5.5 Fisheries Interactions

Marine mammals and sea turtles may be impacted by fisheries through entrapment or entanglement in actively fished gear, or may be impacted through entanglement in, or ingestion of, derelict fishing gear. Additionally, some species considered in this Opinion may be impacted indirectly if a fishery reduces the available prey base for higher trophic level organisms (e.g., marine mammals). Due to their highly migratory nature, many species considered in this Opinion have the potential to interact with fisheries both in and outside of the Action Area. Assessing the impact of fisheries on such species is difficult, due to the large number of fisheries that may interact with the animals, and the inadequate protected species monitoring that occurs in many of those fisheries.

The federally managed U.S. commercial fisheries of primary concern that occur in the Action Area are the California pelagic longline, California set net, California/Oregon drift gillnet, and Hawaii pelagic longline fisheries. Other federally managed fisheries (commercial and recreational), as well as state managed commercial and recreational fisheries may also interact with marine mammals and sea turtles in the Action Area. NMFS issues limited numbers of Exempted Fishing Permits, which allow experimental fisheries, exempt from usual fishing regulations. For a more comprehensive list of U.S. commercial fisheries that may interact with marine mammals and sea turtles in the Action Area see National Marine Fisheries Service (NMFS 2013b), Appendix 3. Additionally, foreign fishing fleets may operate within the portion of the Action Area that extends beyond 200 nm from the U.S. coast (the U.S. EEZ). For a more comprehensive description of many of the fisheries that may interact with the species considered in this Opinion, both in and outside of the Action Area (NMFS 2004) (NMFS 2011b).

5.5.1.1 *Entanglement and Entrapment*

The following sections discuss the *Environmental Baseline* of entanglement and entrapment stressors of fisheries in the Action Area.

5.5.1.1.1 Cetaceans

The vast majority of documented cases of baleen whale entanglements with fishing gear are from actively fished gear (NOAA 2014). Entanglement in fishing gear can result in serious injury and mortality to cetaceans. For example, between 2007 and 2011, 16 documented humpback whale interactions occurred with pot and trap fisheries off the U.S. west coast, and in all instances, the whale either died or was seriously injured. During the same time period and in the same area, gill nets and unidentified fisheries accounted for 10 documented interactions with humpback whales, with one mortality and nine serious injuries (Carretta et al. 2013b). From November 2009 through April 2010, the Hawaii Whale Entanglement Response Network received 32 reports of entangled humpback whales from fishing gear including longline, monofilament (hook and line), and local crab pot (trap) gear (Navy 2013c). Reports of fin whale entanglement are less common than for humpbacks off the U.S. west coast. Only one fin whale death has been observed in fisheries off the California coast (observed in the swordfish drift gillnet fishery) since 1990. In Hawaii, the two longline fisheries that may interact with large marine mammals (the deep-set longline fishery and the shallow-set longline fishery) did not document a fin whale interaction between 2007 and 2011 (Bradford and Forney 2013; McCracken 2013). Similar to fin whales, blue whales do not often interact with fishing gear in the Action Area; no fishery mortality has been observed in commercial fisheries off the coast of California since 1990 (Caretta et al. 2014), and no documented blue whale interactions with Hawaii commercial fisheries occurred from 2007 through 2011 (Carretta et al. 2014). Off the U.S. west coast, the fishery that is most likely to directly take sperm whales is the California drift gillnet fishery (SAR Report for sperm whales). Since 1990, in over 8,000 observed drift gillnet sets off the U.S. west coast, a total of 10 sperm whales have been observed entangled (Carretta and Enriquez 2012), though it is thought that when pingers are used as a deterrent, the risk of sperm whale entanglement (and cetacean entanglement in general) decreases (Barlow and Cameron 2003) (Carretta and Barlow 2008). In Hawaii longline fisheries from 2007 through 2011, one sperm whale was observed hooked or entangled in the gear. The whale was cut free, but the hook, 0.5m wire leader, 45g weight, 12m of branchline, and 25 to 30 ft of mainline likely remained attached. There are no records of fisheries related incidental mortality of eastern North Pacific right whales (Carretta et al. 2014). Similarly, no serious injuries or mortalities were observed for sei whales off the U.S. West Coast between 2004 and 2008. Sei whale take reduction is likely the result of the 1997 Take Reduction Plan for the offshore drift gillnet fishery ((Carretta and Enriquez 2012)). In Hawaii, one sei whale was observed entangled in fishing gear in 2011, though the source of the line could not be determined. The whale was seriously injured as a result of the interaction (Bradford and Lyman 2013). No other fishery interactions were documented in Hawaii with sei whales between 2007 and 2011. Carretta (2013c) estimated less than one (0.5, CV = 1.7) individual per year from the MHI insular false killer whale stock are killed or seriously injured during the course of fishing operations in the Hawaiian EEZ.

Mortality and serious injury numbers are minimum estimates as some interactions go unobserved. For example, whales may swim away with portions of the net (Carretta et al. 2014), not allowing fishery observers or fishers to document the interaction (Carretta et al. 2014). Additionally, since cetaceans occurring in the Action Area are migratory, these populations are likely to interact with fisheries and derelict gear from outside the Action Area. For example, many of the humpback whales that occur in the Action Area migrate to and from Alaska. From 2003 to 2007, an average of 3.4 humpback whales per year were seriously injured or killed due to entanglements with commercial fishing gear in Alaskan waters (Allen and Angliss 2013b).

5.5.1.1.2 Sea turtles

A significant number of sea turtles are killed or injured in fisheries worldwide each year. Finkbeiner et al. (2011) estimated current annual sea turtle bycatch estimates in U.S. fisheries of 137,800 individuals, with 4,600 mortalities. In total for the U.S. Pacific commercial fleet, they estimate 50 annual loggerhead sea turtle interactions with 20 mortalities, 30 annual olive ridley interactions with 20 mortalities, 30 annual leatherback interactions with 10 mortalities, and 10 annual green sea turtle interactions with 10 mortalities (Finkbeiner et al. 2011). The majority of sea turtle interactions with the Pacific fleet were estimated to occur in the Hawaii Pelagic Longline fishery, with 46 loggerhead interactions, 26 olive ridley interactions, 23 leatherback interactions, and 11 green sea turtle interactions per year. These rates are significantly lower than those prior to the implementation of numerous fisheries regulations, partially aimed at addressing the growing concern of sea turtle bycatch and bycatch mortality in U.S. fisheries. Prior to the implementation of these regulations, Finkbeiner et al. (2011) estimate 400 annual loggerhead sea turtle interactions with 200 mortalities, 100 annual olive ridley interactions with 70 mortalities, 100 annual leatherback interactions with 50 mortalities, and 40 annual green sea turtle interactions with 20 mortalities (Finkbeiner et al. 2011).

More recently, Caretta et al (2014b) reported on sea turtle bycatch from four federally managed fisheries in California in 2012. From 83 sets during 15 fishing trips (18.7 percent observer coverage) in the California swordfish and thresher shark drift gillnet fishery, only one leatherback sea turtle was observed (which was released alive). No sea turtle bycatch was observed in the California halibut and white seabass set gillnet, California yellowtail, barracuda, and white seabass drift gillnet, and California pelagic longline fisheries, with 5.5 percent, 0.7 percent, and 100 percent observer coverage, respectively. In the Hawaii pelagic longline fisheries (shallow and deep-set combined) from 2005 to 2014, bycatch resulted in mean mortality rates of two loggerhead, six leatherback, 27 olive ridley, and three green sea turtles per year (NMFS 2014).

As mentioned previously, sea turtles are highly migratory organisms and are likely to interact with other fisheries outside of the Action Area. For example, bycatch continues to be reported in gillnet and longline fisheries operating in “hotspot” areas where loggerheads are known to congregate, such as off Baja California (Peckham et al. 2007a). Additionally, bycatch in coastal pound net fisheries in Japan is another source of mortality (Ishihara 2007). Wallace et al. (2010) compiled a comprehensive database of reported data on marine turtle bycatch in gillnet, longline, and trawl fisheries worldwide from 1990 to 2008 and estimated a total bycatch of ~85,000 turtles over that time period. However, the authors suggest this estimate likely underestimates the true total by at least two orders of magnitude, due to a small percentage of fishing effort observed and reported, particularly from small-scale fisheries (Wallace et al. 2010).

5.5.1.1.3 Pinnipeds

Monk seals are also subject to entanglement in fishing gear. The species may become entangled in gear that originated outside of the Hawaiian archipelago, or may interact (become entangled or hooked) with actively fished gear in the Hawaiian Islands. The species is also known to take baited hooks or hooked fish, with numerous such cases observed each year (Carretta et al. 2014). The nearshore gillnet fishery, managed by the State of Hawaii, is of particular concern, as three seals have recently been confirmed dead in these gillnets (2006, 2007, 2010). Between 1982 and 2006, a total of 118 monk seals were observed entangled in fishing gear, with 32 serious injuries and 7 mortalities. From 1982 to 2000, there was an estimated minimum rate of 2.3 serious injuries or deaths per year attributable to fishery related marine debris (NMFS 2007c).

The latest stock assessment report for Guadalupe fur seals indicated that, while entanglement in drift and set gillnet fisheries may cause incidental mortality of the species, there have been no reports of fisheries related mortalities or injuries of Guadalupe fur seals in the United States or Mexico from actively fished gear (Carretta et al. 2014). However, the report notes that juvenile Guadalupe fur seals have been observed in central and northern California with net abrasions around the neck, and attached fish hooks, monofilament line, and polyfilament string (Hanni et al. 1997b), suggesting that derelict fishing gear does interact with the species. Entanglement in debris was evident with 15.4 percent of the Guadalupe fur seals taken for rehabilitation from 1986 to 1998 (Goldstein et al. 1999; Program 2014).

5.5.1.2 ***Competition for Prey Base***

Fisheries may impact marine mammals and sea turtles by harvesting their prey. For example, because MHI insular false killer whales feed on large prey at the top of the food chain (e.g., squid, tuna), they may be impacted by competition with fisheries (Research 2010). Limited prey availability may be restricting the recovery of the northwestern Hawaiian Islands monk seals (Navy 2013c) and it has been suggested that this may partially have resulted from fishing pressure.

5.6 Marine Debris

Anthropogenic marine debris is prevalent throughout the Action Area, originating from a variety of oceanic and land-based sources. Debris can be introduced into the marine environment by its improper disposal, accidental loss, or natural disasters (Watters et al. 2010b), and can include plastics, glass, derelict fishing gear, derelict vessels, or military expendable materials. Though debris abundance is well understood in shallow-water, shoreline, and surface water habitats, debris can also settle into deepwater benthic habitats in the Action Area (Watters et al. 2010b). Marine debris affects marine habitats and marine life worldwide, primarily by entangling or choking individuals that encounter it. Despite debris removal and outreach to heighten public awareness, marine debris in the environment has not been reduced (Academies 2008). Though marine debris is abundant on deepwater seafloor habitats in the Action Area, interactions with the species considered in this Opinion are much more likely at the sea surface, in the water column, or on shoreline habitats (Laist 1997b) (Watters et al. 2010a).

As noted above in the fisheries interactions section of the Environmental Baseline, entanglement or entrapment in derelict fishing gear can pose a threat to many of the species considered in this Opinion. For example, in the northwestern Hawaiian Islands, derelict fishing gear has been identified as a top threat to the monk seal (Donohue and Foley 2007). Fur seals (e.g., Guadalupe fur seal) appear to be particularly attracted to floating marine debris and consequently, suffer a high rate of entanglement in derelict fishing lines and nets (Derraik 2002). A young pup may become so entangled that its body becomes constricted as it grows. Eventually, this can lead to death via strangulation or severing of the arteries (Derraik 2002). The vast majority of reported cases of entangled baleen whales in the U.S. are humpbacks, with most of these interactions likely involving actively fished, rather than derelict, gear (Program 2014). In Alaska, only 24 percent of documented entanglements were from unknown sources, possibly including marine debris, with the rest of the cases being from a known fishery and likely being actively fished (Jensen et al. 2009a). As noted previously, it is likely that some animals interact with fishing gear outside of the Action Area, become entangled, and bring that gear with them when they migrate to the Action Area. For example, 10 humpbacks with entangled gear observed in Hawaii have also been sighted with entangled gear in Alaska, with one animal traveling over 2,450 nautical miles with gear attached (Lyman 2012).

Anthropogenic marine debris can also be accidentally consumed while foraging. Recently weaned juveniles, who are investigating multiple types of prey items, may be particularly vulnerable to ingesting non-food items (Baird and Hooker 2000) (Schuyler et al. 2013). This can have significant implications for an animal's survival, potentially leading to starvation or malnutrition, or internal injuries from consumption. Parker et al. (2005) conducted a diet analysis of 52 loggerhead sea turtles collected as bycatch from 1990 to 1992 in the high seas drift gillnet fishery in the central north Pacific. The authors found that 34.6 percent of the individuals sampled had anthropogenic debris in their stomachs (e.g., plastic, Styrofoam, paper, rubber, etc). Similarly, a study of green sea turtles found that 61 percent of those observed stranded had ingested some form of marine debris, including rope or string, which may have originated from fishing gear (Bugoni et al. 2001). In 2008, two sperm whales stranded along the California coast, with an assortment of fishing related debris (e.g., net scraps, rope) and other plastics inside their stomachs (Jacobsen et al. 2010a). One whale was emaciated, and the other had a ruptured stomach. It was suspected that gastric impaction was the cause of both deaths. (Jacobsen et al. 2010a) speculated that the debris likely accumulated over many years, possibly in the North Pacific gyre that would carry derelict Asian fishing gear into eastern Pacific waters.

5.7 Scientific Research

Scientific research permits issued by the NMFS currently authorize studies on listed species in the North Pacific Ocean, some of which extend into portions of the Action Area. Authorized research on ESA-listed whales includes close vessel and aerial approaches, biopsy sampling, tagging, ultrasound, exposure to acoustic activities, and breath sampling. Research activities involve non-lethal "takes" of these whales. Sea turtle research includes capture, handling, restraint, tagging, biopsy, blood sampling, lavage, ultrasound, and tetracycline injection. Lethal take of male Hawaiian monk seals has been authorized in specific instances of mobbing. The removal of specific males involved in mobbing preserves the health of female and young individuals that will provide greater contributions to the survival and recovery of the species. Table 21 describes the cumulative number of takes for each listed species in the Action Area authorized by scientific research permits.

Table 21. Authorized takes of listed whales, pinnipeds, and sea turtles in the Pacific Ocean under the Endangered Species Act and the Marine Mammal Protection Act.

Species	2009-2016 lethal take	2009-2016 sub-lethal take
Blue Whale	0	151,037
Fin Whale	0	222,093
Western North Pacific Gray Whale	0	1,264
Sei Whale	0	63,784
Humpback Whale	0	472,963
North Pacific Right Whale	0	12,762
Sperm Whale	0	195,287
Main Hawaiian Island Insular False Killer Whale	0	17,261
Guadalupe Fur Seal	1	15,068
Hawaiian Monk Seal	173	49,967
Green Turtle	0	10,142
Hawksbill Turtle	0	1,615
Leatherback Turtle	0	1,480
Loggerhead Turtle	0	631
Olive Ridley Turtle	0	2,861

Several investigators have reported behavioral responses that suggest individual whales might experience stress responses to scientific research activities. Baker et al. (1983) described two responses of whales to vessels, including: (1) “horizontal avoidance” of vessels 2,000 to 4,000 meters away, characterized by faster swimming and fewer long dives; and (2) “vertical avoidance” of vessels from 0 to 2,000 meters away during which whales swam more slowly, but spent more time submerged. (Watkins 1981a) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startle reaction, and moving away from the vessel with strong fluke motions. Other researchers have noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. (Jahoda et al. 2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They concluded close vessel approaches caused whales to stop feeding and swim away from the approaching vessel. The whales also tended to reduce the time they spent at surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. In their study, whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 meters from whales and having approaching vessels move at low speeds to reduce visible reactions.

5.8 Whale Watching

As of 2010, commercial whale watching was a $1 billion global industry per year (Lambert et al. 2010). Private vessels may partake in this activity as well. Whale watching, particularly of humpback whales, is extensive in Hawaiian waters during the winter. Numerous commercial operations are also based out of San Diego, indicating that whale watching occurs within both the Hawaii and California portions of the HSTT Action Area. NMFS has issued certain regulations and guidelines relevant to whale watching. For example, under 50 CFR 224.103, except as authorized under the MMPA and ESA, federal regulations prohibit approaching (by any means) humpback whales within 100 yards when on or in the water, and within 1,000 feet when operating an aircraft within 200 nautical miles of the Islands of Hawaii. In California, NMFS' whale watching guidelines include recommendations to: 1) be alert and avoid disturbing whales and changing their normal behavior; 2) always attempt to stay 100 yards away from a whale; and 3) if a situation arises where you cannot avoid a whale by 100 yards, do not move into the path of a whale, move faster than a whale, make erratic speed or directional changes unless you are avoiding a collision with a whale, get between two whales, chase any whales, or feed any whales (NMFS 2011a). As noted previously, many of the cetaceans considered in this Opinion are highly migratory, so may also be exposed to whale watching activity occurring outside of the Action Area.

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. Whale watching has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close or strikes the whale. Preferred habitats may be abandoned if disturbance levels are too high. Animals may also become more vulnerable to vessel strikes if they habituate to vessel traffic (Swingle et al. 1993; Wiley et al. 1995). Several investigators have studied the short term effects of whale watch vessels on marine mammals (Amaral and Carlson 2005; Au and Green 2000a; Corkeron 1995; Erbe 2002b; Félix 2001; Magalhaes et al. 2002; Richter et al. 2003a; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002d). 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. In some circumstances, the whales 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. Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from whale watching (NMFS 2006a). Christiansen et al. (2014) estimated the cumulative time minke whales spent with whale watching boats in Iceland to assess the biological significance of whale watching disturbances and found that, though some whales were repeatedly exposed to whale watching boats throughout the feeding season, the estimated cumulative time they spent with boats was very low. The authors suggested that the whale watching industry, in its current state, is likely not having any long-term negative effects on vital rates Christiansen et al. (2014). To our knowledge, similar studies have not been conducted in California or Hawaii.

5.9 Ongoing Military Training and Testing Activities

The majority of the training and testing activities the Navy conducts in the HSTT Action Area and proposes to continue to conduct are similar, if not identical, to activities that have been occurring in the same locations for decades. Ongoing U.S. Navy training and testing activities in the HSTT Action Area are discussed here as part of the baseline.

5.9.1 The U.S. Navy's Recent Pacific Fleet Training in the Hawaii Training Range Complex

There were 15 individual major training events that took place in the Hawaii Range Complex from January 8, 2009 to December 25, 2014. These MTEs are summarized in Table 22 below.

Table 22. Hawaii Range Complex Major Training Exercise Summary

Exercise Type	8 Jan 2009 – 14 Jan 2010	15 Jan 2010 – 14 Jan 2011	15 Jan 2011 – 14 Jan 2012	15 Jan 2012 – 14 Jan 2013	15 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
USWEX	2	0	6	1	1	2	12
RIMPAC	0	1	0	1	0	1	3
Total	**2**	**1**	**6**	**2**	**1**	**3**	**15**

Notes: RIMPAC = Rim of the Pacific, USWEX = Undersea Warfare Exercise

5.9.1.1 *Marine Animal Sightings*

There were 305 reported sightings of at least 1,799 marine mammals and sea turtles during MTEs in the Hawaii Range Complex from January 8, 2009 to December 25, 2014. These sightings are summarized in Table 23**.**

Table 23. Hawaii Range Complex Sighted Marine Mammals and Sea Turtles

Marine Animal Types	8 Jan 2009 – 14 Jan 2010	15 Jan 2010 – 14 Jan 2011	15 Jan 2011 – 14 Jan 2012	15 Jan 2012 – 14 Jan 2013	15 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
Estimated Number of Animals Sighted While Sonar Active							
Dolphin	0	3	45	208	0	50	306
Whale	3	0	12	13	0	1	29
Pinniped	0	0	0	0	0	0	0
Turtle	0	0	0	0	0	0	0
Unknown	1	0	1	3	5	1	11
Subtotal while Active	4	3	58	224	5	52	346
Estimated Number of Animals Sighted While Sonar Passive							
Dolphin	0	255	33	228	0	262	778
Whale	133	34	58	155	3	140	523
Pinniped	1	0	0	1	0	0	2
Turtle	0	25	0	1	0	0	26
Unknown	7	0	4	1	1	91	104

Marine Animal Types	8 Jan 2009 – 14 Jan 2010	15 Jan 2010 – 14 Jan 2011	15 Jan 2011 – 14 Jan 2012	15 Jan 2012 – 14 Jan 2013	15 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
Subtotal while Passive	141	314	95	386	4	493	1,433
Total	**145**	**317**	**153**	**610**	**9**	**545**	**1,779**

5.9.1.2 *Hawaii Range Complex Mitigation Events*

There were 26 total mitigation events where sonar was powered down, shut down, or delayed due to the sighting of marine mammals or sea turtles during MTEs from January 8, 2009 to December 25, 2014.

5.9.2 The U.S. Navy's Pacific Fleet Training in the SOCALTraining Range Complex

SOCAL Range Complex Major Training Exercise Summary from January 22, 2009 to December 25, 2014. There were 38 individual MTEs (Table 24) that took place in the SOCAL Range Complex during this reporting period.

Table 24. SOCAL Range Complex Major Training Exercises 22 Jan 2009 to 25 Dec 2014.

Exercise Type	22 Jan 2009 – 21 Jan 2010	22 Jan 2010 –21 Jan 2011	22 Jan 2011 – 21 Jan 2012	22 Jan 2012 – 21 Jan 2013	22 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
COMPTUEX	1	5	2	1	1	2	12
JTFEX	1	0	3	1	0	1	6
IAC	1	4	3	2	1	2	13
SUSTEX	3	1	1	1	1	0	7
Total	**6**	**10**	**9**	**5**	**3**	**5**	**38**

Note: COMPTUEX = Composite Training Unit Exercise; JTFEX = Joint Task Force Exercise; IAC = Integrated Anti-submarine Warfare Course; SUSTEX = Sustainment Exercise

5.9.2.1 *Marine Animal Sightings During Major Training Exercises*

There were 1,785 reported sightings of at least 18,966 marine mammals and sea turtles during MTEs in the SOCAL Range Complex from January 22, 2009 to December 25, 2014 (Table 25).

Table 25. SOCAL Range Complex marine mammals and sea turtles sighted from U.S. Navy ships and aircraft during Major Training Exercises.

Marine Animal Types	22 Jan 2009 – 21 Jan 2010	22 Jan 2010 –21 Jan 2011	22 Jan 2011 – 21 Jan 2012	22 Jan 2012 – 21 Jan 2013	22 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
Estimated Number of Animals Sighted While Sonar Active							
Dolphin	456	2,040	901	524	24	175	4,120
Whale	131	219	65	63	38	23	539
Pinniped	16	96	15	12	0	6	145
Turtle	0	1	0	0	0	0	1
Generic	9	11	15	8	1	2	46
Subtotal while Active	612	2,367	996	607	63	206	4,851
Estimated Number of Animals Sighted While Sonar Passive							
Dolphin	1,866	3,421	2,354	2,389	1,099	1,413	12,542
Whale	267	433	122	243	52	71	1,188

Marine Animal Types	22 Jan 2009 – 21 Jan 2010	22 Jan 2010 –21 Jan 2011	22 Jan 2011 – 21 Jan 2012	22 Jan 2012 – 21 Jan 2013	22 Jan 2013 – 25 Dec 2013	26 Dec 2013 – 25 Dec 2014	6-Year Total
Pinniped	133	103	16	58	26	21	357
Turtle	0	0	0	0	1	0	1
Generic	4	3	11	3	6	0	27
Subtotal while Passive	2,270	3,960	2,503	2,693	1,184	1,505	14,115
Total	**2,882**	**6,327**	**3,499**	**3,300**	**1,247**	**1,711**	**18,966**

5.9.2.2 *SOCAL Range Complex Sonar Mitigation Events*

There were 344 total mitigation events due to sighting of marine animals within prescribed mitigation zones during MTEs from January 22, 2009, to December 25, 2014. Mitigation consisted of sonar being powered down, shut down, or delayed.

5.9.3 Surveillance Towed Array Sensor System (SURTASS) Low Frequency Active (LFA) Sonar

Ongoing operations of SURTASS LFA sonar in the western and central North Pacific Ocean over the eleven-year period spanning 2002 through 2013 involved 156 completed missions conducted over 480 days during which LFA sonar was transmitted for a total of 930 hr (or about 39 of a possible 4,015 days). During those missions, only 10 marine mammals or sea turtles were visually observed, eight marine animals were detected passive-acoustically, and 149 marine mammals/animals were detected active-acoustically by the high frequency marine mammal monitoring (HF/M3) sonar system. These detections of marine animals led to suspensions/delays of LFA sonar transmissions.

The acoustic (passive and active [HF/M3]) and visual mitigation and monitoring measures have been employed aboard four SURTASS LFA sonar vessels operating in the western North Pacific Ocean since August 2012. For the 2012 through 2013 annual reporting period, 47.3 hours (hr) of LFA sonar were transmitted during 12 missions over 24.4 mission days (Table 27). In accordance with the mitigation monitoring protocol, LFA sonar was suspended 13 times due to three passive and 10 active acoustic detections of marine mammals. No ESA-listed species were observed, nor were any dead or injured marine species observed during the 2012 to 2013 reporting period. The LOAs for SURTASS LFA permit a combined total of 1,728 hr of LFA sonar transmissions in the western and central North Pacific Ocean. However, in the 2012 to 2013 LOA reporting period, all four SURTASS LFA vessels transmitted a grand total of 47.3 hr of LFA sonar or 2.7 percent of the permitted sonar transmit time (Table 27).

During the first two quarters of the current LOA reporting period, two missions were completed over 4 days, during which LFA sonar was transmitted for 9.75 hr. There were no visual, passive acoustic, or active acoustic detections of marine animals, shutdowns of the LFA sonar, or reports of dead or injured marine animals (Table 28). Only 9.75 hr for all vessels, or 0.6 percent of the permitted total sonar transmissions, have been conducted thus far (Table 28).

The results of the mitigation monitoring and its effectiveness support the Navy’s and NMFS’ conclusions that the Navy’s protocols for the use of SURTASS LFA sonar provide an effective means of minimizing risk of injury to protected marine species.

Table 26. Summary of SURTASS LFA sonar missions and mitigation measure detections from 2002 through 2nd Quarter 2014.

Year [1]	Vessel	Mission Summary			Visual Detections	Passive Acoustic Detections	Active Acoustic HF/M3 Detections	HF/M3 Unavailable[2]	Mitigation Protocol Suspensions/ Delays
		Number	Days	LFA Sonar Hours					
2002-2003	R/V *Cory Chouest*	7	34.2	82.2	0	0	3	0	3
2003-2004	R/V *Cory Chouest*	5	72.5	173.7	0	0	10	0	10
	USNS IMPECCABLE	5			0	0	6	2	8
2004-2005	R/V *Cory Chouest*	3	22.5	41.9	0	0	1	11	12
	USNS IMPECCABLE	2			0	0	1	0	1
2005-2006	R/V *Cory Chouest*	12	95.6	173.2	1	0	47	10	58
	USNS IMPECCABLE	6			2	0	3	0	5
2006-2007	R/V *Cory Chouest*	6	94.6	161.5	0	0	30	50	80
	USNS IMPECCABLE	12			1	0	0	0	1
2007-2008	R/V Cory Chouest	6	49.8	135.8	0	0	0	16	16
	USNS IMPECCABLE	8			1	0	19	7	27
2008-2009	USNS ABLE	3	23.7	32.5	1	0	1	3	5
	USNS IMPECCABLE	6			2	0	1	0	3
2009-2010	USNS ABLE	10	17.1	43.6	0	0	0	0	0
	USNS IMPECCABLE	21			1[3]	3[4]	3[4]	4	7
2010-2011	USNS ABLE	8	62.3	64.0	1[3]	1	1[3]	0	1
	USNS EFFECTIVE	1			2[5]	0	2[6]	1	4
	USNS IMPECCABLE	7			0	0	2	0	2
2011-2012	USNS ABLE	6	28.5	66.3	0[7]	1	1	0	3[8]
	USNS EFFECTIVE	5			0	0	0	0	0
	USNS IMPECCABLE	3			0	0	0	0	0
	USNS VICTORIOUS	1			0	0	0	0	0

Year [1]	Vessel	Mission Summary			Visual Detections	Passive Acoustic Detections	Active Acoustic HF/M3 Detections	HF/M3 Unavailable[2]	Mitigation Protocol Suspensions/ Delays
		Number	Days	LFA Sonar Hours					
2012-2013	USNS ABLE	3	24.4	47.3	0	0	0	0	0
	USNS EFFECTIVE	4			0	3	9	12	16
	USNS IMPECCABLE	2			0	0	0	0	12
	USNS VICTORIOUS	3			0	0	1	1	1
2013-2014 Through 2nd Quarter	USNS ABLE	0	4	9.75	0	0	0	0	0
	USNS EFFECTIVE	2			0	0	0	0	0
	USNS IMPECCABLE	0			0	0	0	0	0
	USNS VICTORIOUS	0			0	0	0	0	0
	Totals	**157**	**549.2**	**1031.8**	**12**	**8**	**141**	**117**	**275**

[1] August to August

[2] LFA sonar suspended during HF/M3 non-availability

[3] Contact made when LFA sonar not transmitting

[4] Marine mammal passive contacts verified by HF/M3 sonar system

[5] One contact confirmed with HF/M3 sonar, second was a sea turtle

[6] Only one contact confirmed visually

[7] 1 visual sighting of a marine mammal after the mission ended and LFA sonar was non-operational

[8] 1 shut down due to loss of passive acoustics system

Table 27. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the Annual LOAs and ITS Reporting Period from August 15, 2010 through August 14, 2013

ANNUAL SURTASS LFA SONAR OPERATIONS								
SURTASS LFA VESSEL	**MISSIONS**	**MISSION DURATION (DAYS)**	**LFA SONAR TRANSMISSIONS (HOURS)**	**VISUAL DETECTIONS**	**PASSIVE ACOUSTIC DETECTIONS**	**HF/M3-ACTIVE ACOUSTIC DETECTIONS**	**LFA SONAR SUSPENSIONS DUE TO DETECTIONS**	**TOTAL SUSPENSIONS OF LFA SONAR[1]**
USNS ABLE (T-AGOS 20)	3	2.5	5.4	0	0	0	0	0
USNS EFFECTIVE (T-AGOS 21)	4	12.5	22.5	0	3	9	12	16
USNS IMPECCABLE (T-AGOS 23)	2	2.5	5.2	0	0	0	0	12
USNS VICTORIOUS (T-AGOS 19)	3	6.9	14.2	0	0	1	1	1
Totals	**12**	**24.4**	**47.3**	**0**	**3**	**10**	**13**	**29**

[1] In addition to LFA sonar suspensions due to visual, passive acoustic, or active acoustic/HFM3 detections, suspensions of LFA sonar transmissions are also due to loss of the passive acoustic system, HFM3 system faults, HFM3 system artifacts, or impedance checks.

Table 28. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the First Two Quarters (August through February) of the Annual LOAs and ITS Reporting Period from August 15, 2013 through August 14, 2014

ANNUAL SURTASS LFA SONAR OPERATIONS								
SURTASS LFA VESSEL	**MISSIONS**	**MISSION DURATION (DAYS)**	**LFA SONAR TRANSMISSIONS (HOURS)**	**VISUAL DETECTIONS**	**PASSIVE ACOUSTIC DETECTIONS**	**HF/M3-ACTIVE ACOUSTIC DETECTIONS**	**LFA SONAR SUSPENSIONS DUE TO DETECTIONS**	**TOTAL SUSPENSIONS OF LFA SONAR[1]**
USNS ABLE (T-AGOS 20)	0	0	0	0	0	0	0	0
USNS EFFECTIVE (T-AGOS 21)	2	4.0	9.75	0	0	0	0	0
USNS IMPECCABLE (T-AGOS 23)	0	0	0	0	0	0	0	0
USNS VICTORIOUS (T-AGOS 19)	0	0	0	0	0	0	0	0
Totals	**2**	**4**	**9.75**	**0**	**0**	**0**	**0**	**0**

[1] In addition to LFA sonar suspensions due to visual, passive acoustic, or active acoustic/HFM3 detections, suspensions of LFA sonar transmissions are also due to loss of the passive acoustic system, HFM3 system faults, HFM3 system artifacts, or impedance checks.

Based on SURTASS LFA pre-operational and post-operational estimates, blue whales, sei whales, fin whales, humpback whales, sperm whales, main Hawaiian island insular false killer whales, Hawaiian monk seals, and sea turtles would be exposed to SURTASS LFA transmissions within the Hawaii Range Complex (Table 29 and Table 30).

Table 29. Estimated percentage ofESA-listed marine mammal stocks that may be affected by the operation of SURTASS LFA sonar, based on a single 7-day mission in each operating area (with mitigation measures applied).

Species	Marine Mammal Stock	Est. Number in Stock	Percent Stock Affected <180 dB
Hawaii North			
Blue whale	Central North Pacific	9,250	0.14
Fin whale	Hawaiian	174	3.59
Sei whale	Hawaiian	77	0.11
Humpback whale	Central North Pacific	10,103	0.09
Sperm whale	Hawaiian	6,919	3.23
Main Hawaiian Island Insular False Killer Whale	Main Hawaiian Island Insular	151	0.22
Hawaiian monk seal	Hawaiian	1,212	0.80
Hawaii South			
Blue whale	Central North Pacific	9,250	0.08
Fin whale	Hawaiian	174	2.25
Sei	Hawaiian	77	0.11
Humpback whale	Central North Pacific	10,103	0.69
Sperm whale	Hawaiian	6,919	0.62
Main Hawaiian Island Insular False Killer Whale	Main Hawaiian Island Insular	151	1.06
Hawaiian monk seal	Hawaiian	1,212	0.28

An entry of N/A in the Percent Stock Affected <180 dB indicates that no exposures of individuals of those stocks are expected to occur.

Table 30. Estimated number of ESA-Listed species that may be "taken" in the form of harassment as a result of their exposure to U.S. Navy Surveillance Towed Array Sensor System Low Frequency Active Sonar in the Hawaii Range Complex from August 2014 through August 2015 (NMFS 2013a).

Species	Hawaii North	Hawaii South
Blue Whale	14	9
Fin Whale	7	5
Western Pacific Gray Whale	0	0
Humpback Whale	10	12
Sperm Whale	224	74
North Pacific Right Whale	0	0
Sei Whale	1	2

Species	Hawaii North	Hawaii South
Main Hawaiian Island Insular False Killer Whale	2	2
Hawaiian Monk Seal	10	5

Thus far, the combination of geographic constraints, operating protocols, monitoring measures, and shut-down procedures have prevented most ESA-listed marine mammals and sea turtles from being exposed to SURTASS LFA sonar at received levels exceeding 180 dB. Accordingly, the available evidence does not indicate that the use of SURTASS LFA has any adverse population-level impact on the the species evaluated in this Opinion or has otherwise adversely impacted the environmental baseline in the Action Area.

5.10 Recovery Actions in the Action Area

Recovery is the process by which species' ecosystems are restored and threats to the species are minimized such that ESA-listed species can be self-sustaining. This section addresses ongoing recovery actions that may compensate for effects from stressors in the *Environmental Baseline* and the action assessed in this Opinion. ESA-listed species with final recovery plans are noted in Table 17. Ongoing conservation actions for ESA-listed species include:

- Monitoring the status of the Eastern North Pacific Stock (CA-OR-WA) of blue whales via shipboard surveys;
- Implementing a number of ship strike reduction measures in southern and central California;
- Placing observers onboard vessels in the California/ Oregon swordfish/ thresher shark drift gillnet fishery to monitor the take of protected species, including other marine mammals;
- Implementing marine mammal take reduction measures identified in the Pacific Offshore Cetacean Take Reduction Plan (including the use of acoustic pingers) to reduce the bycatch of blue whales and other marine mammals
- Mitigate ship strikes and respond to humpback whales in distress (see Alaska and Hawaii regulations);
- Educate whale watch vessels and boat operators on practicing safe boating around whales;
- Monitor humpbacks in U.S. waters via shipboard surveys and mark recapture studies;

- Research humpback population structure and abundance including the Structure of Populations, Levels of Abundance, and Status of Humpbacks (SPLASH) and More North Atlantic Humpbacks (MoNAH) projects as well as work done at the Hawaiian Islands Humpback Whale National Marine Sanctuary.
- Public education campaigns, including projects to reduce Hawaiian monk seal-human interactions on the MHI, are building awareness about conserving this species and habitat.

Volunteer groups are being expanded to help rescue and rehabilitate Hawaiian monk seals and prevent undue stress by keeping beachgoers away from resting animals. Direct efforts to disentangle seals and remove debris from haul-out sites have led to the removal of 492 metric tons (over 1 million lbs) of marine debris in NWHI since 1996, reducing injuries and death due to entanglement and digestion of marine debris.

5.11 Conclusion on the Impact of the Environmental Baseline

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 most of the species in this Opinion are wide ranging and subject to stressors in locations well beyond the 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 17, 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 discussed thoroughly in the *Status of Listed Resources* section of this Opinion.

6 EFFECTS OF THE ACTION ON SPECIES AND CRITICAL HABITAT

Effects of the Action means the direct and indirect effects of an action on the species or critical habitat, together with the effects of interrelated or interdependent actions and cumulative effects, that will be added to the *Environmental Baseline* (50 CFR §402.02). Indirect effects are those that are caused by the action and are later in time, but still are reasonably certain to occur. This effects analyses section begins with an overview of the risk assessment criteria for marine mammals and sea turtles. The analyses are organized by stressor and broken down by exposure and response.

The ESA does not define "harassment" nor has NMFS defined this term, pursuant to the ESA, through regulation. However, the MMPA defines "harassment" as "any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal stock in the wild or has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering" [16 U.S.C. 1362(18) (A)]. For military readiness activities, this definition of "harassment" has been amended to mean, in part, "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 behaviors are abandoned or significantly altered" (Public Law 108-136, Nov. 24 2003, 117 Stat. 1433).

As the statutory definitions are currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA or harassment pursuant to the ESA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patters, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The Navy modeled estimates calculated using the behavioral response function do not differentiate between the different types of potential behavioral reactions. Nor do the estimates provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. We therefore consider the available scientific evidence to determine the likely nature of the modeled behavioral responses and the potential fitness consequences for affected individuals. In this Opinion, we assume all modeled instances of behavioral harassment meet the following definition of harassment. "Harassment" is defined such that it corresponds to the MMPA and U.S. Fish and Wildlife Service's definitions: "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering."

This section concludes with an *Integration and Synthesis of Effects* that integrates information presented in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion with the results of the exposure and response analyses to estimate the probable risks the action poses to endangered and threatened species.

6.1 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.

6.1.1 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. 2007d). 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 (see Department of the Navy 2013), 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.

6.1.2 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. 2001; 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 U.S. 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 U.S. Department of the Navy (2013) 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.

6.1.3 **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).

6.1.4 **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 (U.S. Department of the Navy 2013d). 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.

6.1.5 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. 2001). 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. 2001)(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 recent beaked whale strandings are consistent with gas emboli and bubble-induced tissue separations (Fernandez et al. 2005; 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).

6.1.6 **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 (Jepson et al. 2003) (Hooker et al. 2012) (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 (Saunders et al. 2008) (Hooker et al. 2012). 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. 2005; 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. 2005; 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 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 Fernández et al. (2005) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals, and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream. 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 et al. (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.

6.1.7 **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 (Miller 1994). 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 10 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).

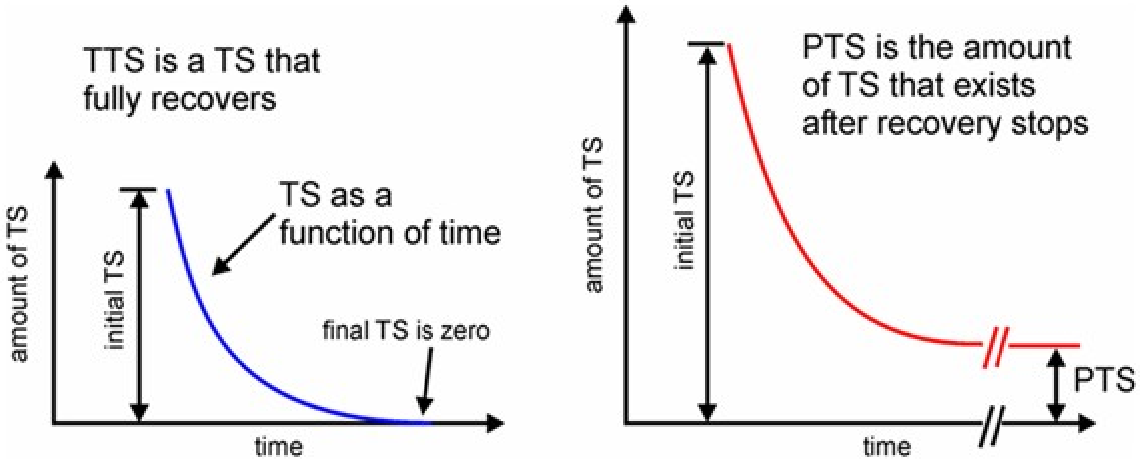

Figure 10. Two Hypothetical Threshold Shifts, Temporary and Permanent

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. 2000; Finneran et al. 2002; Lucke et al. 2009; Mooney et al. 2009a; Mooney et al. 2009b; Nachtigall et al. 2003; Schlundt et al. 2000). 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. 2000). 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 (Kryter et al. 1965; Ward 1997; Kastelein et al. 2014a).

- 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; 1959a, b). 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. 2000). 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 longterm effects of PTS on marine mammal fitness. Concievably, 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 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. 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. For additional discussion on the effects of hearing loss on marine mammals see section 6.6.1.3.4.

6.1.8 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 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 2009a) 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. 1994a), 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. Melcón 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. 2007), DeRuiter et al. (2013) 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.

6.1.9 **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. 2009a; Williams and Ashe 2006; Williams and Noren 2009; Pirotta et al. 2015). 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. 1990a) 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.

6.1.10 **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. 1995). More recent reviews (Ellison et al. 2012; Nowacek et al. 2007; Southall et al. 2009a; Southall et al. 2007d) 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. (2007) 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. 2007). 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. 2011a). Passive acoustic monitoring of beaked whales, classified as Blainville's beaked whales and Cross-seamount type beaked whales, at Pacific Missile Range Facility, Kauai, Hawaii showed statistically significant differences in dive rates, diel occurrence patterns, and spatial distribution of dives after the initiation of a training event. However, for the beaked whale dives that continued to occur during MFAS activity, differences from normal dive profiles and click rates were not detected with estimated receive levels up to 137 dB re 1 uPa while the animals were at depth during their dives (Manzano-Roth et al. 2013).

6.1.10.1 *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.

6.1.10.1.1 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. 1995b; Southall et al. 2007d; Gordon et al. 2003). While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al. 1995), 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. 1986a; Malme 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. 2007b).

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. (2012) 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.

6.1.10.1.2 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).

6.1.10.1.3 Pinnipeds

A review of behavioral reactions by pinnipeds to impulsive noise can be found in Richardson et al. (1995) and Southall et al. (2007). 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.

6.1.10.2 *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.

6.1.10.2.1 Mysticetes

Specific to U.S. 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. 2001). 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. 2001; Oleson et al. 2007; Stafford and Moore 2005), 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).Southall et al. (Southall et al. 2011a) reported that 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. (2013b) 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. (2013b) noted that behavioral responses observed were temporary and whales typically resumed normal behavior quickly after the cessation of sound exposure. Perhaps the most significant reponse 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.

Melcón et al. (2012a) 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. Despite being classified as “low-frequency hearing specialists,” 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. in press) 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.

6.1.10.2.2 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; Tyack et al. 2011b)(Southall et al. 2009b). 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. 2012; Southall et al. 2010).

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 (Defence Science and Technology Laboratory 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. 2012; Miller et al. 2014; 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 (see Miller 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, U.S. 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. 2011b). 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. 2009b).

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. 2009b). 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 (Defence Science and Technology Laboratory 2007; (Claridge and Durban 2009; McCarthy et al. 2011; Moretti 2009; Tyack et al. 2011b). 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. 2011b). Moretti et al. (2014a) 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., Scott et al. 1990; Claridge 2006; 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. 1985)(Watkins and Schevill 1975). 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 1975).

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 et al. 2001; Finneran et al. 2003a; Finneran et al. 2010; Finneran and Schlundt 2004; Schlundt et al. 2000) (Finneran 2010) (Finneran 2011). 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. 2000). 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. 2000). 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 U.S. 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. 2007). 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.

6.1.10.2.3 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.

6.1.10.3 *Behavioral Reactions of Marine Mammals to Vessels*

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 2007; Hildebrand 2005; Holt et al. 2008b; Melcon et al. 2012b; Richardson et al. 1995b)(Kerosky et al. 2013). 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 2000c; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009a; Stensland and Berggren 2007a; Stockin et al. 2008b; Williams and Noren 2009). Noren et al. (2009a) 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. 2009a; Richardson and Wursig 1995; Watkins 1981e).

Long-term and cumulative implications of vessel sound on marine mammals remains largely unknown (NMFS 2012a, b). 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. 2013).

Bassett et al. (2012) 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 (Southall et al. 2005; Mintz and Filadelfo 2011).

6.1.10.3.1 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 1981). 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 (Castellote et al. 2012b) (Au and Green 2000; Richter et al. 2003; Williams et al. 2002).

Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012b) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012) 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 (1981) 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 2000c).

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. (2007a, b) 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 Action Area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004). 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 (Terhune and Verboom 1999, (Nowacek et al. 2004). 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) (NOAA 1997).

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 time (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); Melcón 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).

6.1.10.3.2 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. (2011) 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 2012). Research suggests that whale-watching distances may be insufficient to prevent behavioral disturbances due to vessel noise (Noren et al. 2009a). In 2012, there were 79 U.S. and Canadian commercial whale watch vessels in the Haro Strait region (Eisenhardt 2012). 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 2002). Killer whales foraged significantly less and traveled significantly more when boats were within 328 ft. (100 m) (Kruse 1991a; Trites and Bain 2000; Williams and Noren 2009; Williams et al. 2002c);Lusseau et al. 2009). These short-term feeding activity disruptions may have important long-term population-level effects (Lusseau et al. 2009; (Noren et al. 2009a). 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. 2008).

6.1.10.3.3 Pinnipeds

Little is known about pinniped reactions to underwater non-impulsive sounds (Southall et al. 2007a) including vessel noise. In a review of reports on reactions of pinnipeds to small craft and ships, Richardson et al. (1995) note that information on pinniped reactions is limited and most reports are based on anecdotal observations. Specific case reports in Richardson et al. (1995) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007a) pinniped responses to vessels are affected by the context of the situation and by the animal's experience. In summary, pinniped's reactions to vessels are variable and reports include a wide entire spectrum of possibilities from avoidance and alert to cases where animals in the water are attracted and cases on land where there is lack of significant reaction suggesting “habituation” or “tolerance” of vessels (Richardson et al. 1995).

A study of reactions of harbor seals hauled out on ice to cruise ship approaches in Disenchantment Bay, Alaska revealed that animals are more likely to flush and enter the water when cruise ships approach within 1,640 ft. (500 m) and four times more likely when the cruise ship approaches within 328 ft. (100 m) (Jansen et al. 2010). Navy vessels would generally not operate in vicinity of nearshore natural areas that are pinniped haul-out or rookery locations.

6.1.10.4 *Behavioral Reactions of Marine Mammals to Aircraft and Missile Overflight*

Thorough reviews on the behavioral reactions of marine mammals to aircraft and missile overflight are presented in Richardson et al. (1995b), Efroymson et al. (2000), Luksenburg and Parsons (2009), 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 (Manci et al. 1988; (Holst et al. 2011). Richardson et al. (1995) 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. 1995). 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.

6.1.10.4.1 Mysticetes

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Efroymson et al. 2000; Koski et al. 1998). Richardson et al. (1995) 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.1995). 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.

6.1.10.4.2 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. 1995).

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. 2008b; 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. 1995). 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. 2008b). 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).

6.1.10.4.3 Pinnipeds

Richardson et al. (1995) noted that data on pinniped reactions to aircraft overflight largely consisted of opportunistic and anecdotal observations. Richardson et al.'s (1995) summary of this variable data notes responsiveness generally was dependent on the altitude of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). Hauled out pinnipeds exposed to aircraft sight or sound often react by becoming alert and in many cases rushing into the water. Stampedes resulting in mortality to pups (by separation or crushing) have been noted in some cases although it is rare. Holst et al. (2011) provides an up-to-date review of this subject.

Helicopters are used in studies of several species of seals hauled out and is considered an effective means of observation (Bester et al. 2002; Bowen et al. 2006; Gjertz and Borset 1992), although they have been known to elicit behavioral reactions such as fleeing (Hoover 1988). In other studies, harbor seals showed no reaction to helicopter overflights (Gjertz and Borset 1992).

Ringed seals near an oil production island in Alaska reacted to approaching Bell 212 helicopters generally by increasing vigilance, although one seal left its basking site for the water after a helicopter approached within approximately 328 ft. (100 m) (Blackwell et al. 2004). Seals in the study near an oil production platform were thought to be habituated and showed no reactions to industrial noise in water or in air, including impact pile-driving, during the rest of the observations.

Pinniped reactions to rocket launches and overflight at San Nicolas Island, California were studied for the time period of August 2001 through October 2008 (Holst et al. 2011). Consistent with other reports, behavioral reactions were found to differ between species. California sea lions startled and increased vigilance for up to 2 minutes after a rocket overflight, with some individuals moving down the beach or returning to the water. Northern elephant seals showed little reaction to any overflight. Harbor seals had the most pronounced reactions of the three species observed with most animals within approximately 2.5 miles (4 km) of the rocket trajectory leaving their haul-out sites for the water and not returning for several hours. The authors concluded that the effects of the rocket launches were minor to individuals and that there are no effects on local populations as evidenced by the increasing populations of pinnipeds on San Nicolas Island (Holst et al. 2011).

6.1.11 **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 anit-submarine warefare is lower than used in the controlled exposure experiments described above, transmitting about once per minute (Navy 2013c). 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.

(2006b) 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 HSTT 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. 2006c; 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. 2011b). 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 study area used by Moore and Barlow 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) (Reed 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 incuding 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) (Kight and Swaddle 2011) (Goldbogen et al. 2013) (McGregor et al. 2013) (Reed et al. 2014) (Williams et al. 2014). McGregor et al. (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. 2009a) (Blickley et al. 2012, Reed et al 2014).

Some, including Goldbogen et al. (2013) and (Stockin et al. 2008a) 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 during major training exercises), 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 temporaily 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.

6.1.12 **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; Perrin and Geraci 2002; Walker et al. 2005)(NRC 2003) (Hoelzel 2003). 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 2003a; Laist et al. 2001), fisheries interactions (Read et al. 2006)(Look 2011), entanglement (Baird and Gorgone 2005)(Johnson et al. 2005; Saez et al. 2013), and noise (Richardson et al. 1995a)(NRC 2003)(Cox et al. 2006).

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 (National Marine Fisheries Service 2011a, b, c, d). 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 U.S. Department of the Navy (2013).

Sonar use during exercises involving U.S. 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 (Marine Mammal Commission 2006). 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 (International Council for the Exploration of the Sea 2005a, b, c). The U.S.-Navy-funded research involving Behavioral Response Studies in SOCALand 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” (Jepson et al. 2003; Fernandez et al. 2005; 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 et al. 2001, 2010a; 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 (2004).

Additionally NMFS undertook a series of necropsy analyses on the stranded animals to determine the cause of the strandings (NMFS 2005b, Norman et al. 2004). Necropsies were performed on 10 of the porpoises and two heads were selected for computed tomographic imaging (Norman et al. 2004).

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. 2004). 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. 2004). 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. 2004). 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 U.S. Department of the Navy [2013], 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 International Council for the Exploration of the Sea (2005b) 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 (McDonald et al. 2006; (Bassett et al. 2010); Baumann-Pickering et al. 2010; Hildebrand et al. 2011; Tyack et al. 2011). 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 U.S. 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 (O'Shea & Brownell 1994; Hooker et al. 2007), 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, International Council for the Exploration of the Sea 2005b, 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.

6.1.13 **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. See Section 3.3.4 for a more thorough discussion on the potential applicability of Population Viability Models (PVAs) to this Opinion.

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) and described in more detail in (New et al. 2014). Its 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 6.1.11, 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 relevent empirical data is collected to support such a model for the species considered in this Opinion, the best assessment of long-term consequences from training activities will be to monitor the populations over time within a given Navy range complex. A U.S. 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 HSTT Action Area. All monitoring reported are available to the public and posted at www.navymarinespeciesmonitoring.us/.

6.1.14 **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 Navy's criteria, thresholds, and methodology for quantifying impacts from acoustic and explosive sources, which were jointly developed with NMFS. For this information in its entirety, please see Navy (2013).

6.1.14.1 *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. (2007) 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. (2007) 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 11). 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).

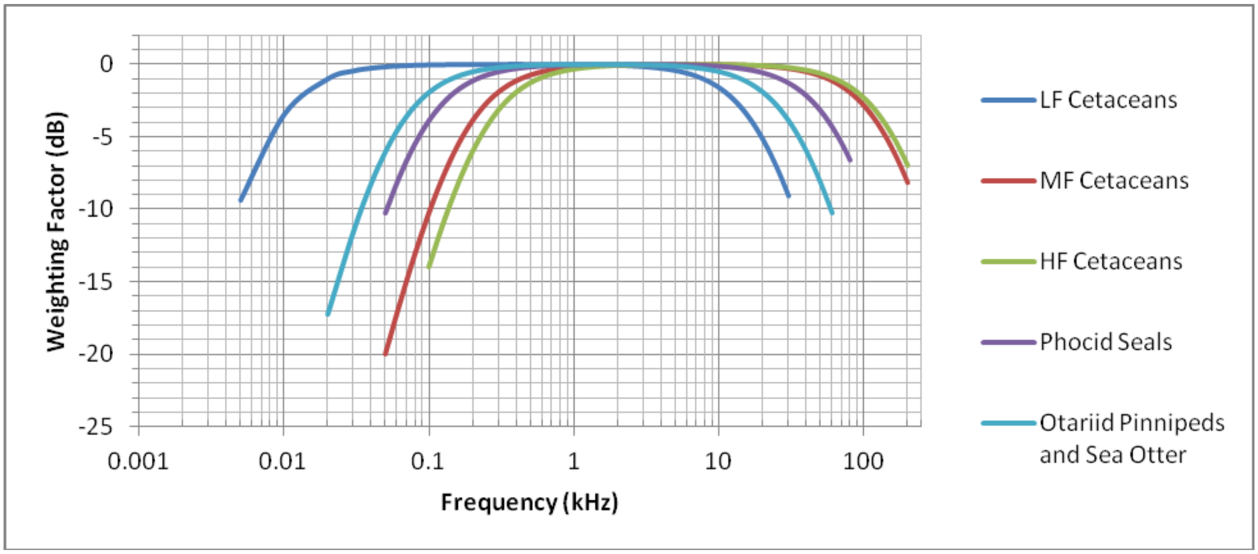

Figure 11. Type I Auditory Weighting Functions Modified from the Southall et al. (2007) M-Weighting Functions

(Finneran and Jenkins 2012) considered data since Southhall et al. (2007) 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 2011). 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 (National Institute for Occupational Safety and Health 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 12. To generate the new Type II weighting functions, Finneran and Schlundt (2011) substituted lower and upper frequency values which differ from the values used by Southall et al. (2007).

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. (2007) 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 12) 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. (2007) weighting functions (Figure 12) 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.

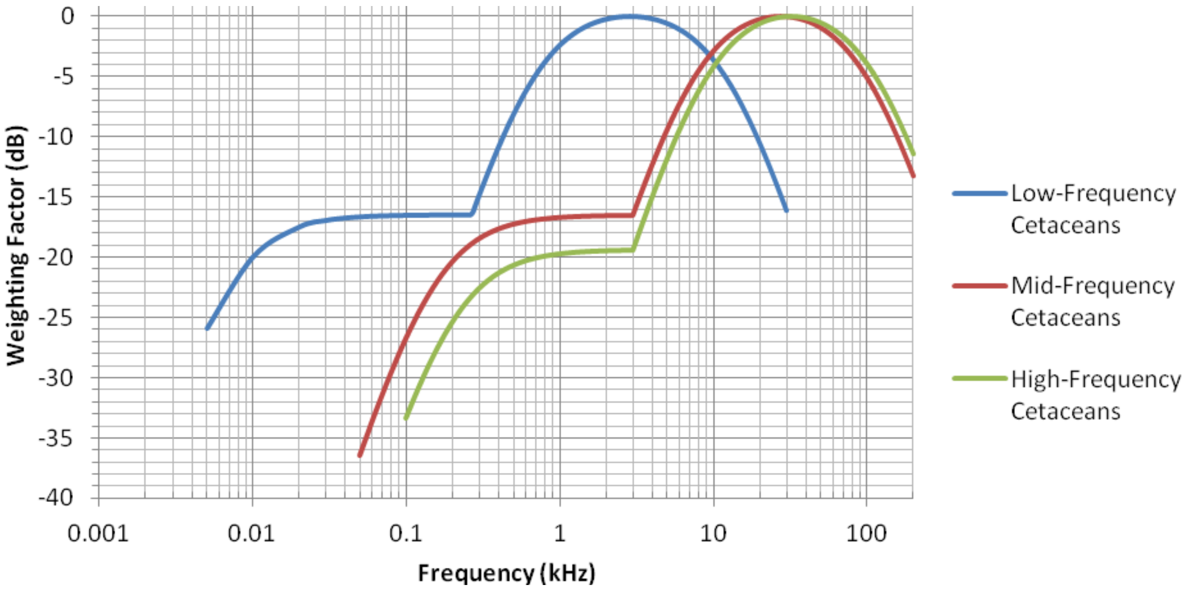

Figure 12. Type II Weighting Functions for Low-, Mid-, and High-Frequency Cetaceans

6.1.14.2 *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.

6.1.14.3 *Hearing Loss – Temporary and Permanent Threshold Shift*

Criteria for physiological effects (Table 31) 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 U.S. 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 31. 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)

Hearing Group	Species	Onset temporary threshold shift	Onset permanent threshold shift
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

6.1.14.3.1 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 (Schlundt et al. 2000); Finneran et al. 2005; Mooney 2009a; Finneran et al. 2010b; Finneran and Schlundt 2010) 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 and Schlundt 2010; Finneran 2011). 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 (Schlundt et al. 2000); Finneran et al. 2002. Therefore, 6 dB was added to the harbor porpoise's impulsive TTS threshold demonstrated by Lucke et al. (2009) to derive the non-impulsive TTS threshold used in the current Navy modeling for high frequency cetaceans. The first direct measurements of TTS from non-impulsive sound was presented by Kastelein et al. (2012b) for harbor porpoise. These data are fully consistent with the current harbor porpoise thresholds used in the modeling of effects from non-impulsive sources.

There are no direct measurements of TTS or hearing abilities for low-frequency cetaceans. The Navy uses mid-frequency cetacean thresholds to assess PTS and TTS for low-frequency cetaceans, since mid-frequency cetaceans are the most similar to the low frequency group (see Finneran and Jenkins (2012) on the development of the thresholds and criteria).

Pinniped TTS criteria are based on data provided by Kastak et al. (2005) for representative species of both of the pinniped hearing groups: harbor seals (Phocidae) and California sea lions (Otariidae). Kastak et al. (2005) used octave band noise centered at 2.5 kHz to extrapolate an onset TTS threshold. More recently Kastelein et al. (2012c) 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 32 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 32. 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

6.1.14.3.2 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. 2000). 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 (National Marine Fisheries Service 2008a, 2008b). 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. (2007). 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).

6.1.14.3.3 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 (Ward et al. 1958, 1959a, b; Miller et al. 1963). 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, 1959a, b) 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. 2007).

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. (2007). The method overestimates or predicts greater effects than have actually been observed in tests on a bottlenose dolphin (Schlundt et al. 2006; (Finneran and Schlundt 2010) 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 33 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 33. 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

6.1.14.3.4 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. (2007) 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 12, to predict PTS.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. 2001).

Criteria and thresholds for predicting injury and mortality to marine mammals from explosive sources are listed in Table 34. 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 34. Criteria and Thresholds for Predicting Physiological Effects to Marine Mammals Underwater for Explosives

Group	Species	Onset TTS	Onset PTS	Onset Slight GI Tract Injury	Onset Slight Lung Injury	Onset Mortality
Low-Frequency Cetaceans	Mysticetes	172 dB re 1 μPa^2-s (low-freq weighting) or 224 dB Peak SPL	187 dB re 1 μPa^2-s (low-freq weighting) or 230 dB Peak SPL	237 dB re 1 μPa	Equation 1	Equation 2
Mid-Frequency Cetaceans	Odontocetes (Toothed Whales)	172 dB re 1 μPa^2-s (mid-freq weighting) or 224 dB Peak SPL	187 dB re 1 μPa^2-s (mid-freq weighting) or 230 dB Peak SPL			
High-Frequency Cetaceans	Porpoises and *Kogia* spp.	146 dB re 1 μPa^2-s (mid-freq	161 dB re 1 μPa^2-s (mid-freq			

Group	Species	Onset TTS	Onset PTS	Onset Slight GI Tract Injury	Onset Slight Lung Injury	Onset Mortality
		weighting) or 195 dB Peak SPL	weighting) or 201 dB Peak SPL			
Phocid Seals (In-Water)	Harbor, beared, 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:

$$(1)\quad \boxed{-39.1M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-sec}$$

$$(2)\quad \boxed{=91.4M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2} Pa-sec}$$

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

6.1.14.3.5 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 (U.S. Department of the Navy 2008a; 63 FR 230, 66 FR 87, 73 FR 143).

6.1.14.3.6 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 U.S. 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. 2001). 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.

6.1.14.4 *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.

6.1.14.4.1 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. 2007a) 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 13) differs from that used for odontocetes (Figure 14) 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.

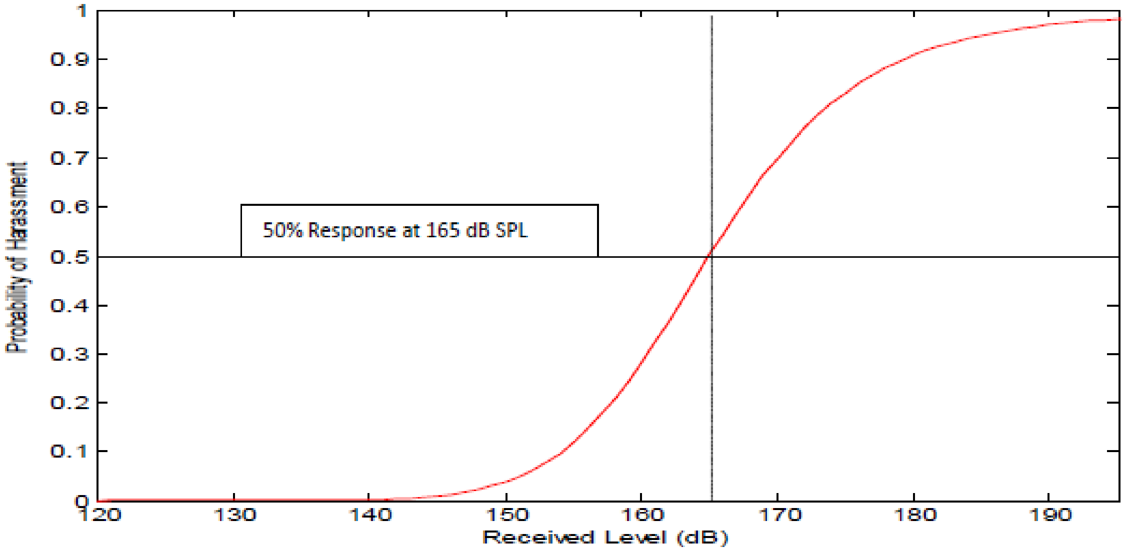

Figure 13. Behavioral response function applied to mysticetes.

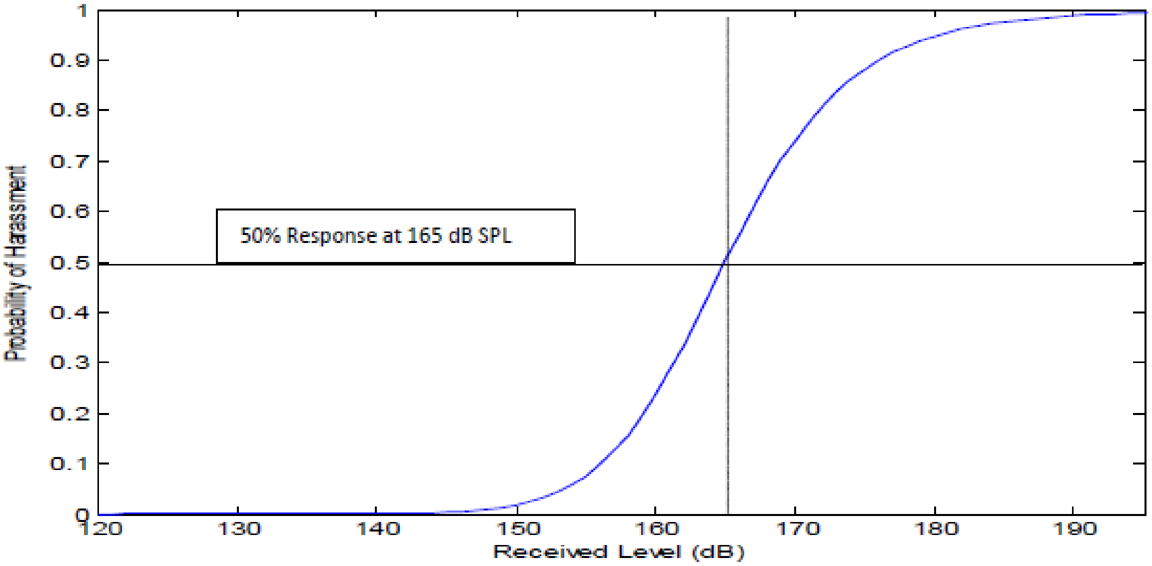

Figure 14. Behavioral response function applied to odontocetes and pinnipeds.

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 35 and Table 36 respectively.

Table 35. 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 36. 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%

6.1.14.4.2 Explosives

The thresholds for a behavioral response from explosives are listed in Table 37. 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 37) are applied to the received sound level before being compared to the threshold.

Table 37. 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 reactions. 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.

6.2 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.

6.2.1 **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) (Office of the Surgeon General 1991; Craig and Hearn 1998), 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.

6.2.2 **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 1995, 1993). 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 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.

6.2.3 **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 (Moein Bartol and Ketten 2006; 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, b) 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.

6.2.4 **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 (National Research Council 2003), 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 (Moein Bartol and Ketten 2006; 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, b) 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.

6.2.5 **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 HSTT 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 (Rees et al. 2008, Zenteno 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).

6.2.6 **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 1967a; 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.

6.2.6.1 *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 Doukara (2012) noted diving behavior following airgun shots in loggerhead turtles, and noted a decreased dive probability with increasing distance from the airgun array.

6.2.6.2 *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 (e.g., see Section 4.3.10.7).

6.2.6.3 *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).

6.2.6.4 *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.

6.2.7 **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 warefare is about once per minute (Navy 2013c). 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) (Reed 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 incuding 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) (Kight and Swaddle 2011) (Goldbogen et al. 2013) (McGregor et al. 2013) (Reed et al. 2014) (Williams et al. 2014). 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. 2009a) (Blickley et al. 2012, Reed et al 2014).

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Ketten 2006; 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, b) 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 (except the resident green sea turtles in San Diego Bay), we do not believe this will result from Navy training and testing activities in the HSTT 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 during major training exercises), 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, with the exception of resident green sea turtles in San Diego Bay, 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 temporaily 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.

The resident green sea turtle population in San Diego Bay spends a considerable amount of time in a small area, where repeated exposures may occur. However, as discussed above, sea turtles may habituate to acoustic stressors and are do not rely on auditory cues for foraging, breeding, or sheltering. Though auditory cues may be used to find nesting beaches, nesting of green sea turtles does not occur in southern California. 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.

6.2.8 Criteria for Predicting Acoustic and Explosive Impacts – Sea turtles

In this Opinion, we consider two primary categories of sound sources that the U.S. 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.

6.2.8.1 *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 class (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 and Ketten 2006, Bartol et al. 1999, Lenhardt 1994, Ridgway 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.

6.2.8.2 *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 (see Section 6.2.8.2). 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 (Bartol and Musick 2003; Lenhardt et al. 1985; 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 6.2.8.2.2, 6.2.8.2.1, 6.2.8.2.4, and 6.2.8.2.3, 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 38. 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 39. 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 40 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 40. 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.**

6.2.8.2.1 Temporary Threshold Shift – Non-impulsive sources

Based on best available science regarding TTS in marine vertebrates (Finneran et al. 2002; Southall et al. 2007) 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 (µPa2 -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 µPa2 -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 µPa2 -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).

6.2.8.2.2 Temporary Threshold Shift – Impulsive sources

Based on best available science regarding TTS in marine vertebrates (Finneran et al. 2005; Finneran et al. 2000; Finneran et al. 2002; Nachtigall et al. 2003; Nachtigall et al. 2004; Schlundt et al. 2000) 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).

6.2.8.2.3 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. 2007; Ward et al. 1958; Ward et al. 1959). 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, 1959) 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).

6.2.8.2.4 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. (2007) 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).

6.2.8.3 *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.

6.2.8.3.1 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[9]. 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 41 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 41. 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)
Kemp’s ridley sea turtle	6.3	McVey and Wibbels (1984) and Caillouet (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

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[9] Previous versions of this Opinion had assumed sea turtles experiencing even slight lung injury (due to acoustic stressors, as modeled by NAEMO) would be expected to die as a result of that injury. However, after giving further consideration to the Criteria for Mortality and Slight Lung Injury used in the NAEMO model, we do not believe this assumption to be correct. As described in this section, onset mortality and onset slight lung injury are defined as the impulse levels that would result in one percent mortality (most survivors have moderate blast injuries and should survive) and zero percent mortality (recoverable, slight blast injuries), respectively. Though very little information exists about the impacts of underwater detonations on sea turtles, the criteria used to determine the onset of mortality and slight lung injury incorporates the best available information regarding the outcomes (e.g., will the animal recover or not) of varying levels of exposure and resulting injury. Therefore, it is not reasonable to assume turtles which are modeled to be exposed to acoustic stressors at levels resulting in recoverable, slight blast injuries would die.

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 surface (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 38.

6.2.8.3.2 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.

6.2.8.4 *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 6.2.6, 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 μPa2 -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 HSTT 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.

6.3 Quantitative Analysis

The Navy performed a quantitative analysis (NAEMO) to estimate the number of marine mammals and sea turtles that could be affected by acoustic sources or explosives used during Navy training activities. Inputs to the quantitative analysis include marine mammal density estimates; marine mammal depth occurrence distributions; oceanographic and environmental data; marine mammal hearing data; and criteria and thresholds for levels of potential effects. The quantitative analysis consists of computer modeled estimates and a post-model analysis to determine the number of potential mortalities, injuries, and harassments. The model calculates sound energy propagation from sonar, other active acoustic sources, and explosives during naval activities; the sound or impulse received by animat dosimeters representing marine mammals distributed in the area around the modeled activity; and whether the sound or impulse received by a marine mammal exceeds the thresholds for effects. The model estimates are then further analyzed to consider animal avoidance and implementation of mitigation measures, resulting in final estimates of potential effects due to Navy testing and training.

Various computer models and mathematical equations can be used to predict how energy spreads from a sound source (e.g., sonar or underwater detonation) to a receiver (e.g., dolphin or sea turtle). Basic underwater sound models calculate the overlap of energy and marine life using assumptions that account for the many, variable, and often unknown factors that can influence the result. Assumptions in previous and current Navy models have intentionally erred on the side of overestimation when there are unknowns or when the addition of other variables was not likely to substantively change the final analysis. For example, because the ocean environment is extremely dynamic and information is often limited to a synthesis of data gathered over wide areas and requiring many years of research, known information tends to be an average of a seasonal or annual variation. El Niño Southern Oscillation events of the ocean-atmosphere system are an example of dynamic change where unusually warm or cold ocean temperatures are likely to redistribute marine life and alter the propagation of underwater sound energy. Previous Navy modeling (U.S. Department of the Navy 2009) makes assumptions indicative of a maximum theoretical propagation for sound energy (such as a perfectly reflective ocean surface and a flat seafloor).

More complex computer models build upon basic modeling by factoring in additional variables in an effort to be more accurate by accounting for such things as variable bathymetry and an animal’s likely presence at various depths. NAEMO accounts for the variability of the sound propagation data in both distance and depth when computing the received sound level on the animals. Previous models captured the variability in sound propagation over range and used a conservative approach to account for only the maximum received sound level within the water column.

6.4 Stressors Associated with the Action

As part of the consultation request, the U.S. Navy provided the list of stressors they evaluated in their FEIS/OEIS (Navy 2013d) (Table 42). During consultation, we made the final ESA determinations of effect for each of the stressor categories and focused our analysis on the specific activities producing those stressors that are likely to interact with ESA-listed species and/or designated critical habitat. The activities are highlighted in Table 42. List of stressors the U.S. Navy analyzed for impacts to biological resources in the HSTT Action Area.Table 42 along with the final determination. Stressors that are likely to adversely affect species and/or critical habitat are carried forward in the full effects analysis. We also summarize our conclusions for stressors that are not likely to adversely affect species and/or critical habitat.

Table 42. List of stressors the U.S. Navy analyzed for impacts to biological resources in the HSTT Action Area.

Components and Stressors for Biological Resources	Final ESA Determinations
Acoustic Stressors **Sonar and other active acoustic sources** **Explosives** **Swimmer defense airguns**	**Likely to Adversely Affect**
Weapons firing, launch, and impact noise Vessel noise Pile driving Aircraft noise	Not Likely to Adversely Affect
Energy Stressors Electromagnetic devices Lasers	Not Likely to Adversely Affect
Physical Disturbance and Strike Stressors **Vessels**	**Likely to Adversely Affect**
In-water devices Military expended materials Seafloor devices	Not Likely to Adversely Affect
Entanglement Stressors Fiber optic cables and guidance wires Parachutes	Not Likely to Adversely Affect
Ingestion Stressors Munitions Military expended materials other than munitions	Not Likely to Adversely Affect
Secondary Stressors	Not Likely to Adversely Affect

Bolded text indicates stressors that are likely to adversely affect ESA-listed species.

6.5 Stressors That Are Not Likely to Adversely Affect Species and Critical Habitat

It has been determined the following stressors are not likely to adversely affect species or critical habitat.

6.5.1 Energy Stressors

Energy stressors fall into two categories, the electromagnetic fields generated by mine countermeasure devices and railguns, and lasers used for targeting, mine detection, and disabling small boat and airborne threas. Naval devices producing electromagnetic fields include towed or unmanned mine countermeasure systems. The electromagnetic field is produced to simulate a vessel's magnetic field. In an actual mine clearing operation, the intent is that the electromagnetic field would trigger an enemy mine designed to sense a vessel's magnetic field. The majority of devices involved in the proposed activities would be towed or unmanned mine warfare systems that mimic the electromagnetic signature of a vessel passing through the water. None of the devices include any type of electromagnetic "pulse". An example of a representative device is the Organic Airborne and Surface Influence Sweep that would be used by an unmanned surface craft at sea. The Organic Airborne and Surface Influence Sweep works 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. Mine neutralization sites are small compared to the overall HSTT Action Area and are generally located in shallow, nearshore areas. There are no threshold criteria to determine the significance of electromagnetic energy effects on ESA-listed marine species.

The kinetic energy weapon (commonly referred to as the rail gun) is under development and will likely be tested and eventually used in training events aboard surface vessels, firing non-explosive projectiles at sea-based targets. The system uses stored electrical energy to accelerate the projectiles, which are fired at supersonic speeds over great distances. The system charges for two minutes, and fires in less than a second; therefore, any electromagnetic energy released would be done so over a very short period. Also, the system would likely be shielded so as not to affect shipboard controls and systems. The amount of electromagnetic energy released from this system would be low and contained on the surface vessel.

The Navy proposes to test the kinetic energy weapon on vessels off the Pacific Missile Range Facility in the Hawaii Range Complex. This kinetic energy weapon would generate an electromagnetic field (within the railgun barrel) to launch a projectile. Because the electromagnetic field is produced within the kinetic energy weapon barrel, ESA-listed species would not be exposed to the electromagnetic field. The exposure of ESA-listed species to the electromagnetic energy produced by railguns is highly unlikely and therefore discountable. Therefore, we conclude the electromagnetic energy produced by railguns is not likely to adversely affect ESA-listed species and the risk of kinetic energy weapon use is not discussed further in this Opinion.

Laser devices can be organized into two categories: (1) low energy lasers and (2) high energy lasers. Low energy lasers are used to illuminate or designate targets, to guide weapons, and to detect or classify mines. High energy lasers are used as weapons to disable surface targets. No high energy lasers would be used in the Action Area as part of the Action, and are not discussed further.

Within the category of low energy lasers, the highest potential level of exposure would be from an airborne laser beam directed at the ocean’s surface. An assessment on the use of low energy lasers by the Navy determined that low energy lasers, have an extremely low potential to impact marine biological resources (Swope 2010). The assessment determined that the maximum potential for laser exposure is at the ocean’s surface, where laser intensity is greatest (Swope 2010). As the laser penetrates the water, 96 percent of a laser beam is absorbed, scattered, or otherwise lost (Ulrich 2004). Based on the parameters of the low energy lasers and the behavior and life history of major biological groups, it was determined the greatest potential for impact would be to the eyes of marine mammals or sea turtles.

6.5.1.1 ***Cetaceans***

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 (Hui 1984; 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.

Fin, humpbacks, and sperm whales have shown positive correlations with geomagnetic field differences. Although none of the studies have determined the mechanism for magneto-sensitivity, the suggestion from these studies is that whales can sense the Earth's magnetic field and may use it to migrate long distances. Cetaceans appear to use the Earth's magnetic field for migration in two ways: as a map by moving parallel to the contours of the local field topography, and as a timer based on the regular fluctuations in the field allowing animals to monitor their progress on this map (Klinowska 1990). Cetaceans do not appear to use the Earth's magnetic field for directional information (i.e. they do not use magnetic fields as an internal compass) (Klinowska 1990).

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.

To experience potentially biologically relevant exposure to low-intensity lasers, an animal's eye would have to be exposed to a direct laser beam for at least 10 seconds or longer to sustain damage. However, with pulse durations less than 10 seconds, combined with the laser platform movement and animal motion, exposures of more than 10 seconds would not be possible. Furthermore, 96 percent of a laser beam projected into the ocean is absorbed, scattered, or otherwise lost (Guenther et al. 1996). Swope (2010) assessed the potential for damage based on species specific eye/vision parameters and the anticipated output from low energy lasers and determined that no animals were predicted to incur damage. Zorn conducted an analysis (2000) of the sensitivity ratio was calculated for each species using the ratio of the irradiance at the retina of the marine mammal to the irradiance at the retina of humans. The sensitivity ratio was used to suggest exposure limits for the various species. They concluded that because the human eye is more sensitive than either the cetacean or pinniped eye, that laser energies that are eye-safe for humans will also be safe for marine mammals, and higher laser irradiances may be permissible if illumination of humans is avoided (Zorn et al. 2000). Due the relatively small footprint of mine fields, the temporary and infrequent firing of lasers, the required duration of exposure to cause potential injury, and the dissipation of laser energy in water, the likelihood of an ESA-listed cetacean being exposed to a low-intensity laser is so low as to be discountable. Therefore, low energy lasers are not likely to adversely affect ESA-listed cetaceans and will not be considered analyzed further in this Opinion.

6.5.1.2 *Pinnipeds*

Normandeau et al. (2011) reviewed available information on electromagnetic and magnetic field sensitivity of marine organisms (including marine mammals) for impact assessment of offshore wind farms for the Department of Interior and concluded there is no evidence to suggest any magnetic sensitivity for sea lions, fur seals, or sea otters (Normandeau et al. 2011). Given the small area associated with mine fields, the infrequency and short duration of magnetic energy use, the low intensity of electromagnetic energy sources, the density of ESA-listed pinnipeds in these areas, and the apparent lack of magnetic sensitivity in these species, the likelihood of ESA-listed pinnipeds being exposed to electromagnetic energy at sufficient intensities to create a biologically relevant response is so low as to be discountable. Electromagnetic energy from mine neutralization equipment is not likely to adversely affect ESA-listed pinnipeds. Electromagnetic energy from mine neutralization equipment is not likely to adversely affect their designated critical habitat because it will not occur in these areas. Therefore, electromagnetic energy from mine neutralization equipment will not be considered further in this Opinion.

To experience potentially biologically relevant exposure to low-intensity lasers, an animal's eye would have to be exposed to a direct laser beam for at least 10 seconds or longer to sustain damage. Swope (2010) assessed the potential for damage based on species specific eye/vision parameters and the anticipated output from low energy lasers and determined that no animals were predicted to incur damage. Zorn conducted an analysis (Zorn et al. 2000) of the sensitivity ratio was calculated for each species using the ratio of the irradiance at the retina of the marine mammal to the irradiance at the retina of humans. The sensitivity ratio was used to suggest exposure limits for the various species. They concluded that because the human eye is more sensitive than either the cetacean or pinniped eye, that laser energies that are eye-safe for humans will also be safe for marine mammals, and higher laser irradiances may be permissible if illumination of humans is avoided (Zorn et al. 2000). Due to the relatively small footprint of mine fields, the temporary and infrequent firing of lasers, the required duration of exposure to cause potential injury, and the dissipation of laser energy in water, the likelihood of an ESA-listed pinniped being exposed to a low-intensity laser is so low as to be discountable. Low energy lasers are not likely to adversely affect ESA-listed pinnipeds. Low energy lasers are not likely to adversely affect their designated critical habitat because it will not occur in these areas. Therefore, low energy lasers will not be analyzed further in this Opinion.

6.5.1.3 ***Sea Turtles***

Sea turtles use geomagnetic fields to navigate at sea, and therefore changes in those fields could impact their movement patterns (Lohmann and Lohmann 1996; Lohmann et al. 1997). 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 1996; Lohmann et al. 1997). Experiments show that sea turtles can detect changes in magnetic fields, which may cause them to deviate from their original direction (Lohmann and Lohmann 1996; Lohmann et al. 1997). For example, Lohmann and Lohmann (1996) found that loggerhead hatchlings tested in a magnetic field of 52,000 nanoteslas (nT) swam eastward, and when the field was decreased to 43,000 nT, the hatchlings swam westward. 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. 1999; 2001).

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. Its effects on sea turtles will not be considered further in this Opinion.

Lasers used as part of proposed training exercises and testing activities would be low-energy lasers used for mine detection and targeting. While all points on a sea turtle's body would have roughly the same probability of laser exposure, only eye exposure is of concern for low-energy lasers. Any heat that the laser generates would rapidly dissipate due to the large heat capacity of water and the large volume of water in which the laser is used. There is no suspected effect due to heat from the laser beam. Eye damage to sea turtles is unlikely because eye damage depends on wavelength with exposures of greater than 10 seconds. As discussed, with pulse durations less than 10 seconds, combined with the laser platform movement and animal motion, exposures of more than 10 seconds would not be possible. Furthermore, 96 percent of a laser beam projected into the ocean is absorbed, scattered, or otherwise lost (Guenther et al. 1996). The use of low-energy lasers is so unlikely to result in significant biological effects to sea turtles as to be discountable. Therefore, energy from low-intensity lasers is not likely to adversely affect ESA-listed sea turtles and the effects of this stressor on sea turtles will not be considered further in this Opinion.

6.5.2 Expended Materials (includes entanglement, strike, and ingestion stressors)

Expended materials from U.S. Navy training and testing may pose a risk of entanglement or ingestion to marine mammals and sea turtles in the HSTT 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. The Fisheries Interactions and Marine Debris sections of the *Environmental Baseline* in this Opinion discuss in detail the threat of fishing gear and other anthropogenic debris to marine mammals and sea turtles in the Action Area.

Expended materials from Navy training and testing may include the following: fiber optic cables, guidance wires, parachutes, explosive and non-explosive munitions (e.g., bombs, small arms ammunition, medium caliber cannons, and missiles), marine markers, flares, chaff, 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 struck by or 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 relatively low densities of threatened or endangered marine mammals and sea turtles in the HSTT 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 and sea turtles. Bottom feeding animals (e.g., sea turtles) 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. The risks of entanglement in or ingestion of expended materials on the seafloor are further discussed below.

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, Hawaiian monk seal, 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.

Some training exercises and testing activities involve the use of small and medium-caliber projectiles that could be encountered by marine mammals or sea turtles. However, the potential for such an encounter is low based on the patchy distribution of both the projectiles and an animal's feeding habitat. Further, an animal would not likely ingest every projectile it encounters. Animals may attempt to ingest a projectile and then reject it, after realizing it is not a food item. Additionally, ingestion of items does not necessarily result in injury or mortality to the individual if the item does not become embedded in tissue (Wells et al. 2008). Therefore impacts of non-explosive practice munitions ingestion would be limited to the unlikely event where a marine mammal or sea turtle might suffer a negative response from ingesting an item that becomes embedded in tissue or is too large to be passed through the digestive system.

As mentioned previously, it is possible that some military expended items (e.g., Styrofoam, target fragments) may remain at the surface for an extended period of time. However, given the large magnitiude of marine debris already present in the North Pacific Ocean from a multitude of sources (Schuyler et al. 2013) (e.g., fishing, plastics, other anthropogenic trash) and the relatively small amount of floating expendible materials expected to result from U.S. Navy activities, it is unlikely that U.S. Navy expendable materials will contribute to the injury or mortality of any ESA-listed animals through ingestion of, or entanglement in, floating debris. It should also be noted that some expended materials (e.g., sonobuoy tubes, 55-gallon drums) are large enough or configured in such a way that they do not pose an ingestion or entanglement threat to the ESA-listed species considered in this Opinion.

Given the information provided above, the likelihood of ESA-listed animals encountering, interacting with (ingesting or becoming entangled), and then being negatively impacted by expended materials, is so low as to be discountable. Therefore, expended materials are not likely to adversely affect ESA-listed animals and the effects of this stressor will not be considered further in this Opinion.

6.5.3 **Secondary Stressors**

This section analyzes potential impacts to marine mammals 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. In contrast, the killing of prey species could encourage ESA-listed species to congregate near the blast site prior to additional explosions. 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 2013c). 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 componenet 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 incontact 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, and sea turtle 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 HSTT activities are so unlikely as to be considered discountable. Therefore, secondary stressors from HSTT activities are not likely to adversely affect ESA-listed species.

6.5.4 Noise and Visual Effects from Navy Aircraft

Studies have shown that aircraft presence and operation can result in changes in behavior of cetaceans (Arcangeli and Crosti 2009; Holt et al. 2009; Luksenburg and Parsons 2009; Noren et al. 2009b; Patenaude et al. 2002; Richter et al. 2006; Richter et al. 2003a; Smultea et al. 2008a). Many of the activities the U.S. Navy conducts in the HSTT 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.

As discussed below, the potential for significant effects rising to the level of take from aircraft noise is discountable; therefore, 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.

6.5.4.1 *Cetaceans*

There are few studies of the responses of marine animals to air traffic and the few that are available have produced mixed results. Some investigators report responses while others report no responses. Richardson et al. (1995) reported that there is no evidence that single or occasional aircraft flying above large whales and pinnipeds in-water cause long-term displacement of these mammals. Several authors have reported that sperm whales did not react to fixed-wing aircraft or helicopters in some circumstances (Au and Perryman 1982, Clarke 1956, Gambell 1968, Green et al. 1992) and reacted in others (Clarke 1956, Fritts et al. 1983, Mullin et al. 1991, Patenaude et al. 2006, Richter et al. 2003, 2006, Smultea et al. 2008, Würsig et al. 1998). Richardson et al. (1985) 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 Kaua‘i 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 (2009) 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 (2009).

6.5.4.2 *Pinnipeds*

Richardson et al. (1995) noted that data on pinniped reactions to aircraft overflight largely consisted of opportunistic and anecdotal observations. Richardson et al.’s (1995) summary of this variable data note that responsiveness generally was dependent on the altitude of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). Hauled out pinnipeds exposed to aircraft sight or sound often react by becoming alert and in many cases rushing into the water. Stampedes resulting in mortality to pups (by separation or crushing) have been noted in some cases although it is rare.

6.5.4.3 *Sea Turtles*

Based on sea turtle sensory biology (Bartol et al. 1999b; Ketten and Bartol 2005; Ketten and Bartol 2006b; 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.

6.5.4.4 *Conclusion*

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 pinnipeds 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 disruption of breeding, feeding, or sheltering and would not rise to the level of take.

6.5.5 Exposure to Non-impulsive Sound from Vessels

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 (Richardson et al. 1995b)(Kipple and Gabriele 2007; McKenna et al. 2012). 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 (2012) in a study off Southern California documented different acoustic levels and spectral shapes observed from different modern ship-types.

6.5.5.1 *Exposure to Noise from Navy Vessels*

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 2005; Au and Perryman 1982; Au and Green 2000a; Bain et al. 2006; Bauer 1986; Bejder et al. 1999; Bejder and Lusseau. 2008; Bejder et al. 2009b; 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. 2003a; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002d; 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 (Au and Perryman 1982; Bain et al. 2006; Bauer and Herman 1986; Bejder et al. 1999; Bejder et al. 2006a; Bejder et al. 2006c; Bryant et al. 1984; Corkeron 1995; David 2002; Felix 2001; Goodwin and Cotton 2004; Hewitt 1985b; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2006; Richter et al. 2003c; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams and Ashe 2007; Williams et al. 2002d; Wursig et al. 1998) (Acevedo 1991b) (Acevedo 1991, Aguilar de Soto et al. 2006, Arcangeli and Crosti 2009, Au and Green 2000, Christiansen et al. 2010, Erbe 2002, Williams et al. 2009, Christiansen et al. 2010, Noren et al. 2009, Stensland and Berggren 2007, Stockin et al. 2008), 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 1991b; Lusseau 2003; Nowacek et al. 2001; Stensland and Berggren 2007b; 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 1982; David 2002; Hewitt 1985b; Kruse 1991b; 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.

6.5.5.2 ***Response to Noise from Navy Vessels***

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. 2002d). 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 1991b). 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 1991b). 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. (1985) 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 disturbance to tour vessel traffic may interrupt social interactions, and postulated that those 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.

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 (for example, Composite Training Unit Exercise, Joint Task Force Exercise/Sustainment Exercise, and Rim of the Pacific 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.

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).

6.5.5.2.1 Cetaceans

We considered the research and reports cited above and conclude that in general blue, fin, humpback, sei, and MHI insular false killer whales are likely to either not react or exhibit an avoidance behavior. Most of these avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives (or longer intervals between blows). Most of the changes in behavior would consist of a shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling). In some instances, the whales are either not likely to respond or are not likely to respond in ways that might be adverse to the whales (the responses might represent an approach or attentive movement, a small change in orientation in the waters, etc.).

Behavioral disruptions of whales result from the presence of vessels or submarines, those disruptions are expected to be temporary. Animals are expected to resume their migration, feeding, or other behaviors with minimal threat to their survival or reproduction. 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. 1995c). Some respond negatively by retreating or engaging in antagonistic responses while other animals ignore the stimulus altogether (Terhune and Verboom 1999; Watkins 1986).

In summary, ESA-listed cetaceans 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 vessel noise on ESA-listed cetaceans is insignificant and not likely to adversely affect them.

6.5.5.2.2 Pinnipeds

As with the cetacean species, we would expect Hawaiian monk seals and Guadalupe fur seals to engage in avoidance behavior when surface vessels move toward them. Seals would likely reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies. Most animals finding themselves in confined spaces, such as shallow bays, during vessel approaches tended to move towards more open, deeper waters. We assume that this movement would give them greater opportunities to avoid or evade vessels as conditions warranted.

ESA-listed pinnipeds are either not likely to respond to vessel noise or are not likely to measureably 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 vessel noise on ESA-listed pinnipeds is insignificant and not likely to adversely affect them.

6.5.5.2.3 Sea Turtles

Based on the information available, endangered and threatened sea turtles may have a brief startle response, but are most likely to ignore Navy vessels entirely and continue behaving as if the vessels and any risks associated with those vessels did not exist (Hazel 2007). Sea turtles 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 vessel noise on ESA-listed sea turtles is insignificant and not likely to adversely affect them.

6.5.6 **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 propogate 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 from navy modeling efforts. Marine mammals and sea turtles within the audible range of munition firing, launching and impact may exhibit a behavioral startle response but are expected to quickly return to normal behavior. Munitions firing, launch, and impacts are sporadic events of short duration reducing the likelihood of subjecting individual marine mammals and sea turtles to prolonged or repeated exposures. Due to the short duration and sporadic nature of munition firing, the low likelihood of an ESA-listed animal being within close enough proximity to detect sound from firing above water, and the high likelihood an ESA-listed animal that does detect noise from munition firing would react very briefly, we do not expect an increase in the likelihood of injury from significant disruption of breeding, feeding, or sheltering is expected for ESA-listed marine mammals or sea turtles. Therefore, the effects of weapon firing, launching, and impact noise on ESA-listed marine mammals and sea turtles would not rise to the level of take and any behavioral impacts would be insignificant.

6.5.7 Effects of In-water Devices

Despite thousands of Navy exercises in which torpedoes and in-water devices (unmanned surface vehicles, unmanned undersea vehicles, towed devices) have been used there have been no recorded instances of marine species strikes. Therefore, the likelihood of striking a marine mammal or sea turtle is so unlikely as to be discountable. Any ESA-listed marine mammal or sea turtle 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.

6.5.8 Seafloor Devices

Seafloor devices include moored mine shapes, anchors, bottom placed instruments, and robotic vehicles referred to as “crawlers.” Seafloor devices are either stationary or move very slowly along the bottom and do not pose a threat to highly mobile organisms. Objects falling through the water column will slow in velocity as they sink toward the bottom and would be avoided by ESA-listed species. The only seafloor device used during training and testing activities that has the potential to strike an ESA-listed species at or near the surface is an aircraft deployed mine shape, which is used during aerial mine laying activities. These devices are identical to non-explosive practice bombs, therefore the analysis of the potential impacts from those devices are covered in the military expended material strike section. We consider the likelihood of mobile ESA-listed species being struck by a slow moving seafloor device is so low as to be discountable. Sea turtles encountering seafloor devices are likely to behaviorally avoid them. Given the slow movement of seafloor devices, the effort expended by sea turtles to avoid them will be minimal, temporary, and will not have fitness consequences. Therefore, behavioral avoidance of mobile seafloor devices by sea turtles will be insignificant and not rise to the level of take.

6.6 Exposure and Response to Stressors that Are Likely to Adversely Affect Species – Training and Testing Activities

Our exposure analyses are designed to determine whether ESA-listed resources are likely to co-occur with the direct and indirect effects of actions and the nature of that co-occurrence. In this step of our analyses, we try to identify the number of the individuals that are likely to be exposed to one or more of the stressors produced by, or associated with, an Action and the populations or subpopulations those individuals represent.

The response analyses are designed to identify how endangered or threatened species (or designated critical habitat, when it is applicable) are likely to respond given their exposure to one or more of the stressors produced by an action. These analyses consider and weigh all of the evidence available, including the best scientific and commercial data available, to identify the probable responses of endangered and threatened species upon being exposed to stressors associated with actions. The results of these analyses are summarized by stressor in the following sections.

6.6.1 Acoustic Stressors

As we discussed in the *Evidence Available for the Consultation*, the Navy's Acoustic Effects Model (NAEMO) was used to estimate exposure to impulsive and non-impulsive acoustic stressors excluding broadband noise generated from vessels. Vessel noise is disussed separately in section 6.5.5.2. An animal is considered "exposed" to a sound if the animal's received sound level is greater than the low-amplitude acoustic threshold below which meaningful biological responses are not anticipated (120 dB) within a similar frequency band.

To conduct the exposure analysis, acoustic sources were divided into categories (bins) based on sound characteristics. Impulsive bins are based on the net explosive weight of munitions and explosive devices or the source level for air and water guns, and pile driving. Their were no modeled exposures from pile driving that would rise to the level of take for any species considered in this Opinion.

Non-impulsive acoustic sources are grouped into bins based on the frequency, source level, and when warranted, the application in which the source would be used. The following factors further describe the considerations associated with the development of non-impulsive source bins:

Frequency of the non-impulsive source:

- Low-frequency sources operate below 1 kHz
- Mid-frequency sources operate at and above 1 kHz, up to and including 10 kHz
- High-frequency sources operate above 10 kHz, up to and including 100 kHz

- Very high-frequency sources operate above 100 kHz but below 200 kHz

Source level of the non-impulsive source:

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

Application in which the source would be used:

- How a sensor is employed supports how the sensor's acoustic emissions are analyzed
- Factors considered include pulse length (time source is on); beam pattern (whether sound is emitted as a narrow, focused beam or in all directions); and duty cycle (how often or how many times a transmission occurs in a given time period during an event).

The Navy provided NMFS with output values as "unprocessed" or "raw" exposure estimates. That is the estimated number of animats exposed to acoustic sources at received levels above 120 dB.

For this consultation, we considered exposure estimates from the NAEMO at several output points. First, 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 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, (2) some animals may respond negatively to an exposure without that response 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), and (3) Navy minimization and mitigation measures may reduce the number or magnitude of unprocessed exposures.

A second set of exposure estimates of listed species were generated and "processed" using dose-response curves and criteria for temporary and permanent threshold shifts developed by the Navy and NMFS Permits Division (Navy 2013b). Neither sets of exposure estimates, the unprocessed or processed, consider standard mitigation actions that the NMFS Permits Division requires under the MMPA rule to avoid marine mammals or that the Navy implements for marine mammals.The estimates also do not consider any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft.

Lastly, the U.S. Navy applied a third step of incorporated species specific avoidance and mitigation to derive the Navy's final MMPA take request. To account for mitigation measures, the acoustic analysis assumes a portion of the model-predicted mortalities or injuries would not occur if an animal is observed by lookouts or other mitigation measures (Navy 2013g).

The Navy used mitigation effectiveness and sightability to partially account for the implementation of mitigation. The mitigation effectiveness is considered over two regions of an event's mitigation zone: (1) the range to onset mortality and (2) the range to onset PTS. The model-estimated mortalities and injuries are reduced by the portion of animals that are likely to be seen [Mitigation Effectiveness x Sightability, g(0)]; these animals are instead assumed to be present within the range to injury and range to TTS, respectively. During an activity with a series of explosions (not concurrent multiple explosions), an animal is expected to exhibit an initial startle reaction to the first detonation followed by a behavioral response after multiple detonations. At close ranges and high sound levels approaching those that could cause PTS, avoidance of the area around the explosions is the assumed behavioral response for most cases.

The ranges to PTS for each functional hearing group for a range of explosive sizes (single detonation) are provided in Table 33. Animals not observed within the ranges to PTS at the time of the initial explosions are assumed to experience PTS; however, animals beyond the initial range to PTS are assumed to move away with each additional explosion. Additionally, odontocetes have been demonstrated to have directional hearing, with best hearing sensitivity facing a sound source. An odontocete avoiding a source would receive sounds along a less sensitive hearing axis, potentially reducing impacts. Because the NAEMO does not account for avoidance behavior, the model-estimated effects are based on unlikely behavior that animals would remain in the vicinity of potentially injurious sound source and stay oriented to face the source. Therefore, only the initial unmitigated exposures result in PTS are counted in the applied mitigation values, with the remaining model-estimated mitigated PTS exposures being downgraded to TTS due to avoidance. This is why the values below for behavior responses or TTS estimates may increase and estimates of PTS or other injuries are decreased after application of mitigiation measures to modeled estimates. The Navy's approach of considering mitigation and avoidance was developed in cooperation with NMFS, and we believe it represents an appropriate application of the best available information regarding the effectiveness of the Navy's mitigation strategies. For our analysis, we consider Navy mitigation measures to be generally effective at minimizing exposure to higher-level effects such as injury and PTS, but less effective at minimizing TTS and behavioral responses due to the greater range to effects from the source and the Navy's inability to detect species at greater ranges (i.e. distance from vessels).

The analysis presented in this Opinion considers all three exposure estimates on an annual basis. Specific numbers of takes presented in this Opinion may vary from the numbers of takes in the referenced documents, such as the FEIS and associated technical reports, due to rounding and the different analyses used in this Opinion.

6.6.1.1 ***Sources of Exposures of ESA-listed Species during Training***

The NAEMO model output (based on unprocessed estimates) indicates that four types of training exercises accounted for the majority of exposures to non-impulsive sound sources (Table 43) and four types of exercises comprise the majority of the exposures to impulsive acoustic sources (Table 44).

Table 43. Proportion of unprocessed exposure estimate to non-impulsive sound sources from specific training exercises in the HSTT Action Area.

Species	Training Activities			
	Composite Training Unit	Submarine Navigation	Tracking/Torpedo – Surface	Undersea Warfare
Blue Whale	40%		35%	
Fin Whale	33%		40%	
Humpback Whale		20%	38%	
Sei Whale	13%	11%	40%	
Sperm Whale	23%		38%	
MHI Insular False Killer Whale			54%	19%
HI Monk Seal		64%	14%	
Guadalupe Fur Seal	38%		37%	
Green Turtle				
Pacific Sea Turtles				78%

Table 44. Proportion of unprocessed exposure estimate to impulsive sound sources from specific training exercises in the HSTT Action Area.

Species	TrainingActivities			
	Bombing Air-to-Sea	Firing at Sea	Mine Neutralization – EOD	Tracking/Torpedo MPA Extended Echo Ranging Sonobuoy
Blue Whale	69%			
Fin Whale	65%		18%	
Humpback Whale	23%	26%		
Sei Whale	49%		23%	
Sperm Whale	26%			34%
MHI Insular False Killer Whale	37%			
HI Monk Seal	22%		49%	
Guadalupe Fur Seal	65%		21%	
Green Turtle			93%	
Pacific Sea Turtles				

6.6.1.2 *Sources of Exposures of ESA-listed Species during Testing Activities*

The NAEMO model output (based on unprocessed estimates) indicates that five types of testing activities accounted for the majority of exposures to non-impulsive sound sources (Table 45) and four types of testing exercises accounted for the majority of exposures to impulsive sound sources (Table 46).

Table 45. Proportion of unprocessed exposure estimate to non-impulsive sound sources from specific testing activities HSTT Action Area.

Species	Testing Activities				
	Combat System Ship Qualification Trials: USW	AUV AT/FP Mine Counter Measures	Counter Measure Testing	ASW Mission Package Testing	Unmanned Vehicle Development and Payload Testing
Blue Whale				31%	15%
Fin Whale			24%	19%	15%
Humpback Whale	24%		16%		
Sei Whale			14%	21%	15%
Sperm Whale	11%		10%	20%	
MHI Insular False Killer Whale	24%		13%		
HI Monk Seal	17%	22%			
Guadalupe Fur Seal			11%	35%	12%
Green Turtle					
Pacific Sea Turtles		21%		36%	22%

Table 46. Proportion of unprocessed exposure estimate to impulsive sound sources from testing activities HSTT Action Area.

Species	Testing Activities			
	Anti-submarine Tracking Test - Helo	Anti-submarine Tracking Test - MPA	Air-to-Surface Missile Test	Sonobuoy Lot Acceptance Test
Blue Whale	20%	35%	21%	
Fin Whale	23%	28%	23%	
Humpback Whale	29%	27%		
Sei Whale	23%	28%		23%
Sperm Whale	19%	36%		
MHI Insular False Killer Whale	34%	32%		
HI Monk Seal	19%		21%	
Guadalupe Fur Seal		34%	24%	
Green Turtle				
Pacific Sea Turtles	33%	20%		

6.6.1.3 *Exposure and Response of Blue Whales to Acoustic Stressors*

As decribed above we provide three exposure estimates for blue whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.3.1 Unprocessed Exposure Estimate

The model estimates blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 97,942 blue whale exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 9,068 blue whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 216 dB SPL. No exposures to non-impulsive sounds are expected above 217 dB SPL. Hypothetically, in the real word, exposures may exceed or may never even approach this precise dB level. Approximately 103,758 of these exposures will occur in the SOCAL area, 3,054 will be in areas around Hawaii, and 198 will be in HSTT transit lane (Table 47).

Table 47. The top three activities that result in the highest percentages of blue whales unprocessed exposures to non-impulsive acoustic sources in the HSTT Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Blue Whale Exposures						
Helo ASW tracking			23%			
MPA ASW tracking	38%		17%			
Unmanned vehicle development	18%		9%			
Countermeasure testing		100%				
Sonobuoy lot testing	13%					
TRACKEX/TORPEX-surface				37%	60%	43%
TRACKEX/TORPEX-subsurface					11%	13%
Surface ship sonar maintenance					18%	
Undersea warfare exercise						8%
COMPTUEX				32%		
Joint task force exercise				10%		

The activities that are the largest contributors to non-impulsive exposure for blue whales occur in the SOCAL Range Complex. SOCAL has more blue whale exposures because of the high number activities in SOCAL, the involvement of sonar in these training exercises and testing activities, and the relatively high density of blue whales.

Relatively few activities (and number of instances in which activities occur) take place in the Hawaii Range Complex and the transit lane, which explains the relatively low number of exposures in those areas, along with the relatively low density of blue whales here compared to the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors are activities that involve the use of sonar and activity frequency is greater than other activities in this area.

The model estimates that blue whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 2,943 blue whale exposure events annually at sound levels between 120 and 156 dB SPL and 1,665 blue whale exposure events annually to impulsive acoustic sources at levels between 157 and 229 dB SPL. Approximately 4,434 of these exposures are predicted to occur in the SOCAL areas, 173 are predicted to be in the areas around Hawaii, and one is predicted to be in the HSTT Action Area transit lane (Table 48).

Table 48. The top three activities that result in the highest percentages of blue whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing Activities			Training exercises		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Blue Whale Exposures						
Missile (A-S)				10%		14%
Helo ASW tracking	21%					
MPA ASW tracking	24%		21%			
BOMBEX (A-S)						29%
TRACKEX/TORPEX - MPA extended echo ranging sonobuoy				65%		22%
Countermeasure testing			32%			
GUNEX (S-S)-ship-medium caliber					65%	
GUNEX (S-S)-ship-large caliber					35%	
Explosive torpedo testing			21%			
Missile exercise	34%					
Mine neutralization-EOD				7%		

The activities that are the largest contributors to impulsive exposure for blue whales in SOCAL are large contributors due to the large sound fields that the activities produce in the marine environment and the relatively high density of blue whales in the SOCAL Range Complex.

Few impulsive activities (and number of instances in which activities occur) take place in the HSTT Action Area transit lane along with the relatively low density of blue whales here compared to SOCAL result in few exposures.

Blue whale density is relatively low in the Hawaii Range Complex and relatively little testing occurs here, which explains the relatively small proportion of exposures here compared to the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors here result from the large sound fields produced by the activities and the frequency of these activities.

6.6.1.3.2 Processed Exposure Estimates

The processed NAEMO results for non-impulsive acoustic source generate 1,788 instances of blue whales exposed to received levels that would cause them to respond with behaviors that NMFS would classify as harassment (as that term is defined for the purposes of military readiness activities under the Marine Mammal Protection Act of 1972) during training exercises and 125 during testing activities. In addition, there would be another 2,421 instances in which blue whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 303 during testing activities. There would be 116 instances in which blue whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and one blue whale during testing activities.

The processed NAEMO results for impulsive acoustic sources generate two instances of blue whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and one from testing activities. In addition, there will be another three instances in which blue whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity from training exercises and one from testing activities. One blue whale could accumulate energy sufficient to result in permanent shift in hearing sensitivity during training exercises and zero from testing activities. No blue whales would experience GI tract or lung injury, and no blue whales would be killed from acoustic exposure during either training exercises or testing activities.

6.6.1.3.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of blue whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises is 1,786, the number of instances of TTS that would be expected is 2,535, and zero blue whales would be expected to experience PTS.

The estimated number of blue whales that could experience behavioral responses due to impulsive acoustic sources during training exercises is one, the number of animals that would be expected to experience TTS increased to three, and zero blue whales would be expected to experience PTS during training exercises. Therefore, we would expect 4,325 instances of behavioral harassment including instances of TTS.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of blue whales that could experience behavioral responses due to non-impulsive acoustic sources during testing activities was 124, the number of animals that would be expected to experience TTS is 303, and zero blue whales would experience PTS, GI tract or lung injury, and zero blue whale would be killed. The estimated number of blue whales that could experience behavioral responses due to impulsive acoustic sources during testing activities is zero. One blue whale would experience TTS and zero whales would experience PTS, GI tract or lung injury, or be killed. Therefore, we would expect 428 instances of behavioral harassment including instances of TTS.

The vast majority of the modeled instances of TTS and behavioral responses (over 95 percent) would be from the Eastern North Pacific blue whale stock. The majority of acoustic effects to blue whales (and listed mysticetes generally) would be primarily from anti-submarine warfare events involving surface ships and hull-mounted mid-frequency active sonar.

6.6.1.3.4 Response of Blue Whales to Acoustic Stressors

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 midfrequency hearing specialists exposed to mid- or high-frequency sounds; Southall et al. 2007) 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 (see HSTT FEIS/OEIS Table 5.3–2), which is usually the high-frequency cetacean hearing group. Therefore, the mitigation zones are even more protective for blue whales and other low-frquency 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 represent an increase 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 HSTT 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. 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 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.

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. See HSTT FEIS/OEIS Chapter 3.4.3.1.5. 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 patters, 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. 2001; Oleson et al. 2007; Stafford and Moore 2005), recent research has indicated blue whales not only hear mid-frequency active sonar transmissions, in some casesthey respond to those transmissions (Southall et al. 2011a) (Goldbogen et al. 2013a; Melcon et al. 2012b).

As summarized in Section 6.1.10.2.1, 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. 2013a; Melcon et al. 2012b). 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 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 above, perhaps the most significant reponse 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. 2012). 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 anit-submarine warefare is lower than used in the controlled exposure experiments described above, transmitting about once per minute 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. (2007), 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 HSTT 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 temporaily 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, et al. 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 hormomal or physicological 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 HSTT 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 ENP blue whale population, which includes blue whales in SOCAL, has likely reached carrying capacity (Monnahan et al. 2014). As stated previously, Navy training and testing activities, including the use of MFA sonar, have been ongoing in SOCAL for decades. Therefore, any potential temporary behavioral impacts on blue whales from the use of MFA during Navy training and testing activities have not prevented the recovery of the ENP blue whale population. The TTS and behavioral response estimates represent an increase 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 fitiness 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) (Reed 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) (Kight and Swaddle 2011) (Goldbogen et al. 2013) (McGregor et al. 2013) (Reed et al. 2014) (Williams et al. 2014). 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 disturances 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.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year 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. (2014) 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 HSTT Action Area.

6.6.1.4 *Exposure and Response of Fin Whales to Acoustic Stressors*

As decribed previously we provide three exposure estimates for fin whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.4.1 Unprocessed Exposure Estimate

The model estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 44,231 fin whale exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 3,437 fin whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 210 dB SPL. No exposures to non-impulsive sounds are expected above 211 dB SPL. Approximately 38,397 of these exposures will occur in the SOCAL areas, 2,937 will be in areas around Hawaii, and 6,333 will be in HSTT transit lane (Table 49).

Table 49. Activities that result in the highest percentages of fin whales unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Fin Whale						
Helo ASW tracking	20%					
MPA ASW tracking	17%		39%			
Unmanned vehicle development	11%		16%			
Countermeasure testing		100%				
Sonobuoy lot testing			14%			
TRACKEX/TORPEX-surface				36%	59%	44%
TRACKEX/TORPEX-subsurface				13%	13%	15%
Surface ship sonar maintenance					17%	
Rim of the Pacific exercise						9%
COMPTUEX				36%		

The activities that are the largest contributors to non-impulsive exposure for fin whales in the SOCAL Range Complex are large contributors due to the combination of the large number of each of these activities in the SOCAL Range Complex, the normal involvement of significant acoustic energy output into the marine environment. The relatively high density of fin whales in the Hawaii Range Complex transit lane and the prevalvence of certain activities here are the basis for TORPEX activities constituting a large majority of exposures to fin whales here. Relatively little testing occurs in the Hawaii Range Complex, which explains the relatively small number of exposures here compared to the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area.

The model estimates that fin whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 1,067 fin whale exposure events annually at sound levels between 120 and 156 dB SPL and 661 fin whale exposure events annually to impulsive acoustic sources at levels between 157 and 219 dB SPL. Approximately 1,551 of these exposures are predicted to occur in the SOCAL areas, 164 are predicted to be in the areas around Hawaii, and 13 are predicted to be in the HSTT Action Area transit lane (Table 50).

Table 50. Activities that result in the highest percentages of fin whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Fin Whale						
Missile Exercise (A-S)				*12%*		*14%*
MPA ASW tracking	*18%*		*22%*			
Helo ASW tracking	*17%*					
Missile testing (A-S)	*39%*					
Countermeasure testing			*30%*			
BOMBEX (A-S)				*61%*		*27%*
TRACKEX/TORPEX-MPA extended echo ranging sonobuoy						*23%*
GUNEX (S-S)-ship-large caliber					*30%*	
GUNEX (S-S)-ship-medium caliber					*70%*	
GUNEX (A-S)-mine neutralization-EOD				*8%*		
Explosive torpedo testing			*21%*			

The activities that are the largest contributors to impulsive exposure for fin whales in the SOCAL Range Complex are large contributors due to the large sound fields that the activities produce in the marine environment here. Few impulsive activities (and number of instances in which activities occur) take place in the transit lanes, which explains why there are few exposures here even though fin whale density is relatively high compared to the SOCAL Range Complex. Relatively little testing occurs in the Hawaii Range Complex, which explains the relatively small number of exposures here compared to the SOCAL Range Complex. As in the

SOCAL Range Complex, the major contributors here result from the large sound fields produced by the activities and the frequency of these activities is greater than most other impulsive activities in the Hawaii Range Complex.

6.6.1.4.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic source generate 70 instances of fin whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment (as that term is defined for the purposes of the Marine Mammal Protection Act of 1972) during training exercises and 57 during testing activities. In addition, there would be another 986 instances in which fin whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 169 during testing activities. There would be 32 instances in which fin whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and zero during testing activities.

The processed NAEMO results for impulsive acoustic sources generate one instance of a fin whale exposed to receive levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and none during testing activities. In addition, there would be one instance in which a fin whale could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and one during testing activities. No fin whales are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities. No fin whales would experience GI tract or lung injury, and no fin whales would be killed from either training or testing activities.

6.6.1.4.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of fin whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises is 701. In addition, there would be another 1,018 instances in which fin whales could accumulate energy sufficient from non-impulsive sources to result in temporary shifts in hearing sensitivity during training exercises. There would be zero instances in which fin whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises.

The estimated number of fin whales that could experience behavioral responses due to impulsive acoustic sources during training exercises is zero, the number of animals that would be expected to experience TTS is one, and zero fin whales would be expected to experience PTS during training exercises.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of fin whales that could experience behavioral responses due to non-impulsive acoustic sources during testing activities was 56, the number of animals that would be expected to experience TTS is 169, and zero fin whales would experience PTS, GI tract or lung injury, and zero fin whales would be killed.

The estimated number of fin whales that could experience behavioral responses due to impulsive acoustic sources during testing activities is zero. The number of animals that would be expected to experience TTS also zero. Further, zero fin whales are expected to experience PTS, GI tract or lung injury or be killed.

6.6.1.4.4 Response of Fin Whales to Acoustic Stressors

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 exercises 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 1982b; Watkins 1981c). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15 Hz to 40 Hz range. Ketten (1997) 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 1997). 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. 2013c; 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 in Section 6.6.1.3.4, 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 (see Section 6.6.1.3.4). 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 abudance in the California Current area, which includes the Southern California Range Complex. The authors predict continued increases in fin whale numbers over the next decade, and that perhaps fin whale densities are reaching "current ecosystem limits." As stated previously, Navy training and testing activities, including the use of MFA sonar, have been ongoing in SOCAL for decades. Therefore, 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. The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year 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.

6.6.1.5 *Exposure and Response of Western North Pacific Gray Whales to Acoustic Stressors*

The Navy did not have sufficient information to distinguish between ESA-listed western north Pacific gray whales and unlisted eastern north Pacific gray whales at the time they ran the NAEMO. Following reports of several listed western north Pacific gray whales along the west coast of the United States, we made a request to the Navy to evaluate the likelihood of of the proposed training exercises and testing activities to affect western north Pacific gray whales.

Evaluating the likely occurance of western north Pacific gray whales in the HSTT Action Area we determine that it is possible that a western north Pacific gray whale would transit southbound and northbound through the SOCAL Range Complex portion of the HSTT Action Area. Mate and Urban-Ramirez (2003) documented a northbound coastal migration route for a satellite tagged gray whale with the exception of the offshore leg between Mexico and the Channel Islands. This offshore route through the SOCAL Range Complex is consistent with previous studies showing both a near shore migration route for gray whales and an offshore route that passes east and west of San Clemente Island (Dohl et al. 1981 as summarized in Bonnell and Dailey 1993, Sumich and Show 2011). Average transit rate reported by Mate and Urban-Ramirez was 5.2 km/hr. Therefore, expected residence within the SOCAL Range Complex would be on the order of hours to days during each of the southbound and northbound legs of gray whale migrations.

To estimate the number of western north Pacific gray whales that would be exposed to active sonar and other acoustic sources, the Navy propotionally allocated the eastern north Pacific gray whale estimate based on the Weller et al (2013) estimate of western north Pacific gray whales along the U.S. West Coast of 23 animals and the estimated eastern north Pacific gray whale population of 19,126 (Carretta et al. 2013) such that 23/19,126 = 0.12 percent. Using this allocation, we calculated the following processed and applied mitigation estimates.

6.6.1.5.1 Processed Exposure Estimate

The proportional allocation by the Navy did not distinguish between non-impulsive and impulsive acoustic sources. Combined, the Navy estimates that there would be three instances of western north Pacific gray whales exposed to received levels that would cause them to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. In addition, there would be another seven instances in which western north Pacific gray whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and two during testing activities. There would be zero instances in which western north Pacific gray whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises or testing activities (Navy 2013f).

6.6.1.5.2 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of western north Pacific gray whales that could experience behavioral responses due to non-impulsive acoustic sources during training exerises is two. Seven western north Pacific gray whales could experience TTS due to non-impulsive acoustic sources during training exercises. Zero western north Pacific gray whales would be expected to experience PTS or other injury during training exercises (Navy 2013f).

The estimated number of western north Pacific gray whales that could experience behavioral responses from impulsive acoustic sources during training exercises is one. Zero western north Pacific gray whales could experience TTS during training exercises. Zero would experience PTS, GI tract or lung injury, and zero western north Pacific gray whale would be killed during training exercises.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of western north Pacific gray whales that could experience behavioral responses due to non-impulsive acoustic sources during testing activities was zero. The number of animals that would be expected to experience TTS from non-impulsive acoustic sources during testing activities is two. Zero western north Pacific gray whales would experience PTS or be killed during testing activities.

The estimated number of western north Pacific gray whales that could experience behavioral responses, TTS of PTS due to impulsive acoustic sources during testing activities is zero.

6.6.1.6 *Response of Western North Pacific Gray Whales to Acoustic Stressors*

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to western north Pacific gray whales.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 gray whale hearing so we assume that gray whale vocalizations are partially representative of their hearing sensitivities. Auditory structure suggests hearing is attuned to low frequencies (Ketten 1992a; Ketten 1992b). Responses of free-ranging and captive individuals to playbacks in the 160 Hz to 2 kHz range demonstrate the ability of individuals to hear within this range (Buck and Tyack 2000; Cummings and Thompson 1971a; Dahlheim and Ljungblad 1990; Moore and Clark 2002; Wisdom et al. 2001). Responses to low-frequency sounds stemming from oil and gas activities also support low-frequency hearing (Malme et al. 1986b; Moore and Clark 2002).

Results of studies on blue whales (Goldbogen et al. 2013c; Southall et al. 2011a) (Melcon et al. 2012b), which have similar auditory physiology compared to gray 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 in Section 6.6.1.3.4, 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 western North Pacific gray whales to TTS to be similar to that of blue whales (see Section 6.6.1.3.4). The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of Western North Pacific gray whales to reduce the viability of this population or its ability to recover. 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 this population.

6.6.1.7 *Exposure and Response of Humpback Whales to Acoustic Stressors*

As decribed previously we provide three exposure estimates for humpback whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.7.1 Unprocessed Exposure Estimate

The model estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year.

The NAEMO provided an unprocessed estimate of 185,275 humpback whale exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 13,541 humpback whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 210 dB SPL. No exposures to non-impulsive sounds are expected above 211 dB SPL. Approximately 29,163 of these exposures will occur in the SOCAL areas, 169,383 will be in areas around Hawaii, and 270 will be in HSTT transit lane (Table 51).

Table 51. Activities that result in the highest percentages of humpback whales unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Humpback Whale						
Helo ASW tracking			15%			
MPA ASW tracking	40%		20%			
Unmanned vehicle development			13%			
Countermeasure testing		100%				
ASW mission package testing	16%					
TRACKEX/TORPEX-surface				36%	58%	25%
TRACKEX/TORPEX-subsurface					12%	
Surface ship sonar maintenance					18%	
Undersea warfare exercise						9%
COMPTUEX				33%		
Submarine navigation						24%
Sonobuoy lot testing	14%					
Joint task force exercise				13%		

The activities that are the largest contributors to non-impulsive exposure for humpback whales in the SOCAL Range Complex are large contributors due to the combination of the large number of each of these activities in the SOCAL Range Complex, the seasonal presence of humpback whales here, and the normal involvement of significant acouistic energy output into the marine environment. The relatively low density of humpback whales in the transit lane and the prevalvence of certain activities here are the basis for TRACKEX/TORPEX activities constituting a large majority of exposures to humpback whales here. Relatively little testing occurs in the Hawaii Range Complex but humpback whale density here is generally higher here than in the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area.

Because humpback whales migrate to the north in the summer, impacts are predicted only for the cool season in the HSTT Action Area. While the humpback breeding areas around Hawaii are important, major training exercises involving mid-frequency active sonar in the humpback breeding areas are rare and infrequent.

The model estimates that humpback whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 5,972 humpback whale exposure events annually at sound levels between 120 and 156 dB SPL and 1,133 humpback whale exposure events annually to impulsive acoustic sources at levels between 157 and 219 dB SPL. Approximately 900 of these exposures are predicted to occur in the SOCAL areas, 6,204 are predicted to be in the areas around Hawaii, and 30 are predicted to be in the HSTT Action Area transit lane (Table 52).

The activities that are the largest contributors to impulsive exposure for humpback whales in the SOCAL Range Complex are large contributors due to the large sound fields that the activities produce in the marine environment here. Few impulsive activities (and number of instances in which activities occur) take place in the transit lane, which explains the few number of exposures here, although humpback whale density is relatively low compared to the SOCAL Range Complex. Relatively little testing occurs in the Hawaii Range Complex, but humpback whale density is seasonally much higher than in other areas, resulting in a larger number of exposures here than in the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors here result from the large sound fields produced by the activities and the frequency of these activities is greater than most other impulsive activities in the Hawaii Range Complex.

Table 52. Activities that result in the highest percentages of humpback whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Humpback Whale						
Missile exercise (A-S)				11%		20%
Missile testing (A-S)	30%		15%			
MPA ASW tracking	22%		38%			
MCM mission package testing			15%			
Helo ASW tracking	19%					
BOMBEX (A-S)				58%		33%
TRACKEX/TORPEX - MPA extended echo ranging sonobuoy						27%

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
GUNEX (S-S)-ship-large caliber					87%	
GUNEX (S-S)-ship-medium caliber					13%	
Mine neutralization-EOD				9%		

6.6.1.7.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate 3,300 instances of humpback whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 273 during testing activities. In addition, there would be another 5,912 instances in which humpback whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 640 during testing activities. There would be 64 instances in which humpback whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and four humpback whales during testing activities.

The processed NAEMO results for impulsive acoustic sources generate one instance of a humpback whale exposed to received levels that cause it to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. In addition, there will be another one instance in which humpback whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and seven during testing activities. No humpback whales are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities. No humpback whales would experience GI tract or lung injury, and no humpback whales would be killed during either training exercises or testing activities.

6.6.1.7.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of humpback whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises is 3,299 and 272 during testing activities. The estimated number of humpback whales that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was 5,974 during training exercises and 649 during testing activities. Zero humpback whales would be expected to experience PTS from either training exercises or testing activities from non-impulsive acoustic sources.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of humpback whales that could experience behavioral responses due to impulsive acoustic sources during training exercises is zero. Zero animals could be expected to experience TTS, PTS, GI tract or lung injury, or would be killed. Therefore, we conclude that acoustic stressors during testing activities will result in 927 instances of behavioral harassment.

The estimated number of humback whales that could experience behavioral responses due to impulsive acoustic sources during testing activities is zero, the number of animals that would be expected to experience TTS is six. Zero humpback whales would be expected to experience PTS or be killed during either training exercises or testing activities.

6.6.1.7.4 Response of Humpback Whales to Acoustic Stressors

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. 1986). These sounds are attractive and appear to rally animals to the feeding activity (D'Vincent et al. 1985; 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. 1995c; 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. 1995c; Thompson et al. 1986). 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. (2006) 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 HSTT 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 in Section 6.6.1.3.4 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 (see Section 6.6.1.3.4).

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 350Hz (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 Navy has agreed that training exercises utilizing mid-frequency active sonar in the designated Humpback Whale Cautionary Area would require a much higher level of clearance than is normal practice in planning and conducting mid-frequency active sonar training. Furthermore, no reported cases of harmful effects to humpback whales attributed to mid-frequency active sonar use have occurred during the Navy's 40-plus years of training in the waters off the Hawaiian Islands. Coincident with this use of mid-frequency active sonar, abundance estimates reflect an annual increase in the humpback whale stock (Mobley Jr 2001; Mobley Jr. 2003). A recent long-term study of humpback whales in Hawaiian waters shows long-term fidelity to the Hawaiian winter grounds, with many showing sighting spans ranging from 10 to 32 years (Herman et al. 2011). The overall abundance of humpback whales in the north Pacific has continued to increase and is now greater than some pre-whaling abundance estimates (Barlow et al. 2011a). The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year 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.

6.6.1.8 *Exposure and Response of Sei Whales to Acoustic Stressors*

As decribed previously we provide three exposure estimates for sei whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.8.1 Unprocessed Exposure Estimate

The model estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 9,674 sei whale exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 949 sei whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 210 dB SPL. No exposures to non-impulsive sounds are expected above 211 dB SPL. Approximately 3,496 of these exposures will occur in the SOCAL areas, 6,895 will be in areas around Hawaii, and 232 will be in HSTT transit lane (Table 53).

Table 53. Activities that result in the highest percentages of sei whales unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Sei Whale						
Helo ASW tracking			17%			
MPA ASW tracking	30%		18%			
Unmanned vehicle development	16%		13%			
Countermeasure testing		100%				
ASW mission package testing	16%					
TRACKEX/TORPEX-surface				38%	60%	40%
TRACKEX/TORPEX-subsurface					12%	13%
Surface ship sonar maintenance					17%	
Rim of the Pacific exercise						10%
COMPTUEX				27%		
Joint task force exercise				14%		

The activities that are the largest contributors to non-impulsive exposure for sei whales in the SOCAL Range Complex are large contributors due to the combination of the large number of each of these activities in the SOCAL Range Complex, the seasonal and generally low presence of sei whales here, and the normal involvement of significant acoustic energy output into the marine environment. The prevalvence of certain activities in the Hawaii Range Complex transit lane are the basis for TRACKEX/TORPEX activities constituting a large majority of exposures to sei whales here. Relatively little testing occurs in the Hawaii Range Complex but sei whale density here is generally higher here than in the SOCAL Range Complex. As in the SOCAL Range Complex, the major Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area.

The model estimates that sei whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 362 sei whale exposure events annually at sound levels between 120 and 156 dB SPL and 145 sei whale exposure events annually to impulsive acoustic sources at levels between 157 and 210 dB SPL. Approximately 228 of these exposures are predicted to occur in the SOCAL areas, 278 are predicted to be in the areas around Hawaii, and 1 is predicted to be in the HSTT Action Area transit lane (Table 54).

Table 54. Activities that result in the highest percentages of sei whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Sei Whales						
Missile exercise (A-S)				12%		19%
MPA ASW tracking			30%			
Helo ASW tracking	21%					
Missile testing (A-S)	25%					
Sonobuoy lot acceptance testing	19%					
BOMBEX (A-S)				51%		29%
TRACKEX/TORPEX - MPA extended echo ranging sonobuoy						25%
GUNEX (S-S)-ship-large calibre					29%	
GUNEX (S-S)-ship-medium calibre					71%	
Countermeasure testing			24%			
Explosive torpedo testing			18%			
Mine neutralization-EOD				17%		

The activities that are the largest contributors to impulsive exposure for sei whales in the SOCAL Range Complex are large contributors due to the large sound fields that the activities produce in the marine environment here. Few impulsive activities (and number of instances in which activities occur) take place in the Hawaii Range Complex transit lane, which explains the few number of exposures here, although sei whale density is relatively low compared to the SOCAL Range Complex. Relatively little testing occurs in the Hawaii Range Complex, but sei whale density is seasonally high in this area, resulting in a roughly equivalent number of exposures compared to the SOCAL Range Complex. As in the SOCAL Range Complex, the major contributors here result from the large sound fields produced by the activities and the frequency of these activities is greater than most other impulsive activities in the Hawaii Range Complex.

6.6.1.8.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic source generate 227 instances of sei whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 17 during testing activities. In addition, there would be another 388 instances in which sei whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 36 during testing activities. There would be 18 instances in which sei whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and zero sei whales during testing activities.

The processed NAEMO results for impulsive acoustic sources generate zero instances of a sei whale exposed to received levels that cause it to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. In addition, there were no instances in which sei whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during either training exercises or testing activities. No sei whales are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities No sei whales would experience GI tract or lung injury, and no sei whales would be killed for training or testing activities.

6.6.1.8.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of sei whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises was 226 and 16 during testing activities. The estimated number of sei whales that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was 404 during training exercises and 35 during testing activities. Zero sei whales would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of sei whales that could experience behavioral responses due to impulsive acoustic sources was zero for both training exercises and testing activities. Zero sei whales would be expected to experience TTS, PTS, GI tract or lung injury, or would be killed for both training exercises and testing activities.

6.6.1.8.4 Response of Sei Whales to Acoustic Stressors

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. 2013c; 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 described in Section 6.6.1.3.4, 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 (see Section 6.6.1.3.4). The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year 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 population ot their ability to recover. 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.

6.6.1.9 *Exposure and Response of Sperm Whales to Acoustic Stressors*

As decribed previously we provide three exposure estimates for sperm whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.9.1 Unprocessed Exposure Estimate

The model output estimates that sperm whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 111,383 sperm whale exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 7,304 sperm whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 210 dB SPL. No exposures to non-impulsive sounds are expected above 211 dB SPL. Approximately 65,554 of these exposures will occur in the SOCAL areas, 52,208 will be in areas around Hawaii, and 926 will be in HSTT transit lane (Table 55).

Table 55. Activities that result in the highest percentages of sperm whales unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Sperm Whale						
Helo ASW tracking			14%			
MPA ASW tracking	41%		24%			

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Unmanned vehicle development	15%		12%			
Countermeasure testing		100%				
TRACKEX/TORPEX-surface				38%	60%	38%
TRACKEX/TORPEX-subsurface					13%	16%
Surface ship sonar maintenance					15%	
ASW mission package testing	17%					
COMPTUEX				33%		
Submarine navigation exercise						10%
Joint task force exercise				12%		

The activities that are the largest contributors to non-impulsive exposure for sperm whales in the SOCAL Range Complex are large contributors due to the combination of the large number of each of these activities in the SOCAL Range Complex, the high sperm whale density, and the normal involvement of significant acoustic energy output into the marine environment. The prevalvence of certain activities in the Hawaii Range Complex transit lane are the basis for TRACKEX/TORPEX activities constituting a large majority of exposures to sperm whales here. As in the SOCAL Range Complex, the major Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area.

The model estimates that sperm whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 1,681 sperm whale exposure events annually at sound levels between 120 and 156 dB SPL and 1,822 sperm whale exposure events annually to impulsive acoustic sources at levels between 157 and 219 dB SPL. Approximately 1,393 of these exposures are predicted to occur in the SOCAL areas, 2,101 are predicted to be in the areas around Hawaii, and 10 are predicted to be in the HSTT Action Area transit lane (Table 56).

Table 56. Activities that result in the highest percentages of sperm whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Sperm Whale						
Missile exercise (A-S)				12%		11%
MPA ASW tracking	27%		33%			
Helo ASW tracking	22%					
Missile testing (A-S)	21%					
BOMBEX (A-S)				57%		23%
TRACKEX/TORPEX - MPA extended echo ranging sonobuoy						37%
GUNEX (S-S)-ship-large caliber					25%	
GUNEX (S-S)-ship-medium caliber					75%	
MCM mission package testing			26%			
Sinking exercise				8%		
Countermeasure testing			14%			

The activities that are the largest contributors to impulsive exposure for sperm whales in the SOCAL Range Complex are large contributors due to the large sound fields that the activities produce in the marine environment here. Few impulsive activities (and number of instances in which activities occur) take place in the Hawaii Range Complex transit lane, which explains the few number of exposures here. Relatively little testing occurs in the Hawaii Range Complex. As in the SOCAL Range Complex, the major contributors here result from the large sound fields produced by the activities and the frequency of these activities is greater than most other impulsive activities in the Hawaii Range Complex.

6.6.1.9.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic source generate 3,151 instances of sperm whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 140 during testing activities. In addition, there would be another 183 instances in which sperm whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 123 during testing activities. There would be zero instances in which sperm whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and three sperm whales during testing activities.

The processed NAEMO results for impulsive acoustic sources generate zero instances of a sperm whale exposed to receive levels that cause it to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. Also there will be zero instances in which sperm whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during either training exercises or testing activities. No sperm whales are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities No sperm whales would experience GI tract or lung injury, and no sperm whales would be killed from either training exercises or testing activities.

6.6.1.9.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of sperm whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises was 3,150 and 139 during testing activities. The estimated number of sperm whales that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was 182 during training exercises and 124 during testing activities. Zero sperm whales would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of sperm whales that could experience behavioral responses due to impulsive acoustic sources during training exercises and testing activities remained at zero. Zero sperm whales would experience PTS, GI tract or lung injury, and no sperm whale would be killed.

6.6.1.9.4 Response of Sperm Whales to Acoustic Stressors

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 midfrequency hearing specialists exposed to mid- or high-frequency sounds; Southall et al. 2007) 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. 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 kHz to 60 kHz. Sperm whales vocalize in high- and mid-frequency ranges; most of the energy of sperm whales clicks is concentrated at 2 kHz to 4 kHz and 10 kHz to 16 kHz. Other studies indicate sperm whales' wide-band clicks contain energy between 0.1 kHz and 20 kHz (Goold and Jones 1995; Weilgart and Whitehead 1993). 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.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. 2007). Therefore, TTS associated with mid-frequency sonar could conceivably interfere with an animal's ability to hear vocalizations or most other biologically important sounds.

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 sperm whales, 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 frees wimming 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 sperm whale 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.

Therefore, for sperm whales and other ESA-listed marine mammals analyzed in this Opinion, even though the modeled TTS estimates represent an increase 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 HSTT Action Area.

The Navy's acoustic analysis indicates that exposures of sperm whales to sound levels are likely to result in behavioral harassment from sonar or other active acoustic stressors during training and testing activities. These exposure instances are anticipated to be in the form of behavioral harassment and no injurious takes of sperm whales from sonar, other active acoustic stressors, or explosives are anticipated. Sperm whales have shown resilience to acoustic and human disturbance, although they may react to sound sources and activities within a few kilometers. Sperm whales that are exposed to activities that involve the use of sonar and other active acoustic sources may alert, ignore the stimulus, avoid the area by swimming away or diving, or display aggressive behavior. Some (but not all) sperm whale vocalizations might overlap with the mid-frequency active sonar or high frequency active sonar frequency ranges, which could temporarily decrease an animal's sensitivity to the calls of conspecifics or returning echolocation signals. However, we do not anticipate TTS of a long duration or severe degree to occur as a result of exposure to these sonar sources. The majority of exposure instances are expected to be in the form of mild responses.

Vocal changes in response to anthropogenic noise can occur across the range of sound production modes used by marine mammals, such as whistling, echolocation click production, calling, and singing. Changes may result in response to a need to compete with an increase in background noise or may reflect an increased vigilance or startle response. Odontocetes such as killer whales off the northwestern coast of the U.S. have been observed to increase the duration of primary calls once a threshold in observing 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). In contrast, both sperm and pilot whales potentially ceased sound production during the Heard Island feasibility test (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.

There is some evidence of disruptions of clicking and behavior from sonars (Goold 1999; Watkins 1985), pingers (Watkins and Schevill 1975), the Heard Island Feasibility Test (Bowles et al. 1994b), 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 1975). Goold (1999) 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 Schevill (1975) 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 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). Results of experimental playbacks of killer whale sounds to five individual male sperm whales in the Norwegian Sea showed that, rather than diving away from the perceived predator, sperm whales responded by interrupting their foraging or resting dives and returning to the surface, changing their vocal production, and initiating a surprising degree of social behavior (Cure et al. 2013).

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 μPa from impulsive sounds produced by 1 g TNT detonators (Madsen and Mohl 2000). Richardson et al. (1995c) 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 μPa 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. 1994b).

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). 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.

Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses produced by echosounders and submarine sonar (Watkins 1985; Watkins and Schevill 1975). 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). Furthermore, because of their apparent role as important predators of mesopelagic squid and fish, changes in their abundance could affect the distribution and abundance of other marine species.

Disruption of feeding behavior can be difficult to correlate with anthropogenic sound exposure. Responses may vary depending on the behavioral state of the individual or group of animals. For example, sperm whales engaged in foraging dives did not abandon dives when exposed to distant signatures of seismic airguns (Madsen et al. 2006). Conversely, preliminary results of controlled playback of sonar may indicate feeding/foraging disruption of killer whales and sperm whales (Miller et al. 2011).

Richardson et al. (1995) noted that avoidance (temporary displacement of an individual from an area) reactions are the most obvious manifestations of disturbance in marine mammals. It is qualitatively different from the startle or flight response, but also differs in the magnitude of the response (i.e., directed movement, rate of travel, etc.). Oftentimes avoidance is temporary, and animals return to the area once the noise has ceased. In the Caribbean, sperm whales avoided exposure to mid-frequency submarine sonar pulses, in the range of 1000 Hz to 10,000 Hz (IWC 2005).

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.

The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of sperm whales to reduce the viability of these populations or their ability to recovery. We anticipate 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.

6.6.1.10 *Exposure and Response of Main Hawaiian Island False Killer Whales to Acoustic Stressors*

The NAEMO output for false killer whales included multiple stocks; only one of which is listed under the ESA, the MHI insular false killer whales. As such, to estimate the number of ESA-listed false killer whales may be affected by the action we prorated the values based on the proportional estimates of whales from each stock. Per previous sections we provide three exposure estimates for MHI insular false killer whales: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.10.1 Unprocessed Exposure Estimate

The model output estimates that MHI insular false killer whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 1,337 MHI insular false killer whale (7 percent of 19,101) exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 98 MHI insular false killer whale exposure events annually to non-impulsive acoustic sources at levels between 157 and 210 dB SPL. Zero exposures to non-impulsive sounds are expected above 211 dB SPL. All of these exposures are predicted to occur in the areas around Hawaii (Table 57). The major Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area.

Table 57. Activities that result in the highest percentages of unprocessed exposures of false killer whales to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
False Killer Whale						
Helo ASW tracking			15%			
MPA ASW tracking			14%			
Submarine sonar testing/maintenance			16%			
TRACKEX/TORPEX-surface						44%
TRACKEX/TORPEX-subsurface						14%
Undersea warfare exercise						10%

The model estimates that MHI insular false killer whales will be exposed to explosions and other impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 42 MHI insular false killer whale exposure events annually at sound levels between 120 and 156 dB SPL and 10 MHI insular false killer whale exposure events annually to impulsive acoustic sources at levels between 157 and 216 dB SPL. All of these exposures are predicted to occur in the areas around Hawaii (Table 58).

The major contributors in the Hawaii Range Complex result from the large sound fields produced by the activities and the frequency of these activities is greater than most other impulsive activities in the Hawaii Range Complex. In addition, these activities are generally closer to shore, which overlaps the distribution of MHI insular false killer whales more than other activities.

Table 58. Activities that result in the highest percentages of false killer whales unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
False Killer Whale						
Missile exercise (A-S)						19%
MPA ASW tracking			35%			
Explosive torpedo testing			21%			
Countermeasure testing			17%			
BOMBEX (A-S)						30%
TRACKEX/TORPEX - MPA extended echo ranging sonobuoy						24%

6.6.1.10.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate 46 instances of MHI insular false killer whales exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 2 during testing activities. In addition, there would be another 3 instances in which MHI insular false killer whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training and 2 during testing activities. There would be zero instances in which MHI insular false killer whales could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during either training exercises or testing activities.

The processed NAEMO results for impulsive acoustic sources generate zero instances of a MHI insular false killer whale exposed to received levels that cause it to respond with behaviors that

NMFS would classify as harassment during either training exercises or testing activities. In addition, there were zero instances in which MHI insular false killer whales could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during either training exercises or testing activities. No MHI insular false killer whales are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities. No MHI insular false killer whales would experience GI tract or lung injury, and no MHI insular false killer whales would be killed from either training exercises or testing activities.

6.6.1.10.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of MHI insular false killer whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises remained at 46 and 2 during testing activities. The estimated number of MHI insular false killer whales that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources remained at 3 during training exercises and 2 during testing activities. No MHI insular false killer whales would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of MHI insular false killer whales that could experience behavioral responses due to impulsive acoustic sources remained at none, and the number of animals that would be expected to experience TTS was also none for either training exercises or testing activities. No MHI insular false killer whales would experience PTS, GI tract or lung injury, and no MHI insular false killer whales would be killed from either training exercises or testing activities.

6.6.1.10.4 Response of Main Hawaiian Islands Insular False Killer Whales to Acoustic Stressors

Except as provided herein, our analysis of the modeled TTS and behavioral responses for sperm whales applies equally to MHI insular false killer whales. Based on the evidence available, we would expect MHI insular false killer whales that are exposed to mid-frequency active sonar in the Hawaii Range Complex to engage in horizontal movements that would allow them to avoid continued exposure. At the same time, we would expect MHI insular false killer whales to experience impaired hearing (e.g. TTS) and/or communication because they vocalize at frequencies that overlap with those of the mid-frequency active sonar systems the Navy plans to employ during training on the Hawaii Range Complex. To preserve the saliency of their vocalizations and the coherence of their social interactions, MHI insular false killer whales might have to make one or more of the vocal adjustments discussed earlier in this narrative. Because any reductions in the active space of whale vocalizations that result from active sonar transmissions associated with the proposed missions would be temporary and episodic, any vocal adjustments MHI insular false killer whales would have to make would also be temporary.

Studies on a captive false killer whales have demonstrated an ability to proactively change hearing sensitivity, apparently for protection, when a warning sound was provided prior to a more intense sound and that the animal learned to change its hearing sensitivity when warned that a loud sound was about to arrive. (Nachtigall and Supin 2013). Controlled exposure experiments in the Bahamas found that after each MFA reception of simulated MFA sonar sounds (3.1 to 3.4 kHz, source level 160 to 211 dB re 1 μPa) false killer whales increased their whistle rate and produced whistles that were more similar in their frequency characteristics to MFA sonar (DeRuiter et al. 2013a).

Main Hawaiian islands insular false killer whale hearing and vocalization frequencies overlap those used by MFA sonar similarly to sperm whales. Therefore, without more specific information, we assume the response of MHI insular false killer whales to TTS to be similar to that of sperm whales (see Section 6.6.1.9.4). The TTS and behavioral response estimates represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of MHI insular false killer whales to reduce the viability or recovery of this population. We anticipate temporary behavioral responses and TTS, 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 this population.

6.6.1.11 *Exposure and Response of Guadalupe Fur Seals to Acoustic Stressors*

As decribed previously for most of the whale species we provide three exposure estimates for Guadalupe fur seals; unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.11.1 Unprocessed Exposure Estimate

The model output estimates that Guadalupe fur seals will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 86,022 Guadalupe fur seal exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 6,649 Guadalupe fur seal exposure events annually to non-impulsive acoustic sources at levels between 157 and 216 dB SPL. No exposures to non-impulsive sounds are expected above 214 dB SPL. Approximately 91,286 of these exposures will occur in the SOCAL areas, zero will be in areas around Hawaii, and 1,385 will be in HSTT transit lane (Table 59).

Table 59. Activities that result in the highest percentages of Guadalupe fur seals unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Guadalupe fur seals						
ASW mission package testing	19%					
MPA ASW tracking	36%					
Unmanned vehicle development	15%					
Countermeasure testing		100%				
TRACKEX/TORPEX-surface				36%	64%	
TRACKEX/TORPEX-subsurface					9%	
Surface ship sonar maintenance					16%	
COMPTUEX				32%		
Joint task force exercise				12%		

The major SOCAL Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area. However, as the species' density is greater closer to shore, activities that tend to be undertaken closer to shore also tend to expose Guadalupe fur seals more than activities that tend to be more offshore or wide-ranging in nature. Eight impulsive exposures potentially resulting in take in the SOCAL Range Complex (Table 60).

Table 60. Activities that result in the highest percentages of Guadalupe fur seals unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Guadalupe Fur Seal						
Missile exercise (A-S)				15%		
MPA ASW tracking	18%					
Missile testing (A-S)	36%					
Helo ASW tracking	16%					
BOMBEX (A-S)				56%		
GUNEX (S-S)-ship-large caliber					30%	

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
GUNEX (S-S)-ship-medium caliber					70%	
Sinking exercise				7%		

The major SOCAL Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area. However, as the species' density is greater closer to shore, activities that tend to be undertaken closer to shore also tend to expose Guadalupe fur seals more than activities that tend to be more offshore or wide-ranging in nature.

6.6.1.11.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate 2,596 instances of Guadalupe fur seals exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 270 during testing activities. In addition, there would be another seven instances in which Guadalupe fur seals could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and zero during testing activities. There would be one instance in which Guadalupe fur seals could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and zero Guadalupe fur seals during testing activities.

The processed NAEMO results for impulsive acoustic sources generate zero instance of a Guadalupe fur seal exposed to received levels that cause it to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. In addition, there will be zero instances in which Guadalupe fur seals could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during either training exercises or testing activities. No Guadalupe fur seals are expected to accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises or testing activities. One Guadalupe fur seal during training exercises and one instance during testing activities of animals that would experience GI tract or lung injury. No Guadalupe fur seals would be killed from either training exercises or testing activities.

6.6.1.11.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of Guadalupe fur seals that could experience behavioral responses due to non-impulsive acoustic sources during training exercises was 2,596 and 269 during testing activities. The estimated number of Guadalupe fur seals that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was seven during training exercises and zero during testing activities. Zero Guadalupe fur seals would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of Guadalupe fur seals that could experience behavioral responses due to impulsive acoustic sources from either training exercises or testing activities was zero. Zero Guadalupe fur seals would experience PTS, slight lung injury, GI tract injury, or be killed.

6.6.1.11.4 Response of Guadalupe Fur Seals to Acoustic Stressors

There is no published information on the hearing range of Guadalupe fur seals, although it is most likely similar to other fur seals. Some evidence suggests northern fur seals produce underwater clicks, and in-air barking, coughing, and roaring sounds (Richardson et al. 1995c; Schusterman 1978). However, Schusterman et al. (2000) reviewed available evidence on the potential for pinnipeds to echolocate and indicated that pinnipeds have not developed specialized sound production or reception systems required for echolocation. Instead, it appears pinnipeds have developed alternative sensory systems (e.g., visual, tactile) to effectively forage, navigate and avoid predators underwater. The underwater hearing range of the northern fur seal ranges from 0.5Hz to 40 kHz (Babushina et al. 1991; Moore and Schusterman 1987) and the threshold is 50 to 60 dB re 1 μPa (Moore and Schusterman 1987). The best underwater hearing occurs between 4 and 17 to 28 kHz (Babushina et al. 1991; Moore and Schusterman 1987). The maximum sensitivity in air is at 3 to 5 kHz for northern fur seals (Babushina et al. 1991). Since fur seals do not rely solely on acoustic cues for underwater feeding and navigation, it is anticipated TTS will not result in fitness consequences to individuals or the populations to which they belong.

Behavioral responses can range from a mild orienting response, or a shifting of attention, to flight and panic. Research and observations show that pinnipeds in the water are tolerant of anthropogenic noise and activity. 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 exposure. To result in significant fitness consequences we would have to assume that an individual animal 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. 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 et al 1993, as cited in New et al. 2014). Similarly, since Guadalupe fur seals have near continuous opportunities to feed throughout the year, they are unlikely to be affected by short-term variations in prey availability.

Therefore, behavioral responses of Guadalupe fur seals to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population.

The Navy plans to conduct training exercises and testing activities on the SOCAL Range Complex that are likely to cause some individual Guadalupe fur seals to experience changes in their behavioral states that, in some circumstances, might have adverse consequences for free-ranging animals (Frid 2003; Frid and Dill 2002; Papouchis et al. 2001). In most respects, behavioral effects from noise-induced threshold shifts in Guadalupe fur seals follow trends similar to those observed in other mammals. The data are characterized by variable shifts at low noise levels; increasing shifts with increasing exposure duration, sound levels, and sound exposure levels; and complete, rapid recovery of sensitivity. These types of responses are not likely to alter the physiology, behavioral ecology, and social dynamics of individual Guadalupe fur seals in ways or to a degree that would reduce their fitness because behavioral responses and TTS are temporary and pinnipeds do not rely solely on acoustic cues for underwater feeding efforts.

Guadalupe fur seals have been exposed to Navy training exercises and testing activities on the SOCAL Range Complex of the HSTT Action Area, including vessel traffic, aircraft traffic, active sonar, and underwater detonations, for more than a generation. Despite this exposure, the Guadalupe fur seal population has been increasing at a rate of approximately 13.7 percent per year. At that rate of growth the population should double every five years. Although we do not know if the Guadalupe fur seal population might have increased at a much higher rate if they had not been exposed to Navy training exercises and testing activities, this rate suggests that the number of Guadalupe fur seals would continue to increases despite being exposed to stressors associated with these training exercises and testing activities. As a result, the Guadalupe fur seals' probable responses to exposure to active sonar and underwater detonations are not likely to reduce the current or expected future reproductive success of Guadalupe fur seals or reduce the rates at which they grow, mature, or become reproductively active. The response estimates for Guadalupe fur seals represent an increase 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 fitiness of affected individuals, is not expected to change. Therefore, these exposures are not likely to reduce the abundance, reproduction rates, and growth rates (or increase variance in one or more of these rates) of the populations those individuals represent.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of Guadalupe fur seals to reduce the viability of this population. We anticipate 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 this population.

6.6.1.12 *Hawaiian Monk Seal*

As decribed previously for most of the whale species we provide three exposure estimates for Hawaiian monk seals: the unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.12.1 Unprocessed Exposure Estimate

The model output estimates that Hawaiian monk seals will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 23,328 Hawaiian monk seal exposure events annually to non-impulsive sounds at levels between 120 and 156 dB SPL and 2,628 Hawaiian monk seal exposure events annually to non-impulsive acoustic sources at levels between 157 and 201 dB SPL. Zero exposures to non-impulsive sounds are expected above 202 dB SPL. None of these exposures will occur in the SOCAL or transit lane areas (Table 61).

Table 61. Activities that result in the highest percentages of Hawaiian monk seals unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Hawaiian Monk Seal						
Submarine sonar maintenance						44%
Non-explosive torpedo testing			15%			
Unmanned vehicle development			11%			
AUV AT/FP mine countermeasures			38%			
TRACKEX/TORPEX-surface						22%
Undersea warfare exercise						8%

The major Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area. However, as the species' density is greater closer to shore, activities that tend to be undertaken closer to shore also tend to expose Hawaiian monk seals more than activities that tend to be more offshore or wide-ranging in nature. Table 62 summarizes the Activities that result in the highest percentages of Hawaiian monk seal unprocessed exposures to impulsive acoustic sources in the HSTT Action Area.

Table 62. Activities that result in the highest percentages of Hawaiian monk seal unprocessed exposures to impulsive acoustic sources in the HSTT Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Hawaiian Monk Seal						
Missile exercise (A-S)						28%
Mine neutralization-EOD						10%
MPA ASW tracking			18%			
Missile testing (A-S)			41%			
BOMBEX (A-S)						40%
Explosive torpedo testing			13%			

The major impulsive Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the marine environment and activity frequency is greater than most other activities in this area. However, as the species' density is greater closer to shore, activities that tend to be undertaken closer to shore also tend to expose Hawaiian monk seals more than activities that tend to be more offshore or wide-ranging in nature.

6.6.1.12.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate 845 instances of Hawaiian monk seals exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment during training exercises and 179 during testing activities. In addition, there would be another 443 instances in which Hawaiian monk seals could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and 168 during testing activities. There would be five instances in which Hawaiian monk seals could accumulate energy sufficient to result in permanent shifts in hearing sensitivity during training exercises and 14 Hawaiian monk seals during testing activities.

The processed NAEMO results for impulsive acoustic sources generate one instance of a Hawaiian monk seal exposed to received levels that cause it to respond with behaviors that NMFS would classify as harassment during training exercises and zero during testing activities. In addition, there will be one instance in which Hawaiian monk seals could accumulate energy sufficient to result in temporary shifts in hearing sensitivity during training exercises and zero during testing activities. No Hawaiian monk seals would experience permanent shifts in hearing sensitivity, experience GI tract or lung injury, or would be killed from either training or testing activities.

6.6.1.12.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of Hawaiian monk seals that could experience behavioral responses due to non-impulsive acoustic sources during training exercises was 845 and 178 during testing activities. The estimated number of Hawaiian monk seals that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was 446 during training exercises and 180 during testing activities. Zero Hawaiian monk seals would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of Hawaiian monk seals that could experience behavioral responses due to impulsive acoustic sources during training exercises remained at zero, one Hawaiian monk seal would be expected to experience TTS. Zero Hawaiian monk seals would experience PTS, GI tract or lung injury, and zero Hawaiian monk seals would be killed from either training exercises or testing activities.

6.6.1.12.4 Response of Hawaiian Monk Seals to Acoustic Stressors

Hawaiian monk seals seem to have typical phocid hearing and can probably hear effectively up to 30 kHz and possibly as high as 60 kHz. The TTS thresholds used in the Navy analysis on harbor seals and elephant seals, based on the minimum statistically significant shifts determined by Finneran et al. (2011) and others, are 6 dB of shift which generally lasts less than 5 to 10 minutes, often much less. While we cannot determine whether the modeled exposures were just above threshold or a higher exposure that might produce a 20 to 30 dB TTS lasting for many minutes or hours, we can say with confidence that the frequency bandwidth of the effect will cover about 1/3 octave centered about 1/3 octave above the inducing sound;that is the effect from exposure to 3 kHz will be centered at 4.5 kHz across 3 to 6 kHz. Schusterman et al. (2000) reviewed available evidence on the potential for pinnipeds to echolocate and indicated that pinnipeds have not developed specialized sound production or reception systems required for echolocation. Instead, it appears pinnipeds have developed alternative sensory systems (e.g., visual, tactile) to effectively forage, navigate and avoid predators underwater. Since pinnipeds do not rely solely on acoustic cues for underwater feeding and navigation, it is anticipated TTS will not result in fitness consequences to individuals or the populations to which they belong.

Behavioral responses can range from a mild orienting response, or a shifting of attention, to flight and panic. Research and observations show that pinnipeds in the water are tolerant of anthropogenic noise and activity. 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 exposure. To result in significant fitness consequences we would have to assume that an individual animal 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. 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 et al 1993, as cited in New et al. 2014). Similarly, since Hawaiian monk seals have near continuous opportunities to feed throughout the year, they are unlikely to be affected by short-term variations in prey availability. Therefore, behavioral responses of Hawaiian monk seals to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population.

In 2010 and 2011, 13 monk seals were tagged in the Hawaii Range Complex for a total of 38,232 hours (1593 days) (Wilson and Littnan 2011). D'Amico and Wilson (2013) overlaid tag data with Navy sonar data to determine if seals were near active sonar and if so, what estimated received levels of sonar they were exposed to. By using geo-spatial databases, it was determined that 8 of these 13 seals were within 36 km of a hull mounted sonar ship for a total of 1473 hours 50 minutes (approximately 62 days) or 3.85 percent of the total tagged time. Considering concurrent active sonar activity during the 2 year period, 4 of the 8 seals were exposed to a total of 14.48 hours (less than 1 day) or 1.0 percent of concurrent sonar activity while the seal was within 36 km of a mid-frequency sonar ship. One seal was exposed to mid-frequency sonar transmissions for 30 minutes within the 5 to 15 km range bin, receiving a maximum receive level of 160 dB. The majority of the time the 4 seals were exposed to mid-frequency sonar transmissions while in the 15 to 36 km range bin, receiving a maximum receive level of 150 dB. The authors concluded there were no significant impacts from mid-frequency active sonar on the Hawaiian monk seals tagged in the Hawaii Range Complex during the 2010 to 2011 time period (D'Amico and Wilson 2013).

We assume that, like the whales discussed previously, Hawaiian monk seals are likely to try to avoid being exposed to vessel traffic, active sonar, and sound-producing activities such as gunnery exercises or sinking exercises. We do not have the information necessary to determine which of the many sounds associated with an activity is likely to trigger avoidance behavior in Hawaiian monk seals (for example, engine noise, helicopter rotors, ordnance discharges, explosions, or some combination of these), but these animals are likely to avoid the general area in which an exercise would occur by remaining close to a shoreline or on a beach. This avoidance will not prevent Hawaiian monk seals from being exposed to received levels of active sonar or explosions, but it would prevent Hawaiian monk seals from being exposed at received levels that would injure Hawaiian monk seals, cause them physiological distress, or alter their reproductive success. The beaches and shallow water areas within the Pacific Missile Range Facility at Kauai (in the main Hawaiian Islands) continue to be an important haul-out and nursing area for Hawaiian monk seals. While there has been a decline in the population of Hawaiian monk seals in the northwestern Hawaiian Islands, in the main Hawaiian Islands, where the Navy trains and tests, the numbers have continued to increase (Littnan 2011). The response estimates for Hawaiian monk seals represent an increase 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 fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA Rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of Hawaiian monk seals to reduce the viability or recovery of this population. We anticipate 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 this population.

6.6.1.13 *Density Estimates for Sea Turtles in Hawaii and SOCAL*

Species density data for green turtles was available for San Diego Bay from the scientific literature. Eguchi et al. (2010) discuss a mark recapture study of green turtles in San Diego Bay. The highest abundance they report is 61 turtles during 2002 and 2003. We used this as a conservative ceiling of the number of turtles present in San Diego Bay. By taking that abundance estimate and dividing by the area of the entire San Diego Bay in summer and the area of the south part of the Bay (where the turtles limit their movements) in winter, we derived warm season and cool season density estimates (Table 63).

The Navy has been funding visual surveys from the coast of Southern California out to approximately 100 nm from 2008 through 2013. As summarized through 2012, over 870 hours of effort was made and 43,500 nm of survey effort completed. To date, not a single sea turtle visual sighting has been made. These surveys, one of which was done exclusively by NMFS SWFSC, are able to sight small surfaced and submerged animals such as sharks, ocean sunfish, and fish schools, and have been demonstrated to sight sea turtles in other areas where turtles are more common (Eguchi and Seminoff 2011). Despite these survey results, based on anecdotal information, species life history, and limited stranding data, we assume these species do occur in waters outside San Diego Bay in SOCAL. However, presumably these species occur in low abundance. The only at-sea sighting that we are aware of for marine turtle species in SOCAL, besides green turtles, occurred in Janurary 2015 during a NOAA Fisheries marine mammal survey about 200 to 250 miles off Southern California. Within several days, researchers observed more than 70 confirmed or likely young loggerhead turtles (NMFS 2015). This sighting could be indicative of regular presence of loggerheads at this life stage in this area during certain times of the year.

Because of the lack of density estimates for other sea turtle species within the Action Area (excluding green sea turtles in San Diego Bay), sea turtle species were combined into one generic sea turtle group (Pacific sea turtles) for modeling around Hawaii and our qualitative analysis of SOCAL exposures to acoustic stressors. In other words, green, hawksbill, loggerhead, leatherback, and olive ridley sea turtles were all included as a group to account for occurrences of sea turtle species in all life stages. Density estimates (

Table 64) around Hawaii are derived from the Pacific Navy Marine Species Density Database.

Table 63. Summary of Density Values for Green Sea Turtles in the HSTT Action Area (Source: NMSDD)

Location	Spring	Summer	Fall	Winter
Hawaii Range Complex	G	G	G	G
W. Transit Corridor	N/A	N/A	N/A	N/A
E. Transit Corridor	N/A	N/A	N/A	N/A
SOCAL	N/A	N/A	N/A	N/A
San Diego Bay entire	N/A	0.3779	0.3779	N/A
San Diego Bay south	1.618293	N/A	N/A	1.618293
Inner Baja Extension	N/A	N/A	N/A	N/A
Outer Baja Extension	N/A	N/A	N/A	N/A

The units for numerical values are animals/km^2. 0 = species is not expected to be present; G = Green sea turtles are treated under the sea turtle group in Hawaii Range Complex; N/A = sufficient data is not available or location is not applicable; S = spatial model with various density values throughout the range.

Seasonal distribution of green sea turtles is illustrated in Figure 15 and Figure 16 below.

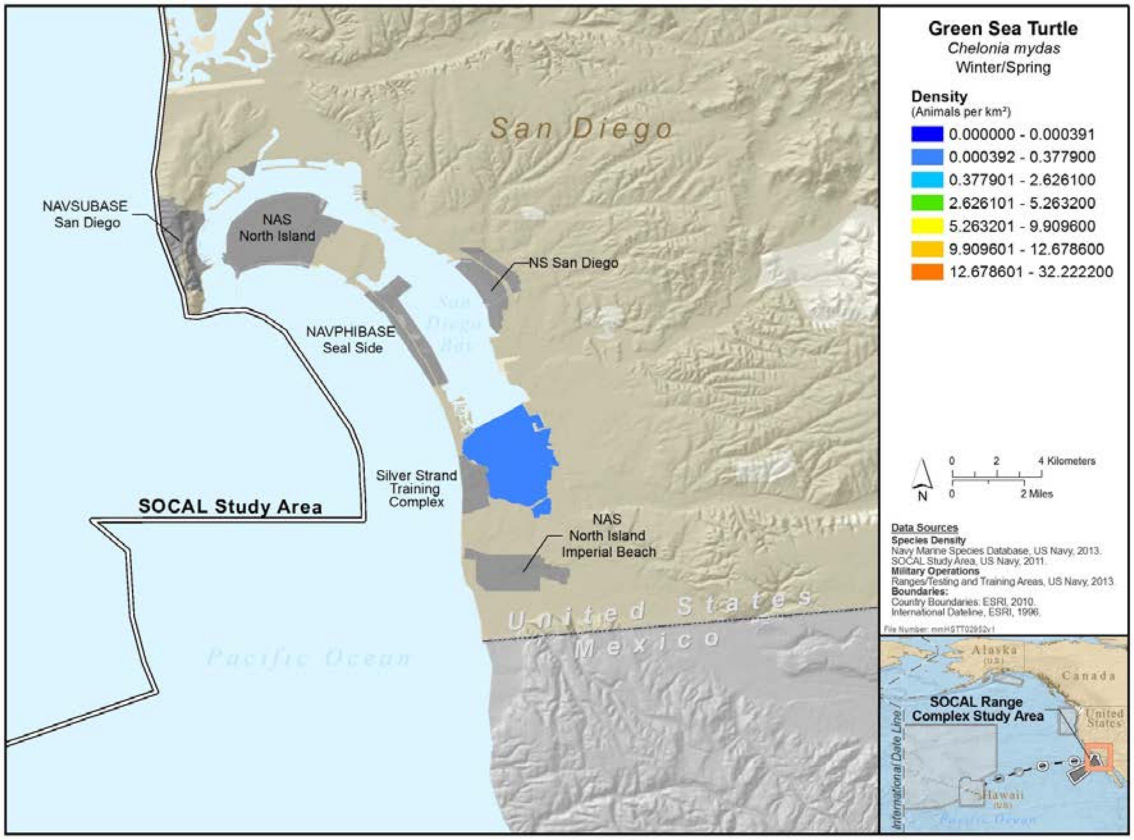

Figure 15. Winter/Spring Distribution of Green Sea Turtles in San Diego Bay (Source: Pacific NMSDD)

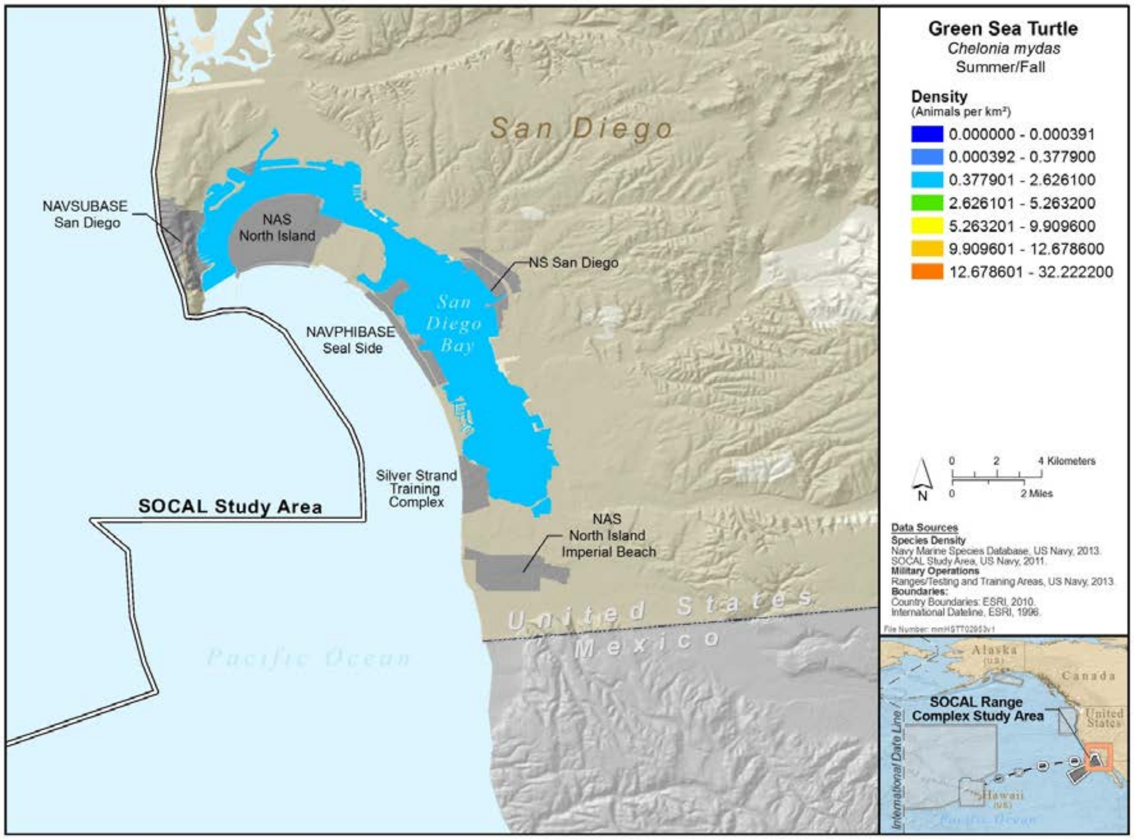

Figure 16. Summer/Fall Distribution of Green Sea Turtles in San Diego Bay (Source: Pacific NMSDD)

Table 64. Summary of Density Values for Sea Turtles in the Hawaii Range Complex

Location	Year-Round
Kauai	0.2786
Lanai	0.4491
Molokai	0.1624
Oahu	1.1252
Other Islands	0.4288
Outside of 100 m isobaths	0.00429
Pearl Harbor	S

The units for numerical values are animals/km2. 0 = species is not expected to be present; S = spatial model with various density values throughout the range.

Pacific sea turtle distribution in the Hawaii Range Complex is illustrated below in Figure 17.

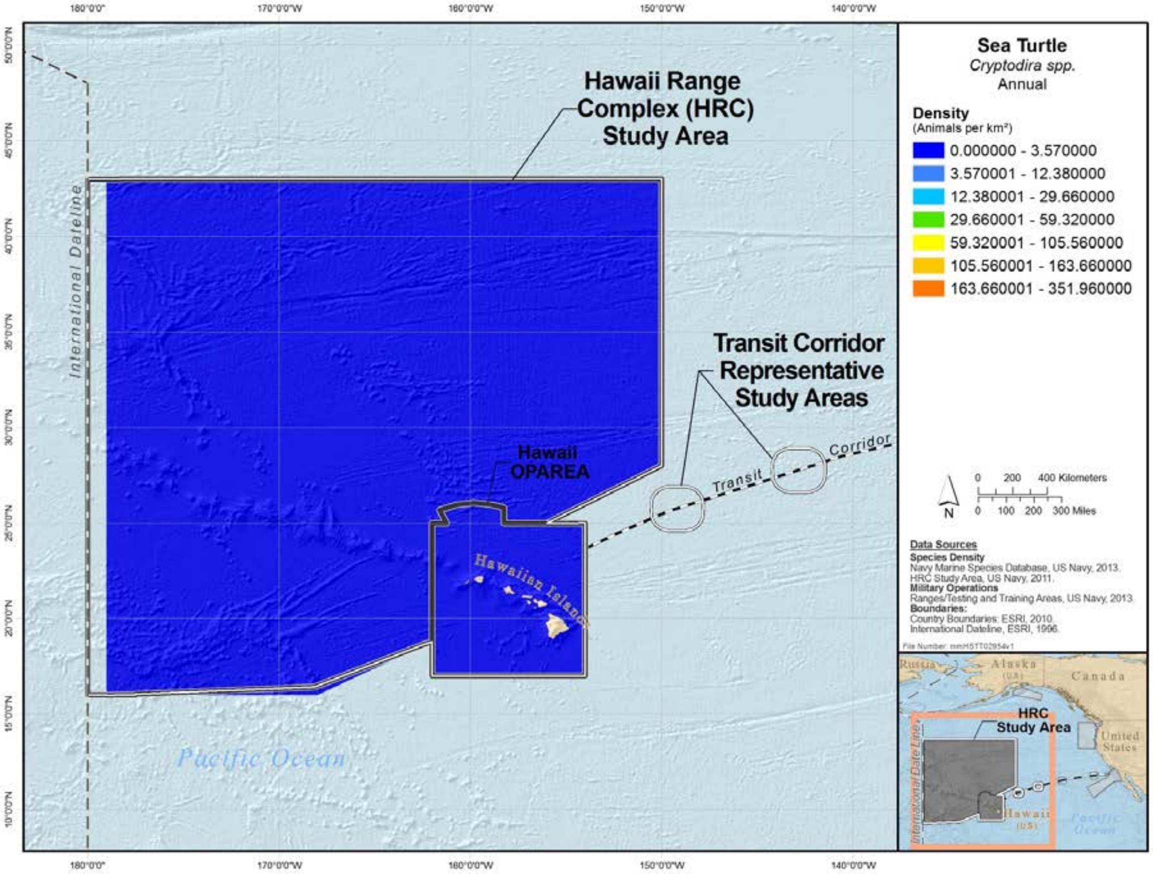

Figure 17. Annual Distribution of Pacific Sea Turtles in the Hawaii Range Complex (Source: Pacific NMSDD)

The distribution of Pacific sea turtles in and around Pearl Harbor is illustrated in Figure 18 and Figure 19 below.

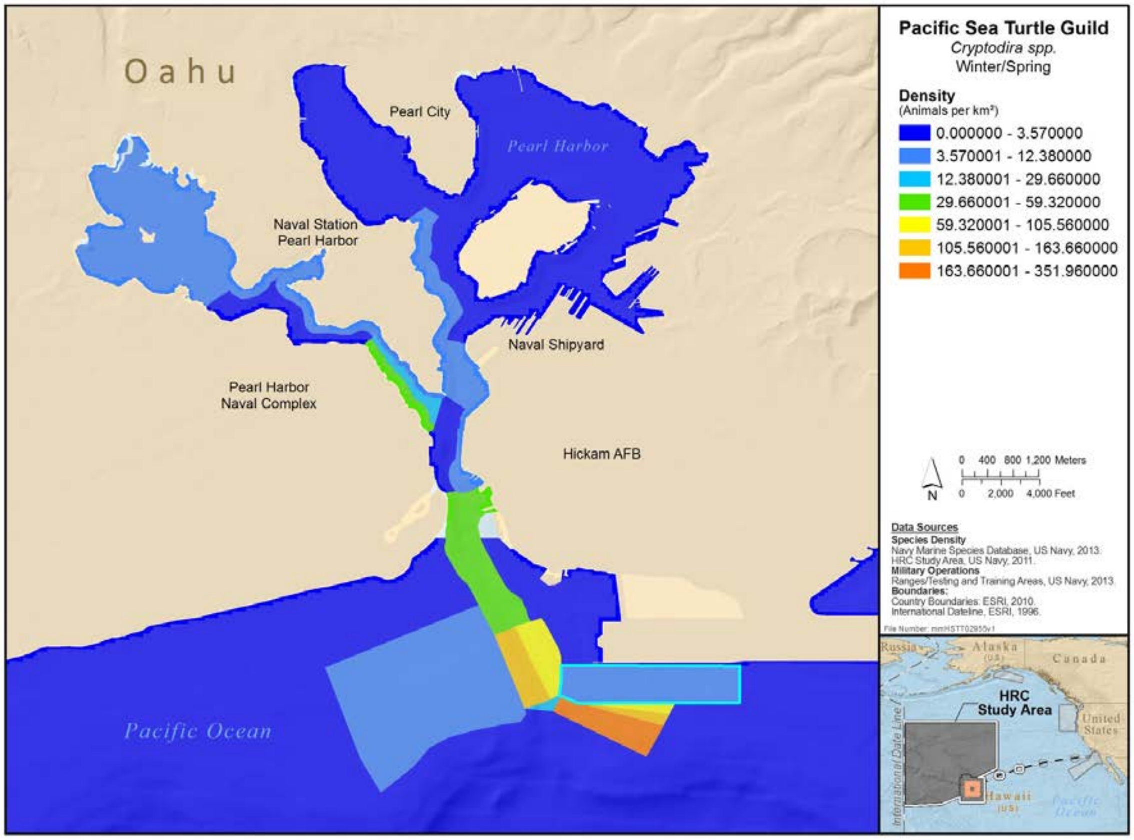

Figure 18. Winter/Spring Distribution of Pacific Sea Turtles In and Around Pearl Harbor, Island of Oahu, Hawaii (Source: Pacific NMSDD)

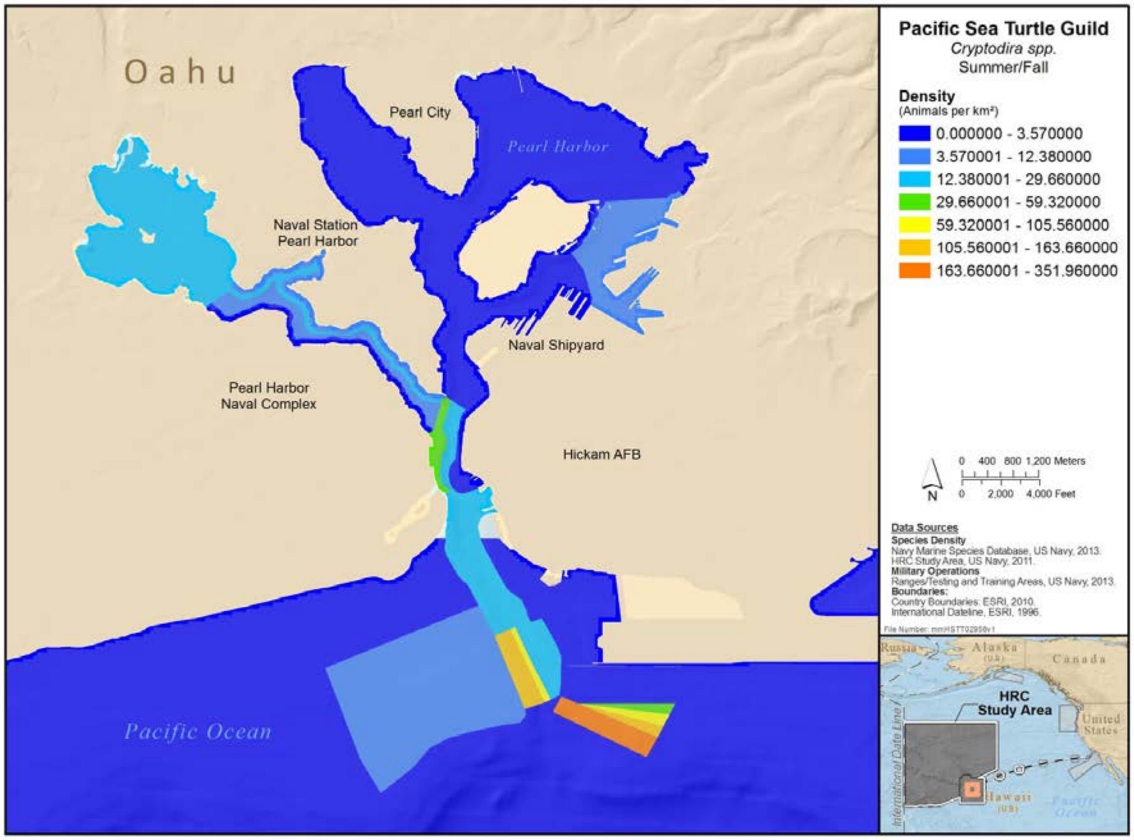

Figure 19. Summer/Fall Distribution of Pacific Sea Turtles In and Around Pearl Harbor, Island of Oahu, Hawaii (Source: Pacific NMSDD)

6.6.1.14 *Green Turtle – SOCAL Range Complex*

As described previously for other species, we provide three exposure estimates for green turtles: an unprocessed estimate, processed estimate, and an applied mitigation estimate.

6.6.1.14.1 Unprocessed Exposure Estimate

The model output estimates that green turtles will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 2,639 green turtle exposure events annually to non-impulsive sounds at levels above 120 dB. We expect these exposures to come from activities as layed out in Table 65. All modeled exposures will occur in Southern California, specifically in San Diego Bay.

Table 65. Activities that result in the highest percentages of green turtle unprocessed exposures to non-impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Green Turtle						
TRACKEX/TORPEX-surface				79%		
Extended echo ranging sonobuoy				21%		
Surface ship sonar testing and maintenance	4%					
Fixed AT/FP mine countermeasures	37%					
AUV AT/FP mine countermeasures	51%					

The major SOCAL Range Complex contributors are activities that regularly discharge non-impulsive acoustic energy into the marine environment where activity frequency is greater than most other activities in this area. In this case, only certain activities occur in San Diego Bay and these are the only activities that can expose green turtles to acoustic sources.

The model estimates that green turtles will be exposed to explosions and other impulsive acoustic stressors such as airguns associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 136 green turtle exposure events annually at sound above 120 dB SPL from impulsive acoustic sources. We expect these exposures to come from activities as layed out in Table 66. All modelled exposures are predicted to occur in Southern California, specifically San Diego Bay.

Table 66. Activities that result in the highest percentages of green turtle unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Green Turtles						
Pierside integrated swimmer defense	100%					
BOMBEX (A-S)				100%		

The major SOCAL Range Complex contributors are activities that regularly discharge impulsive acoustic energy into the marine environment and where activity frequency is greater than most other activities in this area. In this case, only certain activities occur in San Diego Bay and these are the only activities that can expose green turtles to the proposed activities.

6.6.1.14.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate zero instances of accumulated energy sufficient to result in temporary shift in hearing sensitivity from training exercises, and zero instances of accumulation of energy sufficient to result in permanent shift in hearing sensitivity from training exercises. There will be 529 instances in which green turtles could accumulate energy sufficient to result in temporary shifts in hearing sensitivity from testing activities, and 28 green turtles that could accumulate energy sufficient to result in permanent shift in hearing sensitivity from testing activities. It is important to note that auditory effects (PTS and TTS) to sea turtles were over-estimated in the Navy's modeling effort due to a mathematical error in the sea turtle thresholds. Since the exact error is known, the Navy therefore elected to not re-run the model. The TTS and PTS criteria for sea turtles is 17dB below what it is supposed to be; consequently, the estimates provided are overestimates. As a result, a substantial portion of these modeled TTS and PTS instances would not be expected to occur or modeled PTS might actually result in TTS or lower-level responses.

The processed NAEMO results for impulsive acoustic sources generated zero instances of green turtles exposed to received levels that cause them to respond with behaviors that NMFS would classify as harassment from either training or testing activities. In addition, there will be zero instances in which green sea turtles could accumulate energy sufficient to result in temporary shifts in hearing sensitivity, or could accumulate energy sufficient to result in permanent shift in hearing sensitivity from either training exercises testing activities. No green turtles would experience GI tract or lung injury, and no green turtles would be killed from either training or testing activities.

6.6.1.14.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy, the estimated number of green turtles that could experience TTS due to non-impulsive acoustic sources during training exercises remained at zero. Zero green turtles would be expected to experience PTS, or be killed from either training exercises or testing activities. Modeled unprocessed exposure estimates indicate zero green sea turtles could experience behavioral responses to non-impulsive acoustic sources during training exercises.

After considering mitigation actions that will be implemented by the Navy, the estimated number of green turtles that could experience TTS due to impulsive acoustic sources during training exercises remained at zero, the number of animals that would be expected to experience PTS is zero, and zero green turtles would experience GI tract or lung injury, and no green turtles would be killed. Modeled unprocessed exposure estimates indicate zero green sea turtles could experience behavioral responses to impulsive acoustic sources during training exercises.

After considering mitigation actions that will be implemented by the Navy plus those that are mandatory under the MMPA rule, the estimated number of green turtles that could experience TTS due to non-impulsive acoustic sources during testing activities is 555, the number of animals that would be expected to experience PTS is 1. Zero green turtles would be killed. Modeled unprocessed exposure estimates indicate zero green sea turtles could experience behavioral responses to non-impulsive acoustic sources during testing exercises.

The estimated number of green turtles that could experience TTS due to impulsive acoustic sources during testing activities is zero. Zero green turtles would be expected to experience PTS, GI tract or lung injury, and no green turtles would be killed. Modeled unprocessed exposure estimates indicate zero green sea turtles could experience behavioral responses to impulsive acoustic sources during testing exercises.

6.6.1.15 *Pacific Sea Turtles – Hawaii Range Complex*

As described previously, we provide three exposure estimates for Pacific sea turtles in the Hawaii Range Complex: the unprocessed estimate, processed estimate, and an applied mitigation estimate. All of these exposures were modeled to occur in the areas around Hawaii.

6.6.1.15.1 Unprocessed Exposure Estimate

The model output estimates that Pacific sea turtles will be exposed to sonar and other non-impulsive acoustic stressors associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 25,085 Pacific sea turtle exposure events annually to non-impulsive sounds at levels above 120 dB SPL. Table 67 lists the activities that result in the highest percentages of Pacific sea turtle unprocessed exposures to non-impulsive acoustic sources in the HSTT Action Area.

Table 67. Activities that result in the highest percentages of Pacific sea turtle unprocessed exposures to non-impulsive acoustic sources in the HSTT Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Pacific Sea Turtles						
ASW mission package testing			23%			
MPA ASW tracking			40%			
Unmanned vehicle development			26%			
Undersea warfare exercise						37%
Extended echo ranging sonobuoy						23%
Rim of the Pacific exercise						31%

The major Hawaii Range Complex contributors are activities that regularly discharge acoustic energy into the nearshore marine environment and where activity frequency is greater than most other activities in this area. In this case, only certain activities occur in areas in and near Pearl Harbor (where Pacific sea turtle density is relatively high versus very low density estimates in all other Hawaii Range Complex locations) and these are the only activities that can expose Pacific sea turtles to acoustic stressors. These exposures stem from RIMPAC and undersea warfare exercises. One exposure may result from testing activities due to ASW mission package testing.

The model estimates that Pacific sea turtles will be exposed to explosions and other impulsive acoustic stressors such as airguns associated with training exercises and testing activities throughout the year. The NAEMO provided an unprocessed estimate of 22,049 Pacific sea turtle exposure events annually at sound levels above 120 dB SPL from impulsive acoustic sources. All of these exposures are predicted to occur in the areas around Hawaii. Table 68 lists activities that result in the highest percentages of Pacific sea turtle unprocessed exposures to impulsive acoustic sources.

Table 68. Activities that result in the highest percentages of Pacific sea turtle unprocessed exposures to impulsive acoustic sources in the Hawaii-SOCAL Training and Testing Action Area.

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Pacific Sea Turtles						
Missile exercise (A-S)						10%
MPA ASW tracking			28%			
Explosive torpedo testing			16%			

Species-largest contributors to exposure	Testing			Training		
	SOCAL Range Complex	Transit Lane	Hawaii Range Complex	SOCAL Range Complex	Transit Lane	Hawaii Range Complex
Missile testing (A-S)			27%			
BOMBEX (A-S)						18%
Mine neutralization-EOD						60%

6.6.1.15.2 Processed Exposure Estimate

The processed NAEMO results for non-impulsive acoustic sources generate 261 instances in which Pacific sea turtles could accumulate energy sufficient to result in temporary shift in hearing sensitivity from training exercises and 77 Pacific sea turtles from testing activities. Four Pacific sea turtles could accumulate energy sufficient to result in permanent shift in hearing sensitivity from training exercises and no animals from testing activities. Again, since TTS and PTS criteria for sea turtles is 17 dB below what it is supposed to be, the estimates provided are overestimates.

The processed NAEMO results for impulsive acoustic source generated 166 instances in which Pacific sea turtles could accumulate energy sufficient to result in temporary shifts in hearing sensitivity from training exercises and one from testing activities. There could be 44 Pacific sea turtles that could accumulate energy sufficient to result in permanent shift in hearing sensitivity from training exercises and five from testing activities. Again, since TTS and PTS criteria for sea turtles is 17dB below what it is supposed to be, the estimates provided are overestimates. No Pacific sea turtles would experience GI tract injuries from either training or testing activities. There could be 13 slight lung injuries from training exercises and one from testing activities. Five Pacific sea turtles could be killed by training exercises and none killed by testing activities.

6.6.1.15.3 Applied Mitigation

After considering mitigation actions that will be implemented by the Navy, the estimated number of Pacific sea turtles that could experience TTS due to non-impulsive acoustic sources during training exercises is 145. Zero Pacific sea turtles would be expected to experience PTS or be killed. Modeled unprocessed exposure estimates indicate 12 Pacific sea turtles could experience behavioral responses to non-impulsive acoustic sources during training exercises.

After considering mitigation actions that will be implemented by the Navy, the estimated number of Pacific sea turtles that could experience TTS due to impulsive acoustic sources during training exercises is 182, the number of animals that would be expected to experience PTS is 21. Zero Pacific sea turtles would experience GI tract injuries, 13 would experience lung injury, and four Pacific sea turtles would be killed. Modeled unprocessed exposure estimates indicate 7,132 Pacific sea turtles could experience behavioral responses to impulsive acoustic sources during training exercises.

After considering mitigation actions that will be implemented by the Navy, the estimated number of Pacific sea turtles that could experience TTS due to non-impulsive acoustic sources during testing activities is 77, and the number of animals that would be expected to experience PTS is zero. Zero Pacific sea turtles would be killed. Modeled unprocessed exposure estimates indicate 3 Pacific sea turtles could experience behavioral responses to non-impulsive acoustic sources during testing exercises.

The estimated number of Pacific sea turtles that could experience TTS due to impulsive acoustic sources during testing activities is one. Five Pacific sea turtles would be expected to experience PTS, zero turtles would experience GI tract or lung injury, and no Pacific sea turtles would be killed. Modeled unprocessed exposure estimates indicate 126 Pacific sea turtles could experience behavioral responses to impulsive acoustic sources during testing exercises.

6.6.1.16 *Response of Sea Turtles to Acoustic Stressors*

With the exception of green turtles in San Diego Bay, the information available did not allow us to break down the modeled estimates described above by species and estimate the probability of each sea turtle species experiencing behavioral responses, TTS, and PTS from the use of active sonar and explosive ordinance in the HSTT Action Area. For modeled acoustic exposures that were expected to result in death (modeled mortality of four Pacific sea turtles per year), we assume that the majority of these deaths were green sea turtles since 97 percent of the turtles that strand in the Hawaiian Islands are green turtles (Balazs and Chaloupka 2006a). Based on available stranding data, the remaining three percent are equally likely to be hawksbill or olive ridley sea turtles. Stranding data suggests the likelihood of leatherback or loggerhead sea turtle mortality is exceptionally low relative to that of green, hawksbill, and olive ridley sea turtles (Balazs and Chaloupka 2006a).

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Ketten 2006; 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, b) 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. 2012b; 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 HSTT 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 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. Therefore, behavioral responses of sea turtles to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population.

Although we assume that sea turtles in the vicinity of an in-water detonation might experience a TTS or PTS, it is not certain at what energy levels and received levels are necessary to induce threshold shifts. The few studies completed on the auditory capabilities of sea turtles (adult green, loggerhead, and Kemp's ridley turtles) suggest that they could be capable of hearing low frequency sounds (Lenhardt 1994b; Moein et al. 1993; Ridgway et al. 1969). Ridgway et al. (Ridgway et al. 1969) reported maximal sensitivity for green turtles occurred at 300 to 400 Hz, with a rapid decline in sensitivity for lower and higher tones. Similarly, Moein et al. (Moein et al. 1993) reported a hearing range of about 250 to 1,000 Hz for loggerhead turtles, and Lenhardt (Lenhardt 1994b) stated that maximal sensitivity in sea turtles generally occurs in the range from 100 to 800 Hz. Calculated in-water hearing thresholds within the useful range appear to be high (e.g., about 160 to 200 dB re 1 μPa) (Lenhardt 1994b). Piniak et al. reported maximum sensitivity between 100 and 400 Hz in water (84 dB re: 1 μPa at 300 Hz) and 50 and 400 Hz in air (62 dB re: 20 μPa at 300 Hz) for leatherback sea turtle hatchlings (Piniak et al. 2012a). In the absence of more specific information that could be used to determine the acoustic harassment range for sea turtles, the U.S. Navy assumed that frequencies >100 Hz (which are the acoustical harassment ranges predicted for odontocetes) would be conservative for sea turtles. Given that sea turtles do not rely on acoustic cues for most important life functions, it is anticipated that TTS and PTS will not result in fitness consequences to individuals or the populations to which they belong.

Acoustic stressors associated with the Navy’s activities in the HSTT Action Area have the ability to cause slight lung injury. Slight lung injury is defined as having a zero percent mortality rate and being completely recoverable. Although slight lung injuries could temporarily affect the fitness of affected individuals by reducing their respiration rate, these effects are expected to stop once the injury has healed. Because sea turtles are long-lived animals, a temporary disruption of behaviors or fitness levels resulting from slight lung injury is not expected to substantially impair an individual turtle when considering its overall lifetime fitness. Because we do not expect fitness consequences to any individuals, we do not expect population-level consequences.

Acoustic stressors from Navy testing activities conducted during the remaining period of the five-year MMPA rule and into the reasonably foreseeable future are likely to cause TTS and PTS in a majority of the known resident green sea turtles in San Diego Bay. However, there is no evidence that TTS or PTS results in energetic effects to individual sea turtles or would be likely to significantly reduce the viability of the population these individuals represent. We do not anticipate any mortality of green turtles from acoustic stressors in San Diego Bay.

Acoustic stressors from Navy training exercises and testing activities conducted during the remaining period of the five-year MMPA rule and into the reasonably foreseeable future are not likely to significantly interact (exposure and response) with a sufficient number of green, hawksbill, leatherback, loggerhead, or olive ridley sea turtles to reduce the viability of those populations. While we anticipate small numbers of injury and mortality (Table 69) from acoustic stressors in the Hawaii Range Complex, we do not anticipate those responses to result in substantial changes in reproduction, numbers, or distribution of those populations.

Table 69. Summary of Effects to Sea Turtles from Acoustic Stressors

Training Non-Impulsive Sources						
Species	**Non-TTS**	**TTS**	**PTS**	**Slight Lung Injury**	**GI Tract Injury**	**Mortality**
Green	0	0	0	0	0	0
Pacific	12	145	0	0	0	0
Training Impulsive Sources						
Green	0	0	0	0	0	0
Pacific	7,132	182	21	13	0	4
Testing Non-Impulsive Sources						
Green	0	555	1	0	0	0
Pacific	3	77	0	0	0	0
Testing Impulsive Sources						
Green	0	0	0	0	0	0
Pacific	126	1	5	0	0	0

6.6.2 Vessel Strike

As discussed in the *Consultation History*, NMFS expanded the scope of the reinitiated consultation to re-assess effects to marine mammals and sea turtles including effects from vessel strike. The December 2013 and corrected April 2014 biological opinions both analyzed vessel strike in the same way it was analyzed by the Navy and proposed in the final draft MMPA rule and draft LOAs. This approach applied risk of strike equally among all ESA-listed whale species without consideration of species-specific abundances in the Action Area. In addition, the number of incidental takes from vessel strikes requested in the Navy's MMPA application was based upon a worst-case scenario that far exceeded both annual and five-year historic strike rates. As a result, the previous analysis may have greatly over-estimated risk for certain species. In this Opinion, we calculate a historic strike rate for the period 2005 to 2014, which we believe most accurately reflects current strike risk, and apportion risk to each ESA-listed species based on each species' relative abundance.

Navy vessels are likely to detect any vessel strikes involving the listed whale species evaluated in this Opinion, and Navy policy (Chief of Naval Operations Instruction [OPNAVINST] 3100.6H) requires Navy vessels to report all whale strikes. That information is collected by the Office of the Chief of Naval Operations Energy and Environmental Readiness Division and provided to NMFS within 72 hours of the incident and on an annual basis. It is unlikely that a whale vessel strike would go undetected. As stated in the HSTT final LOA application (page 185-186):

The ability of a 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. Key points in discussion of Navy vessels in relationship to potential ship strike include:

- Many Navy ships have their bridges positioned closer to the bow, offering good visibility ahead of the ship.

- There are often aircraft associated with the training or testing activity, which can detect marine mammals in the vicinity or ahead of a vessel’s present course.

- Navy ships are generally much more maneuverable than commercial merchant vessels if marine mammals are spotted and the need to change direction necessary. Navy ships operate at the slowest speed possible consistent with either transit needs, or training or testing need. 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 also added as a mitigation measure in previous MMPA permits is for Navy vessels to maneuver to keep at least 500 yds. away from any observed whale in the vessel's path and avoid approaching whales head-on, so long as safety of navigation is not imperiled.

- In many cases, Navy ships will likely move randomly or with a specific pattern within a sub-area of the HSTT for a period of time from one day to two weeks as compared to straight line point-to-point commercial shipping.

- Navy overall crew size is much larger than merchant ships allowing for more potential observers 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 marine mammals. Additional lookouts, beyond already stationed bridge watch and navigation teams, are stationed during some training events.

- Navy lookouts receive extensive training including Marine Species Awareness Training designed to provide marine species detection cues and information necessary to detect marine mammals.

This assessment is further supported by Large Whale Ship Strike Database (Jensen and Silber 2004), which states Federal ships carry substantial crew, a number of whom are generally on the bridge at any one time (bridge crew on Navy vessels often consists of a half dozen individuals or more). Such crews are more likely to spot a whale and/or register that a collision has occurred than a container ship or tanker with only one or two individuals at the helm.

6.6.2.1 *Overview of Navy Vessel Movement in the HSTT Action Area*

This section provides a general overview of vessel movement and locations in the HSTT Action Area, which illustrates the context of our vessel strike analysis. Specific data on how often ships get underway, where they intend to conduct their missions and other information regarding their schedules is classified for National security reasons. In order to provide NMFS with information that is supportive of their ESA analysis, this information was derived from unclassified sources, and in its aggregate remains unclassified.

The number of Navy vessels in the Action Area varies based on training and testing schedules. Most activities include one or two vessels, with an average of one vessel per activity, and last from a few hours up to two weeks. Multiple ships can be involved with major training events and can operate for extended periods beyond the horizon and out of visual sight from each other. In a given year, Navy vessel movement would likely be concentrated in portions (see Figure 20) of the SOCAL and Hawaii range complexes.

Vessels strikes may occur from surface operations and sub-surface operations (excluding bottom crawling, unmanned underwater vehicles). Vessels include ships, submarines and boats ranging from 16 ft (5 m) Rigid Hull Inflatable Boats (RHIB) to aircraft carriers (CVN) with lengths up to 1,092 ft (333 m). Navy ships greater than 60 ft (18 m) in length generally operate at 10 to 15 knots for fuel conservation when cruising. Submarines generally operate at 8 to 13 knots during transit and slower for certain tactical maneuvers. Craft less than 60 ft (18 m) in length have more variable speeds, dependent on the mission. While these speeds are representative, some vessels operate outside of these speeds due to unique training and testing or safety requirements for a given event. Examples include increased speeds needed for flight operations, full speed runs to test engineering equipment, and time critical positioning needs. Examples of decreased speeds of less than 5 knots or completely stopped include launching small boats, certain tactical maneuvers, and target launch or retrievals.

On average ships have a fuel allocation of approximately 21 steaming days per ship per quarter, and for purposes of this analysis, it can be assumed this equates to four underway periods per year. Also, based on maintenance and deployment requirements there could be approximately 50 ships per quarter involved in activities in the HSTT Action Areas. Almost all underway time will be farther than 12 nm from shore and a majority would likely be greater than 25 nm from shore. Transit in and out of ports to access operational areas or the transit lane makes up a small percentage of this overall time. During training activities in the transit lanes, ship speeds generally range from 14 to 18 knots; however, depending on the specific activity, these vessels would also operate at higher or lower speeds.

Mintz (2012) reviewed vessel movement in the HSTT Action Area, comparing Navy vessel traffic with commercial shipping. Navy surface ships accounted for 97,000 hours of accumulated at-sea time within subareas of the HSTT Action Area whereas commercial shipping accounted for 875,000 hours. Therefore, Navy shipping represented only 11% of all vessel movement within the HSTT Action Area.

Figure 20 displays locations within the HSTT Action Area where the Navy concentrates the majority of their effort. Data in Figure 20 is based on totaled ship-hours for each 15-minute geographical box (i.e., 0.25 degrees latitude by 0.25 degrees longitude) and each box shaded according to the total number of ship-hours it contained (i.e., its "density").

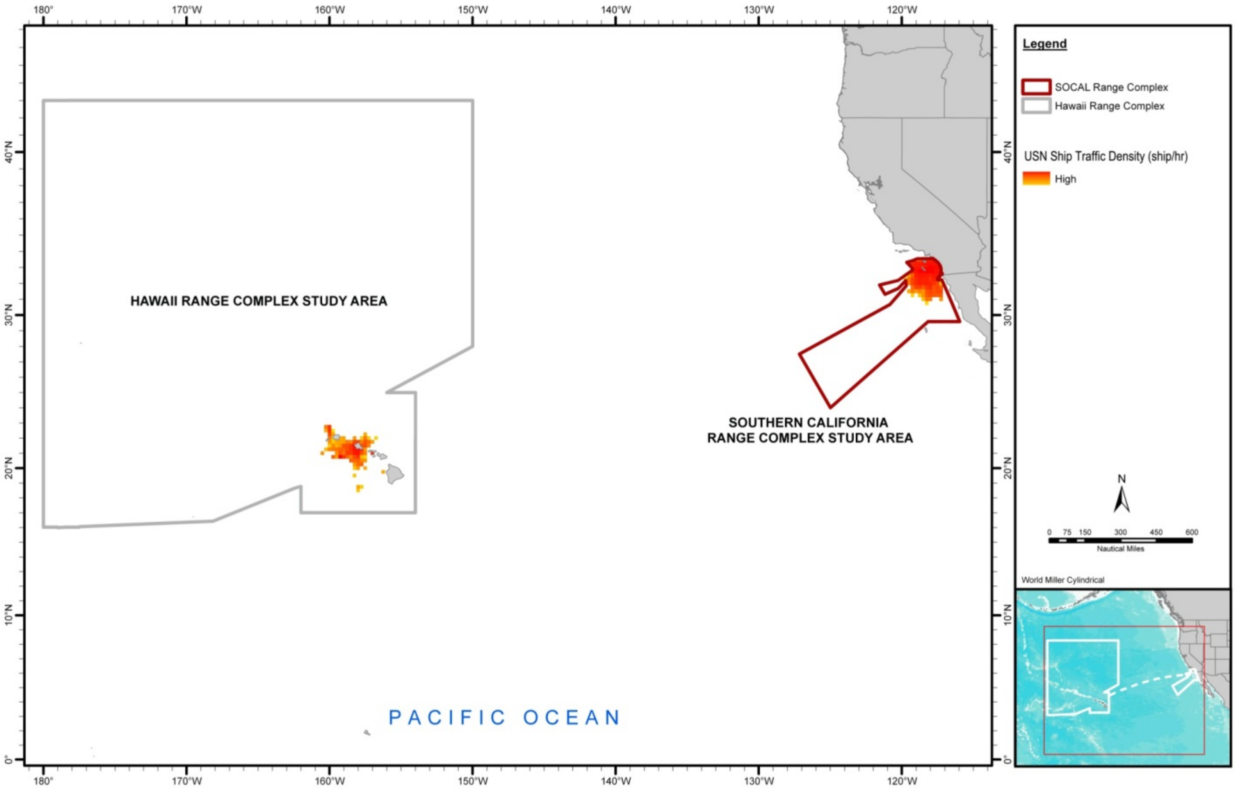

Figure 20. High density Navy Surface Ship transit and movement within the HSTT Action Area – (Mintz, 2012)

6.6.2.2 *Relative Abundance of ESA-listed species in the Action Area*

Density estimates used in the vessel strike analysis were derived from the Pacific Navy Marine Spatial Density Database (NMSDD). For more information on the NMSDD, see the *Approach to the Assessment* section of this Opinion.

The total range area (including the operations area minus land area within the HSTT Action Area) estimates used in the calculation are:

Hawaii Range Complex = 7,853,842.07 km²

SOCAL Range Complex = 409,213.01 km²

The NMSDD does not include any of the land areas (islands) in the total area used for the ranges. This means the total area in the NMSDD is slightly less than the total area of the range. The estimates are for the entire range complex, including the areas where the Navy trains less frequently. Also, the density data does not include the transit lane between SOCAL and Hawaii. Nevertheless, the NMSDD represents the best available data regarding species density and abundance within the HSTT, and we do not believe that the limitations of the data significantly affect our analysis and conclusions set forth below because the marine mammal densities are generally lower in the open ocean and transit lane areas of the Action Area.

From the NMSDD density data, we estimated relative abundance of each species (i.e., the abundance of that species relative to the total abundance of all whale species), both annually and by season, in each range complex. To calculate the relative abundance of each species, we divided the abundance of each individual species by the total number of whales of all species present in each range complex (e.g., 100 fin whales/1000 total whales; relative abundance of fin whales = 0.1). Table 70 and Table 71 below summarize the relative abundance of each species, both annually and by season. ESA-listed species are highlighted. We include both ESA-listed and non-ESA-listed whales in the calculation of relative abundance for each species because the historic strike rate is not exclusive to ESA-listed whales.

Table 70. Relative Abundance of Large Whales by Species and by Season in the Hawaii Range Complex (Source: Pacific NMSDD)

Taxonomic Name	Common Name	Hawaii Range Complex										
		Estimated Abundance				Total	Estimated Relative Abundance				Ave Abun-dance	Total Annual Relative Abun-dance
		Fall	Winter	Spring	Summer	Annual	Fall	Winter	Spring	Summer	Annual	Annual
Balaenoptera acutorostrata	Common or dwarf minke whale	1,665.04	4,681.03	3,131.51	1,647.97	11,126	0.108205	0.178931	0.120100	0.114296	2,781	0.135609
Balaenoptera borealis	Sei whale	1,068.59	900.35	1,122.38	256.86	3,348	0.069444	0.034416	0.043045	0.017815	837	**0.040811**
Balaenoptera edeni	Bryde's whale	979.01	979.01	979.01	979.01	3,916	0.063622	0.037422	0.037547	0.067900	979	0.047732
Balaenoptera musculus	Blue whale	416.67	416.67	416.67	79.58	1,330	0.027078	0.015927	0.015980	0.005519	332	**0.016206**
Balaenoptera physalus	Fin whale	556.84	556.84	556.84	556.84	2,227	0.036187	0.021285	0.021356	0.038620	557	**0.027149**
Eschrichtius robustus	Gray whale[1]											0
Megaptera novaeangliae	Humpback whale	401.32	9,659.87	9,659.87	401.32	20,122	0.026080	0.369246	0.370476	0.027834	5,031	**0.24527**
Physeter macrocephalus	Sperm whale	10,300.41	8,967.31	10,207.95	10,496.88	39,973	0.669385	0.342773	0.391496	0.728017	9,993	**0.487223**
	Total	15,388	26,161	26,074	14,418	82,042	1.0	1.0	1.0	1.0	20,510	1.0

[1] Gray whales including Eastern and Western North Pacific stocks are not likely to occur in the Hawaii Range Complex on a regular basis and thus relative abundance is too low to estimate.

Table 71. Relative Abundance of Large Whales by Species and by Season in the SOCAL Range Complex (Source: Pacific NMSDD)

Taxonomic Name	**Common Name**	**SOCAL Range Complex**										
		Estimated Abundance				**Total**	**Estimated Relative Abundance**				**Ave Abun-dance**	**Total Annual Relative Abun-dance**
		Fall	**Winter**	**Spring**	**Summer**	**Annual**	**Fall**	**Winter**	**Spring**	**Summer**	**Annual**	**Annual**
Balaenoptera acutorostrata	Common or dwarf minke whale	174.33	142.76	154.46	179.98	652	0.048388	0.020982	0.026166	0.048107	163	0.032494
Balaenoptera borealis	Sei whale	35.22	34.55	38.10	38.10	146	0.009776	0.005078	0.006454	0.010184	36	**0.00728**
Balaenoptera edeni	Bryde's whale	4.42	4.42	4.416	4.416	18	0.001226	0.000649	0.000748	0.001180	4	0.000881
Balaenoptera musculus	Blue whale	998.32	30.374	30.37	998.32	2,057	0.277095	0.004464	0.005145	0.266845	514	**0.102608**
Balaenoptera physalus	Fin whale	2,017.57	2,988.75	2,077.51	2,144.91	9,229	0.560002	0.439271	0.351943	0.573321	2,307	**0.460267**
Eschrichtius robustus	Gray whale[1]		2,499.27	2,499.27		4,999					2,499	0.249293
Megaptera novaeangliae	Hump-back whale	129.46	277.80	277.80	129.46	815	0.035932	0.040829	0.047060	0.034603	204	**0.040622**
Physeter macrocephalus	Sperm whale	243.48	825.97	821.05	246.024	2,137	0.067581	0.121397	0.139091	0.065761	534	**0.106555**
	Total	3,602.80	6,803.88	5,902.98	3,741.20	20,050.87	1.0	1.0	1.0	1.0	5,012.72	1.0

[1] Gray whale abundances, including Eastern and Western North Pacific stocks, in SOCAL could not be estimated for summer and fall seasons due to very rare occurrence of this migratory species during these times and a lack of sighting data.

Figure 21 illustrates the abundance of each large whale species in the Hawaii Range Complex. This provides an indication of the variation in strike risk by species throughout the year. Sei, sperm, fin, and blue whale abundances remain relatively constant throughout the year in Hawaii. Humpback whale abundance is highest during winter and spring and lower in summer and fall.

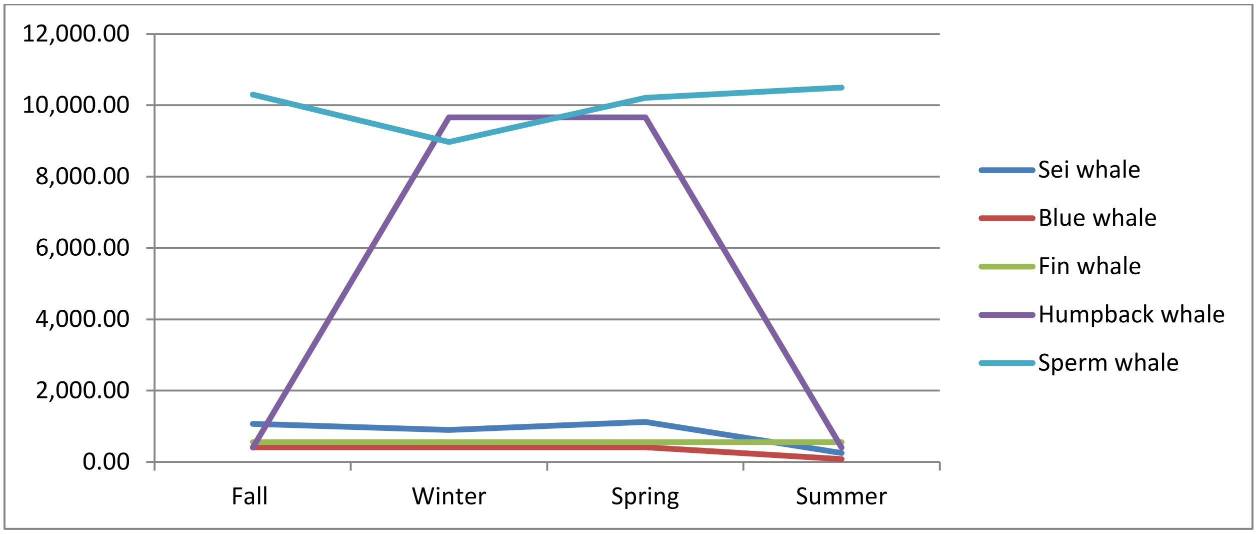

Figure 21. Abundance of ESA-listed whale species in the Hawaii Range Complex by season

Figure 22 illustrates the abundance of each ESA-listed large whale species in SOCAL. Sei and humpback whale abundances remain relatively constant throughout the year in SOCAL. Fin whale abundance is highest during winter, and lower throughout spring, summer, and fall. Sperm whale abundance is highest during winter and spring and lower in the fall and summer. Blue whale abundance is highest during fall and summer and lowest during winter and spring.

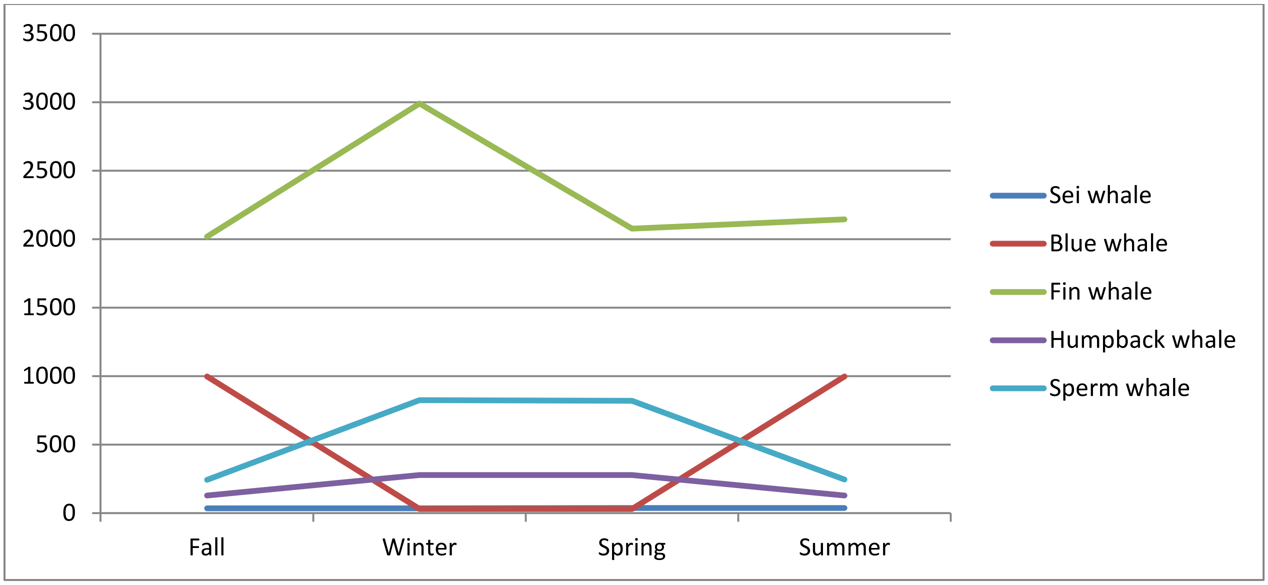

Figure 22. Abundance of each ESA-listed whale species in the SOCAL Range Complex by season

From the figures above, we conclude that the overall risk of ESA-listed species vessel strike is highest in the winter in both Hawaii and SOCAL range complexes, largely due to seasonal changes in humpback and fin whales abundances, respectively. However, risk to particular species may vary.

6.6.2.3 *Historic Whale Strike Rates during Training and Testing in Hawaii and SOCAL Range Complexes*

The Navy's vessel strike history of whales over the most recent 10-year period (Table 72) reflects enhanced standard operating procedures and minimization measures the Navy has implemented that are also required under the current MMPA rule and LOAs. In addition, the Navy has indicated that the action that is the subject of this analysis (i.e., the current level of activity) does not involve a change in the rate at which vessels are used as compared to the last decade and is not likely to change the probability of a vessel strike in any meaningful way [FEIS at 3.4-264]. Therefore, the average annual strike rate over the 10-year period 2005 through 2014 is used to predict vessel strike in future years. This is the best representation of strike risk during training and testing activities in the HSTT Action Area for the following reasons: 1) it encapsulates documented changes in the California Current Ecosystem since at least 2000 to 2001 that could mean differences in large whale distributions, 2) it recognizes the Navy's increasing awareness of marine mammals through its comprehensive mitigation and monitoring efforts over the past two five-year MMPA cycles, and 3) it reflects the risk associated with the most recent trends in Navy training and testing activities conducted in the HSTT Action Area.

Table 72. Number of Navy Ship Strikes in the HSTT Action Area by Year

	SOCAL		Hawaii	
Year	**Number of Strikes**	**Species**	**Number of Strikes**	**Species**
2005	0		0	
2006	**1**	**Unknown**	**0**	
2007	**0**		**1**	**Sperm Whale**
2008	**0**		**1**	**Unknown**
2009	**2**	**Fin Whales**	**0**	
2010	0		0	
2011	0		0	
2012	0		0	
2013	0		0	
2014	0		0	
Total	**3**		**2**	

From Navy strike history data (Table 72), we derive an overall annual strike rate for the HSTT Action Area of 0.5 whales/per year (Table 73). It is important to note no strikes have occurred in the last five years during training and testing in the HSTT Action Area. As discussed in the *Environmental Baseline*, in 2011, a Military Sealift Command vessel struck a whale while transiting between Guam and Oregon. This vessel was not associated with Pacific Fleet training or testing activities, and therefore is not included in the risk assessment for vessel strike from training and testing activities.

Table 73. Annual Strike Rates by Range Complex (2005 through 2014)

Range Complex	**Annual Strike Rate (whales struck per year)**
SOCAL Range Complex	0.3
Hawaii Range Complex	0.2
Total HSTT Area	**0.5**

6.6.2.4 *Estimation of Strike Risk*

To estimate strike risk for ESA-listed whales, we performed the following calculations:

Step One: Calculate the number of whales of each species struck per year by multiplying overall strike rate by the relative abundance

We apportioned the annual strike rate in Table 73 by species and season using information from Table 70 and Table 71 to obtain a species-specific strike rate (Table 74). For SOCAL, we multiplied the annual strike rate (0.3) times the relative abundance of each species to obtain a species-specific strike rate per year. We did the same for the Hawaii strike rate (0.2).

Table 74. Apportioned Annual Strike Rate Based on the 10-Year Average (2005 through 2014)

	Hawaii Range Complex		SOCAL Range Complex	
	Relative Annual Abundance	**Apportioned Annual Strike Rate**	**Relative Annual Abundance**	**Apportioned Annual Strike Rate**
Blue Whale[1]	**0.016206**	**0.003241**	**0.102608**	**0.030783**
Sei Whale[1]	**0.040811**	**0.008162**	**0.00728**	**0.002184**
Fin Whale[1]	**0.027149**	**0.005430**	**0.460267**	**0.138080**
Humpback Whale[1]	**0.24527**	**0.049054**	**0.040622**	**0.012187**
Gray Whale –ENP	0.00	0.00	0.249293	0.074697
Gray Whale –WNP[1, 2]	**0.00**	**0.00**	**0.000299**	**0.000090**
Sperm Whale[1]	**0.487223**	**0.097455**	**0.106555**	**0.031967**
Bryde's Whale	0.047732	0.009546	0.000881	0.000264
Minke Whale	0.135609	0.027122	0.032494	0.009748
Total (All Species)	1.0 (100%)	**0.2**	1.0 (100%)	**0.3**

[1] ESA-listed Species

[2] Relative abundance of the western North Pacific stock of gray whales is estimated to be 0.12 percent of the total population of gray whales in the Action Area based on sighting data.

We determined that sperm whales have the highest relative abundance and therefore, highest annual strike rate in the Hawaii Range Complex, followed by humpback whales and then sei whales. Blue, and western North Pacific gray whales, have much lower relative abundances and subsequently, a lower strike rate. Fin whales have the highest relative abundance and therefore, highest annual strike rate in the SOCAL Range Complex, followed by sperm whales, blue whales, and humpback whales. Sei and western North Pacific gray whales have much lower relative abundances and a lower strike rate.

Our analysis does not consider different behaviors of these species such as time spent at the surface (e.g., blow rates, loafing, feeding, etc.) or ability to detect and avoid ships. It is meant to estimate the rate of vessel strike assuming all individuals are available (at or near the surface) to be struck. These assumptions suggest the calculated risk of vessel strike may be overestimated, particularly for species that spend more time diving below the surface.

Step Two: Calculate the number of years it would take to strike one whale of a species.

To determine the number of years that, on average, would be expected to pass between strikes of individuals of each species we divided one year by the apportioned strike rate (#/yr) by species

Step Three: Calculate the percent chance of striking one individual of a species in a given year

The annual apportioned strike rate is translated into the percent chance one individual of a particular species will be struck by multiplying the result from Step 1 (annual apportioned strike rate) by 100. The results of these calculations are shown in Table 75 below for the Hawaii Range Complex and Table 76 for SOCAL.

Table 75. Annual Risk of Vessel Strike by Species in the Hawaii Range Complex

Hawaii Range Complex					
Common Name	**Average Annual Abundance**	**Total Annual Relative Abundance**	**Step 1 Annual Apportioned Strike Rate**	**Step 2 Number of Years to Strike One Whale**	**Step 3 Percent Chance That One Individual Will Be Struck in a Given Year**
	All Seasons	**All Seasons**			
Common or dwarf minke whale	2,781	0.135609	0.027122	36.87	2.71%
Sei whale	**837**	**0.040811**	**0.008162**	**122.52**	**0.82%**
Bryde's whale	979	0.047732	0.009546	104.76	0.96%
Blue whale	**332**	**0.016206**	**0.003241**	**308.55**	**0.32%**
Fin whale	**557**	**0.027149**	**0.005430**	**184.16**	**0.54%**
Gray whale (WNP)	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
Gray whale (ENP)	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
Humpback whale	**5,031**	**0.24527**	**0.049054**	**20.39**	**4.91%**
Sperm whale	**9,993**	**0.487223**	**0.097445**	**10.26**	**9.75%**
Total	**20,510**				**20.0%**

Table 76. Annual Risk of Vessel Strike by Species in the SOCAL Range Complex

SOCAL Range Complex					
Common Name	**Average Annual Abundance**	**Average Annual Relative Abundance**	**Step 1 Annual Apportioned Strike Rate**	**Step 2 Number of Years to Strike One Whale**	**Step 3 Percent Chance That One Individual Will Be Struck in a Given Year**
	All Seasons	**All Seasons**			
Common or dwarf minke whale	163	0.032494	0.009748	102.58	0.97%
Sei whale	**36**	**0.007280**	**0.002184**	**457.88**	**0.22%**
Bryde's whale	4	0.000881	0.000264	3783.76	0.03%
Blue whale	**514**	**0.102608**	**0.030783**	**32.49**	**3.08%**
Fin whale	**2,307**	**0.460267**	**0.138080**	**7.24**	**13.81%**
Gray whale (WNP)	**3**	**0.000299**	**0.000090**	**11142.76**	**0.01%**
Gray whale (ENP)	2,496	0.248991	0.074697	13.39	7.47%
Humpback whale	**204**	**0.040622**	**0.012187**	**82.06**	**1.22%**
Sperm whale	**534**	**0.106555**	**0.031967**	**31.28**	**3.20%**
Total	**5,013**				**30%**

From our calculations above, we determined the chance of striking a fin whale in SOCAL is 13.81 percent in a given year or one fin whale being struck every 7.24 years on average (Table

76). The chance of striking a sperm whale in Hawaii in a given year is 9.75 percent or one sperm whale being struck every 10.26 years on average (Table 75). Humpback whales have a 4.91 percent chance of being struck in Hawaii in a given year which equates to one humpback whale being struck every 20 years on average (Table 75). Other ESA-listed whales including sei and blue whales have a substantially lower risk of being struck. While the risk of striking sei and blue whales is not as high as other whales in a given year, sufficient risk remains so as not to be discountable during the remainder of the five-year MMPA rule and LOAs, and for the Navy's action into the reasonably foreseeable future. We determined the chance of striking a western north Pacific gray whale was sufficiently low to be discountable during the remainder of the five-year MMPA rule and LOAs and into the reasonably foreseeable future due to extremely low abundance in the HSTT Action Area. Based on the historic strike rate of 0.2 whales struck per year, there is a 20 percent chance of striking a large whale in the Hawaii Range Complex in a given year and a 30 percent chance of striking a large whale in SOCAL in a given year.

To estimate take we project the historic 10-year strike history for SOCAL and Hawaii forward over a 10-year period. We then calculate species-specific take based on our probability analysis. We believe that a 10-year period establishes a reasonable time frame to allow us to meaningfully estimate and evaluate take (injury and mortality) from vessel strike. A smaller time frame would risk underestimating the potential impacts of the action on the survival and recovery of the listed whales assessed in this Opinion. In addition, a ten year time frame can be extrapolated forward into the reasonably foreseeable future for purposes of our jeopardy analysis.

Step 4. Calculate the number of ESA-listed whales expected to be struck over a 10-year period

To calculate (Table 77) the number of ESA-listed whales that are expected to be struck over a 10 year period, we multiply the apportioned annual strike rate by 10. If the status of the species and Navy strike history remain the same, we would expect the estimated number of strikes in a 10-year period to continue into the reasonably forseeable future.

Table 77. Estimated 10-year strike rate for the SOCAL and Hawaii range complexes

	SOCAL Range Complex		**Hawaii Range Complex**		**Total**
Common Name	**Annual Apportioned Strike Rate**	**Expected 10-year strike rate**	**Annual Apportioned Strike Rate**	**Expected 10-year strike rate**	**Expected 10-year strike rate**
Common or dwarf minke whale	0.009748	0.09748	0.027122	0.271217	0.368697
Sei whale	**0.002184**	**0.02184**	**0.008162**	**0.081621**	**0.103461**
Bryde's whale	0.000264	0.00264	0.009546	0.095464	0.098104
Blue whale	**0.030783**	**0.30783**	**0.003241**	**0.032413**	**0.340243**
Fin whale	**0.13808**	**1.3808**	**0.005430**	**0.054298**	**1.435098**
Gray whale (WNP)	**0.00009**	**0.0009**	**0.00**	**0.00**	**0.0009**
Gray whale	0.074697	0.74697	0.00	0.00	0.74697

	SOCAL Range Complex		**Hawaii Range Complex**		**Total**
Common Name	**Annual Apportioned Strike Rate**	**Expected 10-year strike rate**	**Annual Apportioned Strike Rate**	**Expected 10-year strike rate**	**Expected 10-year strike rate**
(ENP)					
Humpback whale	**0.012187**	**0.12187**	**0.049054**	**0.490541**	**0.612411**
Sperm whale	**0.031967**	**0.31967**	**0.097445**	**0.974446**	**1.294116**
Total	*0.3 whales*	*3 whales*	*0.2 whales*	*2 whales*	*5 whales*

Note: The whales in bold are ESA-listed. The total expected number of ESA-listed whales to be struck over 10 years is 3.8 whales.

Based on our analysis above, we would expect up five large whales to be struck in the HSTT Action Area over a 10-year period consisting of no more than four (3.8) ESA-listed whales. Of these four ESA-listed whales we expect no more than two (1.6) ESA-listed whales to be struck in the Hawaii Range Complex and no more than two (2.2) ESA-listed whales in the SOCAL Range Complex. Based on our probability analysis, we estimate that two strikes likely would involve fin whales and two strikes likely would involve other ESA-listed whales (blue, humpback, fin, sei or sperm whales), not to exceed one of each species. We also anticipate this risk to be greater in geographic areas in SOCAL and Hawaii with high vessel traffic densities (See Figure 20) and higher during seasons with greatest abundance of ESA-listed whale species (See Figure 21 and Figure 22) although we do not know the degree to which risk is diminished outside of these areas or seasons.

We acknowledge that strike risk is influenced by a variety of factors that may change over time. For example, our projection of future strike risk is based upon the number of Navy strikes occurring during the 10-year period 2005 to 2014. However, all five strikes during that period occurred prior to 2010. If recent trends continue, it would significantly affect the calculation of strike risk beyond the 10-year window used for this analysis. Future strike risks beyond ten years are also likely to be affected by updated population and relative abundance data documented in stock assessment reports or other scientific studies, changes in Navy activity levels, and other changes in the environmental baseline for the affected species. However, to be conservative and to analyze the impact of the Navy's activities into the reasonably foreseeable future, we anticipate this rate to continue into successive 10-year periods into the reasonably foreseeable future.

We believe that analyzing strike risk based on a ten year period and the Navy's actual strike levels over that period based on our probability analysis is reasonable and appropriate and will allow us to meaningfully evaluate strike risks and the resultant impacts of the Navy's ongoing activities in the HSTT Action Area. However, because MMPA take authorization is limited to five years, we note that with regard to the remainder of the five-year period of the current MMPA Rule and LOAs, we would expect up to three (2.5) large whales consisting of no more than two (1.9) ESA-listed whales to be struck. We also note that any proposed new MMPA take authorization for Navy activities in the HSTT Action Area would require consultation under the

ESA, and therefore we will need to reassess strike risks under both the MMPA and ESA prior to the expiration of the current take authorization on December 24, 2018, and not less than every five years thereafter. These periodic reassessments will allow us to update the environmental baseline to reflect any Navy ship strikes that have occurred, any changes in relevant species abundance, and any other relevant factors, and to adjust our calculations of strike risk accordingly. In this manner, we will be able to ensure that our assessment of strike risks from the Navy's ongoing activities and the resultant impacts on listed species is up-to-date and that no impacts are omitted from consideration.

6.6.2.5 ***Response of Whales to Vessel Strike***

Navy operational orders for ship movements are designed to minimize collisions between surface vessels and large whales that might occur in the Action Area. These measures, which include lookouts and watchstanders on the bridge of ships, requirements for course and speed adjustments to maintain safe distances from whales, and having any ship that observes whales to alert other ships in the area, appear to have reduced the historic strike rate over the last ten years. Within the SOCAL Range Complex, the Navy vessels conducting training and testing have struck three whales over the last 10 years including an unknown species in 2006 and two fin whales in 2009. In the Hawaii Range Complex, the Navy struck two whales over the last ten years during relevant training and testing activities, which included one sperm whale and one unknown.

In the event of a ship strike with a whale, researchers have found that the lethality of the collision increases with ship speed (Silber et al. 2010; Vanderlaan and Taggart 2007; Wiley et al. 2011). Vanderlaan and Taggart Vanderlaan and Taggart (2007) found the probability of a lethal strike increased from 20 percent to 100 percent at speeds between 9 and 20 knots, and that lethality from ship strike increased most rapidly between 10 and 14 knots.

We assume that Navy vessels could operate over the full range of ship speeds. The disparity in size between a large whale weighing over 150 tons and a destroyer, the most prevalent type of ship in the U.S. Pacific Fleet surface force, weighing up to 10,800 tons leads us to conclude that most ship strikes would result in the death of the struck animal. Based on this, we expect that if a blue, fin, humpback, sei, or sperm whale is struck by a Navy vessel that it would likely die as a result of that collision.

6.6.2.6 ***Exposure Estimate of Vessel Strike During Training and Testing - Pinnipeds***

The rarity of ship strikes involving pinnipeds combined with the Navy's established operating policies and procedure intended to reduce interactions of Navy assets and listed species, leads NMFS to assume that the exposure risk of collision from surface vessels or submarines is small enough to be discountable and is not discussed further in this Opinion.

6.6.2.7 *Exposure Estimate of Vessel Strike During Training and Testing - Sea Turtles*

Sea turtles are vulnerable to vessel collisions because they regularly surface to breathe and often rest at or near the surface. Unlike when a vessel strikes a large whale, it is difficult to detect when a vessel strikes a turtle. This is largely due to the relatively small size of a sea turtle compared to the vessels used by the Navy in military readiness training and testing activities. When sufficient data is available, sea turtle stranding network data can be used to estimate the magnitude of sea turtle vessel strike in a given area.

The sections below describe how we estimate anticipated ship strike of sea turtles for the HSTT Action Area. In earlier versions of this Opinion, we indicated that turtles in close enough proximity to be at risk of PTS would also be vulnerable to ship strike. We hypothesized that PTS exposures modeled for non-impulsive sonar sources might be a reasonable proxy for ship strike risk since sonar use is co-extensive with vessel movement. However, we recognize that not all Navy vessels transmit sonar and that sonar-bearing vessels do not always transmit active sonar. Thus, this approach would be an underestimate of the number of individual sea turtles that might encounter Navy vessels of sufficient size and speed to be a threat for strike. On the other hand, the modeled area of potential exposure for active sonar resulting in PTS is greater than the width (beam) of most Navy vessel hulls. Therefore, only a portion of the modeled PTS exposures, equal to the average beam width of Navy vessels relative to the modeled area of exposure for active sonar resulting in PTS, might be representative of strike risk. In addition, modeleted PTS is not a good estimate for strike risk in this consultation because all modeled PTS is for green turtles in San Diego Bay resulting from explosive detonations or other starionary or non-hull mounted sources of acoustic energy, none of which are linked to vessel movements in the HSST Action Area. Therefore, the modeled PTS estimates cannot serve as a means of quantifying sea turtle vessel strikes in this consultation. For these reasons, we determined that the use of modeled PTS estimates is not an appropriate means of estimating ship strike risk for HSTT activities.

6.6.2.7.1 Hawaii

For vessel strike of turtles in Hawaii, our analysis is based on the vessel strike analysis performed in the 2009 Biological Opinion for the continued authorization of Pelagic Troll and Handline Fisheries in the West Pacific Region (NMFS 2009b) (NMFS 2009a). This methodology used turtle stranding data to estimate the total number of turtles that were struck in the Main Hawaiian Islands annually and then determined the proportion of those turtles that were likely struck by fishing vessels. We focus on the MHI because Navy vessels are based out of Pearl Harbor and are concentrated around this area.

Turtles struck far offshore are less likely to strand than those struck in more nearshore waters. However, we have no empirical data to indicate Navy vessels strike turtles in offshore waters and also no data to indicate what percent of those that may be struck in offshore waters strand. As described in Section 6.6.1.13, the density of sea turtles is substantially lower in offshore waters compared to nearshore. Therefore, turtle vessel strike in Hawaii is calculated solely based on stranding data for the MHI, as presented in the 2009 Pacific Islands Region Biological Opinion.

Step 1. Determine the number of Navy vessel trips in the MHI per year.

The Navy obtained data from the Harbormaster on ship movements in and out of Pearl Harbor from 2010 to 2014. The average number of Navy ship transits in and out of Pearl Harbor per year was 988. While this is the best available estimate of the number of ship transits in and out of Pearl Harbor for Navy training and testing, this number also includes local training for Pearl Harbor home-ported vessels, stops by West Coast-based ships prior to deployment to the Western Pacific, and any operational vessel movement, not related to training and testing.

Step 2. Estimate the percent of total vessel trips in the MHI that are composed of Navy transits in and out of Pearl Harbor.

The 2009 Biological Opinion estimated an average of 577,872 vessel trips occur in the MHI each year. This includes commercial fishing vessels, non-fishing commercial vessels, non-commercial vessels (e.g., registered recreational boats), cargo ships, ferries, cruise ships, and military vessels. By dividing the number of Navy transits by this total number of vessel trips, we estimate that Navy vessel transits account for 0.17 percent of all vessel trips in the MHI.

Step 3. Estimate the number of turtles struck in the MHI by Navy vessels.

The 2009 Biological Opinion estimated between 25 and 50 green sea turtles and 0.20 to 0.40 hawksbill sea turtles are killed annually from vessel strike in the MHI. This is based on stranding data (average of 10 stranded green sea turtles and 0.08 hawksbill sea turtles observed per year with signs of vessel strike) and the estimated percent of struck turtles expected to strand (20 to 40 percent) (NMFS 2008. We then multiply the percent of all vessel trips in the MHI that are composed of Navy transits by the number of turtles killed annually in MHI. To err on the side of conservation for the species, we assume only 20% of all vessel-struck turtles strand.

Green turtles: 50 stranded turtles x 0.17 percent = 0.09 green turtles per year

Hawksbill turtles: 0.40 stranded turtles x 0.17 percent = 0.0007 hawksbill turtles per year.

Leatherback and loggerhead sea turtle strandings in the MHI are less frequent than hawksbill sea turtle strandings. The stranding data for these species are so infrequent, meaningful estimations of vessel strikes cannot be calculated. Therefore, we assume no more than 0.0007 individuals of these species will also be struck per year.

6.6.2.7.2 SOCAL

We calculate ship strike for SOCAL based on green turtle stranding data from San Diego Bay. We do not estimate ship strike for Navy vessel movement outside of San Diego Bay in SOCAL.

Relative to Hawaii, there is not a high level of sea turtle stranding activity on the West Coast (R. LeRoux, personal communication, December 5, 2014), including in the SOCAL Action Area. Additionally, available survey data indicate that sea turtles are rarely observed off the southern California coast (Eguchi and Seminoff 2011), indicating very sparse density and low population abundance. Because the available scientific and commercial data indicate such low sea turtle abundance in SOCAL (outside of San Diego Bay), we do not anticipate ship strike of sea turtles to occur in this area.

Between 1990 and 2014, there have been 10 green turtle strandings in San Diego Bay with evidence of ship strike. Though available information does not allow us to estimate the percentage of vessel struck turtles that are observed stranded in San Diego Bay, we expect this observed stranding rate to be fairly high due to the human population along the bay (T. Eguchi, personal communication, February 25, 2015) and that San Diego Bay is relatively enclosed. Therefore, using stranding rates from Hawaii as a proxy (i.e., 20 to 40 percent of struck turtles are likely to strand)(NMFS 2008), we assume 40 percent of vessel struck sea turtles will strand. From the 10 observed strandings in San Diego Bay with evidence of vessel strike over the past 25 years, we estimate a total of 25 turtles were struck by vessels during this time period (10 turtles / 0.4 = 25 turtles). This comes to an annual strike rate of one sea turtle per year (25 turtles / 25 years = one turtle per year).

Navy vessels account for a large percentage of the vessel traffic in San Diego Bay, but significant traffic also occurs from the many recreational and commercial vessels that frequent the Bay. We conservatively estimate that Navy vessel traffic accounts for 50 percent of the vessel movement within San Diego Bay. Therefore, we expect that on average, the Navy will strike up to one green turtle in San Diego Bay per year (1 turtle per year x 0.5 = 0.5; rounded up to 1 turtle). We do not expect strikes of other sea turtles species in San Diego Bay because of low, to no, abundance of any species besides green turtles (see Section 6.6.1.13).

6.6.2.8 ***Response of Sea Turtles to Vessel Strike***

We conclude that encounters of sea turtles with vessels that result in injury or mortality are possible, as are encounters with recreational boating and other vessels of similar class. Collisions with vessels would likely result in blunt trauma, lacerations, and mortality.

6.7 Cumulative Effects

Cumulative effects include the effects of future State, tribal, local, or private actions that are reasonably certain to occur in the Action Area considered in this 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.

During this consultation, we searched for information on future state, tribal, local, or private actions reasonable certain to occur in the Action Area. Activities that are likely to occur in the Action Area during the remaining period of the five-year rule and into the reasonably foreseeable future include continuing commercial shipping and other vessel traffic, seismic surveys, fisheries interactions, scientific research, and whale watching. We expect these actions to continue at rates and intensities similar to those already considered in the *Environmental Baseline*.

6.8 Integration and Synthesis

Our effects analyses identified the probable risks the Navy's training exercises and testing activities pose to ESA-listed individuals that will be exposed to those activities. 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 U.S. Navy conducts in the HSTT Action Area will continue to introduce a suite of stressors into the marine and coastal ecosystems of the Hawaiian Islands, areas around the SOCAL Range Complex, and the transit lanes in between. The stressors include: vessel strike, 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, helicopters, and through the hulls of hulks that are the targets of sinking exercises. 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 6.1.11 and 6.2.7. 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 HSTT Action Area would be exposed to combinations of stressors produced by the Navy's 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 purposes of this Opinion we assume that the Navy's activities in the HSTT Action Area and associated impacts will continue into the reasonably foreseeable future at the levels set forth herein with respect to vessel strikes and in the Final EIS/OEIS and MMPA rule with respect to other stressors. 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 on this stressor is not warranted. However, repeated exposure of individuals to acoustic stress can cause auditory fatigue and hearing loss. For the reasons discussed above, 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 HSTT 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 level training would typically involve a few hours of sonar use, with a typical nominal ping of one every minute (duty cycle).

As dicussed above in section 6.6.1.3.4, a small subset of major training exercises conducted in the HSTT Action Area can last several weeks, and during those exercises there may be extended periods of prolonged sonar use. The majority of HSTT major training exercises are often only one week or less in length, and not every major training exercise has anti-submarine warfare events where active sonar is used. However, even the longest periods of active sonar use rarely last longer than 12 hours per vessel. The sonar use would be conditioned on the training objectives and length of a specified anti-submarine warfare period, which in and of itself can be a sub-set of time within any given major training exercise. Vessels equipped with the most powerful sonar systems would also generally be moving at speeds of 10 to 15 knots and typically separated by miles 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 minimizes the risk of significant behavioral responses or repeated disturbances.

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, and for reasons stated earlier, the Navy’s HSTT activities do not create conditions of chronic, continuous underwater noise and are unlikely to lead to more than temporary changes in habitat selection or are unlikely to lead to long-term hormonal or physiological stress responses in marine mammals. For a more thorough discussion of repeated exposures to marine mammals (see Section 6.1.11) and response of blue whales to acoustic stressors as discussed in Section 6.6.1.3.4.

Ship strike also does not accumulate in the environment, as the likelihood of being struck by an individual vessel is independent of other vessels; however, the likelihood of a strike occurring does increase with an increase in vessel traffic. The exposure risk of ESA-listed species to Navy vessel strike already accounts for Navy vessel traffic. Since ship strikes do not accumulate in the environment, an accumulative effects analysis of this stressor is not warranted.

Our assessment that the continuation of the Navy's 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 Navy training in the HSTT Action Area using many of the same systems. Most of the training activities the Navy conducts in the SOCAL and Hawaii range complexes 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. 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 conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours. The effects of the action in relation to each species' status (Section 4.3) and the *Environmental Baseline* are presented below.

6.8.1 Blue Whale

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of blue whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, some of the primary anthropogenic threats to the survival and recovery of blue whales have been whaling and ship strikes. The threat of whaling has been eliminated and despite increases in shipping traffic worldwide, recent evidence indicates that the blue whale population in the eastern North Pacific is at carrying capacity. Monnahan et al. (2014b) calculated that this population currently totals 2,138 blue whales.

Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that 1,786 blue whales could experience behavioral responses from non-impulsive acoustic sources during training and that 2,535 whales could experience TTS. We do not expect any blue whales to experience PTS from non-impulsive sources during training. In addition to non-impulsive stressors, we estimated that one blue whale could experience behavioral responses from impulsive acoustic sources during training and three would experience TTS. Zero blue whales would be expected to experience PTS during training.

We estimated that 124 blue whales could experience behavioral responses from non-impulsive acoustic sources during testing and that 303 whales that could experience TTS. We do not expect any blue whales to experience PTS, GI tract or lung injury, or to be killed from non-impulsive sources during testing. In addition to non-impulsive stressors, we estimated that zero blue whale could experience a behavioral response from impulsive acoustic sources during testing and one would experience TTS. Zero blue whales would be expected to experience PTS, GI tract or lung injury, or would be killed during testing.

Most of the blue whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the Hawaii and SOCAL range complexes. Given the nature of Navy testing and training as described above, 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. As described previously, including in Sections 6.1.11 and 6.6.1.3.4 of 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.

Our analysis on the risk of vessel strike indicates that on average, we would anticipate a blue whale to be struck once every 309 years in the Hawaii Range Complex and once every 33 years in the SOCAL Range Complex based on their abundance relative to other large whales in these areas. It is impossible to predict when in these periods of time a whale might be struck.

In our analysis of the overall strike risk for ESA-listed whales, we concluded that four ESA-listed whales would be struck over a 10-year period. Of those four, two would likely consist of fin whales and the remaining two would consist of other ESA-listed whale species including blue whales. While we determined that the risk of striking a blue whale in a 10-year period is relatively low, it is not sufficiently low to be discountable. Therefore we anticipate that up to one blue whale could be struck within a notional 10-year period. As such, we must assess the effects of potential removal of one blue whale over a 10-year period. We anticipate the long-term rate (i.e., one every 33 years in SOCAL and one every 309 in Hawaii Range Complex) would continue into the reasonably foreseeable future. In this scenario, if a blue whale is struck within the first ten years, we would not expect a second blue whale to be struck for another 33 years in SOCAL and another 309 years in the Hawaii Range Complex, on average.

A single death of a blue whale from vessel stike in a 10-year period would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to the population or sub-population. As we also discussed in section 6.1.13, this lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. The death of a male blue whale would have substantially less of an effect on the population than the loss of a female. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years. Blue whales reach sexual maturity at around 5 years of age. Under ideal conditions, mature females give birth to one calf every two to three years (2.5 years on average). Assuming that a female blue whale's lifespan is 80 to 90 years and that it reaches sexual maturity at age five, the maximum reproductive potential of that female would be 32 to 36 offspring with approximately half of those being female. Of those potential offspring, it is difficult to assess how many would survive (given natural mortality and anthropogenic effects) to sexual maturity to contribute to reproduction.

As stated previously, the ENP blue whale population is estimated to total 2,138 whales. Assuming a sex ratio of 1.41:1 (females : males)(Costa-Urrutia, Sanvito et al. 2013), this means 1,507 females likely exist in this population. Removing one female from this population via vessel strike over 10 years would reduce the reproductive potential of this population by 0.07 percent. This is not an appreciable reduction in the numbers of ENP blue whales or the reproductive rate of the population. If the analyzed rate of vessel strike for this species was to recur in subsequent 10-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. It is worth noting that the ENP blue whale population is a subset of the range-wide population of blue whales. Therefore, we also conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the species range-wide.

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 HSTT 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 HSTT activities is not expected to impact the fitness of any individuals of this species. Based on available information, we also conclude the removal of one blue whale over 10 years from collision with vessels, in the next 10-year period and into the reasonably foreseeable future, is not an appreciable reduction in the numbers or the reproductive capability of this species in the North Pacific Ocean.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival or the recovery of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. 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 the species as listed pursuant to the ESA. This determination is further supported by (Monnahan et al. 2014a), who suggests that the eastern North Pacific blue whale population is at carrying capacity and recovered to pre-whaling levels in light of existing stressors including U.S. Navy training and testing and other activities as we discussed in the *Environmental Baseline* of this Opinion. Other factors such as shifts in distribution may be occurring and thus the increasing trend may not apply to the species' entire range in the North Pacific. Regardless of the reason for increases in abundance of blue whales in Southern California, there is currently no evidence to indicate that Navy training and testing activities are having or are likely to have a measurable effect on population growth.

6.8.2 **Fin Whale**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of fin whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, some of the primary anthropogenic threats to the survival and recovery of fin whales have been whaling, entanglement in fishing gear, and ship strikes. The threat of whaling has largely been eliminated. While efforts have been made to reduce the interaction of fin whales with fishing gear and the likelihood of ship strike, these threats remain. The degree to which these specific threats are inhibiting the recovery of fin whale populations is unknown, but recent evidence indicates that populations are increasing in certain areas of the North Pacific, including off the U.S. West Coast. 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).

Our analysis on the effects of acoustic stressors on fin whales concluded that non-impulsive acoustic source will result in 701 instances of fin whales exposed to received levels that will lead to behavioral responses that NMFS classifies as harassment during training activities and 56 instances resulting in behavioral responses during testing activities. In addition, there would be 1,018 instances in which fin whales will experience TTS during training exercises and 169 instances during testing activities. There would be zero instances in which fin whales will experience PTS during training exercises and zero during testing activities.

Our analysis on the effects of acoustic stressors on fin whales also concluded that impulsive acoustic stressors will result in zero instances of a fin whale exposed to received levels that will result in a behavioral response during training and none during testing activities. There will also be one instance in which a fin whale could accumulate energy sufficient to result in TTS during training and zero during testing activities. No fin whales are expected to accumulate energy sufficient to result in PTS from either training exercises or testing activities. No fin whales would experience GI tract or lung injury, and no fin whales would be killed from acoustic stressors associated with either training or testing activities.

Most of the fin whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the Hawaii and SOCAL range complexes. For the reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including Sections 6.1.11 and 6.6.1.4.4, the available scientific information does not provide evidence that exposure to acoustic stressors from Navy training and testing activities are likely to impact the fitness of any individuals of this species. Therefore, exposure to acoustic stressors are not likely to result in adverse population or species level impacts.

Of the overall risk of vessel strike to large whales in SOCAL, the chance of striking a fin whale is greatest at 13.81 percent. This would equate to one fin whale being struck every 7.24 years on average. In the Hawaii Range Complex, the chance of striking a fin whale is 0.54 percent or one fin whale being struck every 184 years on average. In our analysis of the overall strike risk for ESA-listed whales, we concluded that four ESA-listed whales would be struck over a 10-year period. Of those four, two would likely consist of fin whales and the remaining two would consist of other ESA-listed whale species.This leads us to conclude that approximately up to one fin whale could be struck over a five-year period such as the MMPA rule and LOA, and up to two could be struck over a notional 10-year period. We anticipate the long-term rate (i.e., one every 7.24 years in SOCAL and one every 184 in Hawaii Range Complex) would continue into the reasonably foreseeable future. In this scenario, if a fin whale is struck within the first ten years, we would not expect a second fin whale to be struck for another 7.24 years in SOCAL and another 184 years in the Hawaii Range Complex, on average.

As such, we must assess the effects of potential removal of one fin whale over a five-year period or two whales over a 10-year period. Death would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to to the population or sub-population. As we also discussed in section 6.1.13, this lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. Fin whales reach sexual maturity at between 4 and 8 years (six years on average) of age. Under ideal conditions, mature females give birth to one calf every two to three years (2.5 years on average). Assuming that a female fin whale’s average lifespan is 75 years and with it reaching sexual maturity at age six, the maximum reproductive potential of that female would be 28 offspring with approximately half of those being female. Of those potential offspring, it is difficult to assess how many would survive (natural mortality and anthropogenic effects) to sexual maturity to further contribute to reproduction.

As stated previously, the North Pacific fin whale population is estimated to total 16,625 whales. Assuming a sex ratio of 1:1 (IUCN), this means 8,312 females likely exist in this population. In the worst case scenario, the one fin whale expected to be struck in five years or the two fin whales expected to be struck over 10 years by Navy vessels would be all females. The two deaths in 10 years would reduce the reproductive potential of this population by 0.02 percent. This is not an appreciable reduction in the numbers or the reproductive capability of fin whales in the North Pacific Ocean. If the analyzed rate of vessel strike for this species was to recur in subsequent 10-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. It is also worth noting that the North Pacific population is a subset of the range-wide population of fin whales. Therefore, we also conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive capability of the species range-wide.

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 HSTT activities. As discussed previously, anthropogenic noise associated with HSTT 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. However, we conclude the removal of 2 fin whales over 10 years from collision with vessels is not an appreciable reduction in the numbers or the reproductive capability of this species in the North Pacific Ocean.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, 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 U.S. 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. That is, acoustic stressors will not have fitness consequences and vessel strike will not result in an appreciable reduction in reproductive capability at the population or range-wide level. 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 the species as listed pursuant to the ESA.

6.8.3 Western North Pacific Gray Whale

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of western north Pacific gray whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, some of the primary anthropogenic threats to the survival and recovery of Western North Pacific gray whales have been whaling, entanglement in fishing gear, and ship strikes. Though the threat of large scale commercial whaling has been eliminated, some aboriginal harvest of this population still may occur (though the focus of aboriginal harvest of this species is the Eastern North Pacific population which has been delisted) and the threats of ship strike and entanglement remain. The most recent abundance estimates of this population total 140 individuals. This is very low relative to pre-whaling abundances estimated to total between 76,000 and 118,000. However, the latest data indicates positive population growth, despite these ongoing anthropogenic threats.

In our risk analysis for acoustic stressors, we estimated that two western north Pacific gray whales could experience behavioral responses and seven whales could experience TTS due to non-impulsive acoustic sources during training exercises. Zero western north Pacific gray whales would be expected to experience PTS or other injury or mortality during training exercises. We also estimated that one western north Pacific gray whale could experience a behavioral responses from impulsive acoustic sources during training exercises while zero gray whales would experience TTS, PTS, GI tract, lung injury and none would be killed during training exercises.

Additionally, we concluded that no western north Pacific gray whales will experience behavioral responses while two whales would experience TTS due to non-impulsive acoustic sources during testing activities. No western north Pacific gray whales would experience PTS, GI tract injury, lung injury, or be killed during testing activities. Lastly, we concluded that no western north Pacific gray whales will experience behavioral responses, TTS, PTS, GI tract or lung injury, or mortality due to impulsive acoustic sources during testing activities.

For the reasons described previously, including in Sections 6.1.11 and 6.6.1.6, the available scientific information does not provide evidence that the very limited number of exposures to acoustic stressors from Navy training and testing activities will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors is not likely to have adverse population or species level impacts.

Since western north Pacific gray whales are present in the HSTT Action Area in such low abundance, we consider the vessel strike risk to this species to be discountable. Therefore, we do not anticipate any injury or mortality of gray whales from vessel strike during the remainder of the five-year period of the MMPA Rule or LOAs or in the reasonably foreseeable future.

The western north Pacific gray whale 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.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of western north Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. 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 western north Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. That is, acoustic stressors will not have fitness consequences at the population or range-wide level. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA.

6.8.4 **Humpback Whale**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of humpback whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of humpback whales have been whaling, entanglement in fishing gear, and ship strikes. The threat of whaling has largely been eliminated. While efforts have been made to reduce the interaction of humpback whales with fishing gear and the likelihood of ship strike, these threats remain. The degree to which these specific threats may inhibit the recovery of humpback whale populations is unknown, but recent evidence indicates that the overall abundance of humpback whales is increasing off the U.S. West Coast and in Hawaiian waters. Barlow et al. (2011a) estimated the humpback whale population in the North Pacific to total 21,808 individuals, suggesting that abundance of the species is now greater than some pre-whaling abundance estimates.

In our risk analysis for acoustic stressors, we determined that 3,299 humpback whales will experience behavioral responses in a given year due to non-impulsive acoustic sources during training activities while 272 would experience behavioral responses during testing activities. We estimated that 5,974 humpback whales will accumulate energy sufficient to result in TTS due to non-impulsive acoustic sources during training exercises and 649 during testing activities. No humpback whales would be expected to experience PTS, GI tract, lung injury, or mortality from either training exercises or testing activities from non-impulsive acoustic sources.

We also concluded that no humpback whales will experience behavioral responses, TTS, PTS, GI tract or lung injury, or would be killed due to impulsive acoustic sources during training. We estimated that no humpback whales that could experience behavioral responses due to impulsive acoustic sources during testing activities in addition to those behavioral responses that may be associated with the six whales that are expected to experience TTS. No humpback whales are expected to experience PTS, GI tract, lung injury, or be killed as a result of impulsive acoustic sources during either training exercises or testing activities.

Most of the humpback whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the Hawaii and SOCAL range complexes. For ther reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including in Sections 6.1.11 and 6.6.1.7.4, 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 is not likely to have population or species level impacts.

Based on the historic strike rate and relative abundance we determined that humpback whales have a 4.91 percent chance of being struck by Navy vessels in a given year in the Hawaii Range Complex which would equate to one humpback whale being struck every 20 years on average. In SOCAL, we determined that humpback whales have a 1.22 percent chance of being struck by Navy vessels in a given year, which equates to one humpback whale being struck every 82 years on average. Even though the potential to strike a humpback whale in Hawaii or SOCAL in a given year is much lower than other ESA-listed whales, the potential is not so low as to be discountable. In our analysis of the overall strike risk for ESA-listed whales, we concluded that four ESA-listed whales would be struck over a 10-year period. Of those four, two would likely consist of fin whales and the remaining two would consist of other ESA-listed whale species including humpback whales. Therefore, we include humpback whales in the estimated two ESA-listed whales of four overall in a 10-year period that could be struck. Due to the low potential to strike humpback whales (≤4.91 percent chance in a given year), we do not anticipate that more than one humpback whale would be struck in a 10-year period.

As such, we must assess the effects of potential removal of one humpback whale over a 10-year period. We anticipate the long-term rate (i.e., one every 82 years in SOCAL and one every 20 in Hawaii Range Complex) would continue into the reasonably foreseeable future. In this scenario, if a humpback whale is struck within the first ten years, we would not expect a second humpback whale to be struck for another 82 years in SOCAL and another 20 years in the Hawaii Range Complex, on average. Death would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to to the population or sub-population. As we also discussed in section 6.1.13, this lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual.

Humpback whales reach sexual maturity at around ten years of age. Under ideal conditions, mature females give birth to one calf every two years on average. Assuming that a female humpback whale’s average lifespan is 77 years and with it reaching sexual maturity at age five, the maximum reproductive potential of that female would be 39 offspring with approximately half of those being female. Of those potential offspring, it is difficult to assess how many would survive (natural mortality and anthropogenic effects) to sexual maturity to contribute to reproduction.

As stated previously, the North Pacific humpback whale population is estimated to total 21,808 whales. Assuming a sex ratio of 1:1 (IUCN), this means 10,904 females likely exist in this population. Removing one female from this population via vessel strike over 10 years would reduce the reproductive potential of this population by 0.009 percent. This is not an appreciable reduction in the numbers or the reproductive capability of humpback whales in the North Pacific Ocean. If the analyzed rate of vessel strike for this species was to recur in subsequent 10-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species’ ability to recover. It is also worth noting that the North Pacific population is a subset of the range-wide population of humpback whales. Therefore, we also conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the species range-wide.

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 HSTT activities. As described previously, anthropogenic noise associated with HSTT activities will not impact the fitness of any individuals of this species. We also conclude the removal of 1 humpback whale over 10 years from collision with vessels is not an appreciable reduction in the numbers or the reproductive capability of this species in the North Pacific Ocean.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. 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 western north Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. That is, acoustic stressors will not have fitness consequences and vessel strike will not result in an appreciable reduction in reproductive capability at the population or range-wide level. 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 the species.

6.8.5 **Sei Whale**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of sei whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of sei whales have been whaling, entanglement in fishing gear, and ship strikes. The threat of whaling has been greatly reduced, though Japan harvested approximately 100 sei whales annually for scientific research purposes in the North Pacific from 2004 to 2013. While efforts have been made to reduce the interaction of sei whales with fishing gear and the likelihood of ship strike, these threats remain. Sei whale abundance in the North Pacific remains below pre-large scale commercial whaling abundance, but the degree to which these specific threats may inhibit the recovery of sei whale populations is unknown. The most recent abundance estimates for sei whales that we are aware of range from 7,260 to 12,620 in the North Pacific Ocean (Braham 1991) and 25,000 individuals worldwide (Tillman 1977a).

In our risk analysis for acoustic stressors, we concluded that 226 sei whales will experience behavioral responses due to non-impulsive acoustic sources during training exercises while 16 whales will experience behavioral responses during testing activities. We estimated that 404 sei whales will accumulate energy sufficient to result in TTS due to non-impulsive acoustic sources during training exercises and 35 instances of TTS will result from testing activities. We do not anticpate any sei whales to experience PTS, GI tract, lung injury, or mortality from non-impulsive sources during either training exercises or testing activities.

We also concluded that zero sei whales will experience behavioral responses due to impulsive acoustic sources during both training exercises and testing activities. Additionally, no instances of TTS, PTS, GI tract or lung injury, or mortality of sei whales are antipated from impulsive acoustic sources during training exercises and testing activities.

Most of the sei whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the Hawaii and SOCAL range complexes. For ther reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including in Sections 6.1.11 and 6.6.1.8.4, 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.

Based on the historic strike rate and relative abundance we determined that sei whales have a 0.82 percent chance of being struck by Navy vessels in a given year in the Hawaii Range Complex which would equate to one sei whale being struck every 123 years on average. In SOCAL, we determined that sei whales have less risk of being struck by Navy vessels with a 0.22 percent chance each year which equates to one sei whale being struck every 458 years on average. Even though the potential to strike a sei whale in Hawaii or SOCAL in a given year is much lower than other ESA-listed whales, the potential is not so low as to be discountable. In our analysis of the overall strike risk for ESA-listed whales, we concluded that four ESA-listed whales would be struck over a 10-year period. Of those four, two would likely consist of fin whales and the remaining two would consist of other ESA-listed whale species including sei whales. Therefore, we include sei whales in the estimated two ESA-listed whales of four overall in a 10-year period that could be struck. Due to the low potential to strike sei whales (less than a one percent chance in a given year), we anticipate that up to one sei whale would be struck in a 10-year period.

As such, we must assess the effects of potential removal of one sei whale over a 10-year period. We anticipate the long-term rate (i.e., one every 458 years in SOCAL and one every 123 in Hawaii Range Complex) would continue into the reasonably foreseeable future. In this scenario, if a sei whale is struck within the first ten years, we would not expect a second sei whale to be struck for another 458 years in SOCAL and another 123 years in the Hawaii Range Complex, on average. Death would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to to the population or sub-population. As we also discussed in section 6.1.13, this lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual.

Sei whales reach sexual maturity at around ten years of age. Under ideal conditions, mature females give birth to one calf every two to three years (2.5 years on average). Assuming that a female sei whale's average lifespan is 70 years and with it reaching sexual maturity at age ten, the maximum reproductive potential of that female would be 24 offspring with approximately half of those being female. Of those potential offspring, it is difficult to assess how many would survive (natural mortality and anthropogenic effects) to sexual maturity to contribute to reproduction.

As stated previously, the most recent and conservative estimate of the North Pacific sei whale population is 7,260 whales. Assuming a sex ratio of 1:1 (IUCN), this means 3,630 females likely exist in this population. Removing one female from this population via vessel strike over 10 years would reduce the reproductive potential of this population by 0.03 percent. This is not an appreciable reduction in the numbers or the reproductive capability of sei whales in the North Pacific Ocean. If the analyzed rate of vessel strike for this species was to recur in subsequent 10-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species' ability to recover. It is also worth noting that the North Pacific population is a subset of the range-wide population of sei whales. Therefore, we also conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the species range-wide.

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 HSTT activities. As described previously, anthropogenic noise associated with HSTT 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. However, we conclude the removal of one sei whale over 10 years from collision with vessels is not an appreciable reduction in the numbers or the reproductive capability of this species in the North Pacific Ocean.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, 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 U.S. 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. That is, acoustic stressors will not have fitness consequences and vessel strike will not result in an appreciable reduction in reproductive capability at the population or range-wide level. 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 the species as listed pursuant to the ESA.

6.8.6 **Sperm Whale**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of sperm whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of sperm whales have been whaling, entanglement in fishing gear, and ship strikes. The threat of whaling has been greatly reduced, though Japan harvested a small number of sperm whales (from 1 to 10 annually) for scientific research purposes in the North Pacific from 2000 to 2013. The threat of entanglement in fishing gear and ship strike persists. Further, it has been speculated that sperm whale populations may be undergoing the dynamics of small population sizes. The degree to which each of these exogenous and endogenous threats may inhibit the recovery of sperm whale populations in the North Pacific in unknown. Though the species has likely increased in abundance since large scale pelagic whaling stopped in 1980, to date there has not been a statistical analysis to detect trends in abundance of the species (Carretta et al. 2014). The most comprehensive abundance estimate for sperm whales we are aware of is from Whitehead (2002a), who estimated that there are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii and western North Pacific, and a worldwide population of 360,000 individuals.

The estimated number of sperm whales that could experience behavioral responses due to non-impulsive acoustic sources during training exercises was 3,150 and 139 during testing activities. The estimated number of sperm whales that could accumulate energy sufficient to result in temporary shifts in hearing sensitivity due to non-impulsive acoustic sources was 182 during training exercises and 124 during testing activities. Zero sperm whales would be expected to experience PTS from either training exercises or testing activities.

After considering mitigation actions that will be implemented by the Navy plus those measures that will satisfy the requirements of the MMPA, the estimated number of sperm whales that could experience behavioral responses due to impulsive acoustic sources during training exercises and testing activities remained at zero. Zero sperm whales would experience PTS, GI tract or lung injury, and no sperm whale would be killed.

Most of the sperm whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the Hawaii and SOCAL range complexes. For ther reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including in Sections 6.1.11 and 6.6.1.9.4, 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.

Based on our analysis of Navy activities and the relative abundance of sperm whales in the Action Area, we determined that sperm whales have a 9.75 percent chance (one sperm whale being struck every 10.3 years on average) of being struck in the Hawaii Range Complex a given year. In the SOCAL Range Complex, we determined that sperm whales have a 3.20 percent chance (one sperm whale being struck every 31 years on average) of being struck in a given year. In our analysis of the overall strike risk for ESA-listed whales, we concluded that four whales would be struck over a 10-year period. Of those four, two would likely consist of fin whales and the remaining two would consist of other ESA-listed whale species including sperm whales. Therefore, we include sperm whales in the estimated two ESA-listed whales of four overall in a 10-year period that could be struck. Due to the relatively high potential to strike sperm whales (9.75 percent chance in Hawaii and 3.20 percent chance in SOCAL in a given year), we anticipate that up to one sperm whale would be struck in a 10-year period.

As such, we must assess the effects of potential removal of one sperm whale over a 10-year period. We anticipate the long-term rate (i.e., one every 31 years in SOCAL and one every 10.3 years in Hawaii Range Complex) would continue into the reasonably foreseeable future. In this scenario, if a sperm whale is struck within the first ten years, we would not expect a second sperm whale to be struck for another 31 years in SOCAL and another 10.3 years in the Hawaii Range Complex, on average. Death would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to to the population or sub-population. As we also discussed in section 6.1.13, this lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual.

Female sperm whales reach sexual maturity at between 8 and 11 years of age (9.7 years on average). Males reach sexual maturity at around 10 years, although most males do not mate until 25 to 27 years of age because they do not have a high enough social status in the group. Under ideal conditions, mature females give birth to one calf every two to five years (3.5 years on average). Assuming that a female sperm whale’s average lifespan is 77 years and with it reaching sexual maturity at age 9.7, the maximum reproductive potential of that female would be 19 offspring with approximately half of those being female. Of those potential offspring, it is difficult to assess how many would survive (natural mortality and anthropogenic effects) to sexual maturity to contribute to reproduction.

As stated previously, the most recent estimate of the sperm whale population from the eastern tropical Pacific, eastern North Pacific, Hawaii and western North Pacific is 76,803 whales. Assuming a sex ratio of 1:1, this means 38,401 females likely exist in this population. Removing one female from this population via vessel strike over 10 years would reduce the reproductive potential of this population by 0.003 percent. This is not an appreciable reduction in the numbers or the reproductive rate of sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii and western North Pacific. If the analyzed rate of vessel strike for this species was to recur in subsequent 10-year periods into the reasonably foreseeable future, based on the available information and this calculated rate of reduction in reproductive potential, we also believe it is unlikely that there would be an appreciable reduction to survival or reproduction rates or the species’ ability to recover. It is also worth noting that this population is a subset of the range-wide population of sperm whales. Therefore, we also conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the species range-wide.

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 HSTT activities. As discussed previously, anthropogenic noise associated with HSTT 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 spern whale recovery populations. However, we conclude the removal of 1 sperm whale over 10 years from collision with vessels is not an appreciable reduction in the numbers or the reproductive capability of this species in the North Pacific Ocean, which is a component of the Pacific Ocean recovery population.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, 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 U.S. 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. That is, acoustic stressors will not have fitness consequences and vessel strike will not result in an appreciable reduction in reproductive capability at the population or range-wide level. 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 the species as listed pursuant to the ESA.

6.8.7 Main Hawaiian Islands Insular False Killer Whale

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of MHI insular false killer whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of MHI insular false killer whales are interactions with fishing gear, competition for prey with fisheries, and exposure to contaminants. Further, it’s likely that MHI false killer whales are undergoing the dynamics of small population sizes (including inbreeding depression and Allee effects). Trends in sighting data and quantitative analysis indicate that this species is declining in abundance and is at high risk of extinction (Caretta et al. 2014). The most recent and best estimate of population size for the MHI insular stock of false killer whales is 151 animals (Caretta et al. 2014).

In our risk assessment for acoustic stressors, we concluded that 46 MHI insular false killer whales will experience behavioral responses due to non-impulsive acoustic sources during training exercises while two instance of behavioral harassment are anticiapted during testing activities. We also estimated three instances in which MHI insular false killer whales will accumulate energy sufficient to result in TTS due to non-impulsive acoustic sources during training activities and two instances of TTS during testing activities. No MHI insular false killer whales would be expected to experience PTS from either training exercises or testing activities.

We also concluded that zero MHI insular false killer whales will experience behavioral responses, TTS, PTS, GI tract or lung injury, or mortality from impulsive acoustic sources during either training exercises or testing activities.

Most of the MHI false killer whales that are exposed to acoustic stressors would be exposed periodically or episodically during testing and training activies conducted in the Hawaii Range Complex portion of the HSTT Action Area. For the reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including in the sections 6.1.11and 6.6.1.10.4 of 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.

Based on our analysis of Navy activities, the ability of MHI insular false killer whales to avoid oncoming vessels, and the low abundance of this species in the Action Area, we do not anticipate any instances of vessel strike during the remaining period of the five-year rule or in the reasonably foreseeable future.

The MHI insular false killer whale does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of this listed entity 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.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of MHI insular false killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. 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 MHI insular false killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. That is, acoustic stressors will not have fitness consequences at the range-wide level. 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 the species as listed pursuant to the ESA.

6.8.8 Hawaiian Monk Seal

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of Hawaiian monk seals, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of Hawaiian monk seals have been hunting, entanglement with fishing gear and other marine debris, and competition for food resources with commercial fisheries. The threat of hunting has largely subsided, though recent reports indicate that some seals in the Hawaiian Islands are being killed intentionally (Caretta et al. 2014). Further, it's likely that Hawaiian monk seals are undergoing the dynamics of small population sizes (including inbreeding depression and Allee effects). The degree to which each of these exogenous and endogenous threats may inhibit the recovery of the species is unknown. The subpopulation of Hawaiian monk seals in the Northwest Hawaiian Islands, where most of the total population resides, appears to be decreasing, whereas the subpopulation in the Main Hawaiian Islands appears to be increasing. Range wide, the population appears to be declining. The best estimate of the total Hawaiian monk seal population size is 1,209 individuals (Carretta et al. 2014).

In our risk analysis for acoustic stressors, we concluded that 845 Hawaiian monk seals will experience behavioral responses due to non-impulsive acoustic sources during training exercises while 178 seals would be behaviorally harrassed during testing activities. We estimated 446 Hawaiian monk seals will accumulate energy sufficient to result in TTS due to non-impulsive acoustic sources during training exercises and we estimated 180 instances of TTS during testing activities. Zero Hawaiian monk seals would be expected to experience PTS from either training exercises or testing activities.

We also concluded that no additional Hawaiian monk seals will experience behavioral responses outside of the one instance of TTS due to impulsive acoustic sources. We also anticipate that no Hawaiian monk seals would experience PTS, GI tract or lung injury, or moratlity from impulsive acoustic stressors during either training exercises or testing activities.

Most of the Hawaiian monk seals that are exposed to acoustic stressors would be exposed periodically or episodically during testing and training activies conducted in the Hawaii Range Complex portion of the HSTT Action Area. For the reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously, including in sections 6.1.11

and 6.6.1.12.4 of 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.

Based on our analysis of Navy activities and the ability of Hawaiian monk seals to avoid oncoming vessels, we do not anticipate any instances of vessel strike during the remaining period of the five-year rule or in the reasonably foreseeable future.

The 2007 Hawaiian monk seal recovery plan defines seven recovery populations, the French Frigate Shoals, Lisianski Island, Midway Atoll, Laysan Island, Pearl and Hermes Reef, Kure Atoll, and Main Hawaiian Islands, and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with food limitation, debris entanglement, predation, fisheries, erosion, and disease. None of these stressors have been determined to be relevant to HSTT activities with regards to this species. Downlisting criteria for Hawaiian monk seals includes the maintenance of 2,900 individuals in the Northwestern Hawaiian Islands (NWHI), at least 5 of the 6 NWHI sub-populations with at least 100 individuals and the MHI population above 500 individuals, and a population growth rate of zero or higher. To qualify for delisting, the downlisting criteria must be met for 20 consecutive years.As described previously, we conclude the temporary effects from acoustic stressors associated with Navy training and testing activities will not impact the fitness of any individuals of this species or the populations to which they belong.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. 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 Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species. That is, acoustic stressors will not have fitness consequences at the range-wide level. 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 the species as listed pursuant to the ESA.

6.8.9 **Guadalupe Fur Seal**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of Guadalupe fur seals, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threat to the survival and recovery of Guadalupe fur seals is interactions with fishing gear. The degree to which this threat may inhibit the recovery of Guadalupe fur seals is unknown. The most recent estimate is 7,408 animals in 1993 (Carretta et al. 2014c), with a population growth rate of 13.7 percent per year (Carretta et al. 2002). Data indicate that the species is currently increasing in abundance exponentially (Caretta et al. 2014).

In our risk analysis for acoustic stressors, we estimated that 2,596 Guadalupe fur seals will experience behavioral responses due to non-impulsive acoustic sources during training exercises while 269 seals would be behaviorally harassed during testing activities. We concluded that seven Guadalupe fur seals will accumulate energy sufficient to result in TTS due to non-impulsive acoustic sources during training with zero instances of TTS during testing activities. We anticipate that no Guadalupe fur seals will experience PTS from non-impulstive acoustic sources during either training exercises or testing activities.

Additionally, we concluded that Guadalupe fur seals will not experience behavioral responses due to impulsive acoustic sources from either training exercises or testing activities and that no Guadalupe fur seals would experience PTS, slight lung injury, GI tract injury, or be killed from impulsive acoustic sources during training or testing.

Most of the Guadalupe fr seals that are exposed to acoustic stressors would be exposed periodically or episodically during testing and training activies conducted in the HSTT Action Area. For the reasons previously stated with respect to blue whales, 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. Many training exercises and testing activities will occur without any marine animals being exposed to impulsive and non-impulsive sound fields associated with active sonar and underwater detonations. As described previously in sections 6.1.11 and 6.6.1.11.4 of 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.

Based on our analysis of Navy activities and the ability of Guadalupe fur seals to avoid oncoming vessels, we do not anticipate any instances of vessel strike during the remaining period of the five-year rule or in the reasonably foreseeable future.

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.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, 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 U.S. 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. That is, acoustic stressors will not have fitness consequences at the range-wide level. 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 the species as listed pursuant to the ESA.

6.8.10 **Sea Turtles**

In determining whether U.S. Navy training and testing activities in the HSTT Action Area are likely to jeopardize the survival and recovery of sea turtles, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources*, and those effects of future non-Federal activities that are reasonably certain to occur within the Action Area in the reasonably foreseeable future.

As described in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion, the primary anthropogenic threats to the survival and recovery of sea turtles are overharvest (directed harvest of both eggs and adults), incidental capture in commercial fisheries, and human development of coastlines. Harvest of sea turtles has been greatly reduced in some locations, though it still occurs in other parts of the world, including areas in the Pacific Ocean. Further, efforts have been made in some areas to reduce incidental capture in fisheries, but the problem persists. For some sea turtles (i.e., leatherbacks and loggerheads), available information indicates that population abundances in the North Pacific Ocean are low enough to experience the dynamics of small population sizes. However, the degree to which this, versus other exogenous threats, contributes to the decline, or inhibits recovery, of these species is unknown.

With the exception of the subpopulation in Hawaii, green turtle abundance is thought to be declining throughout the Pacific Ocean. Worldwide nest counts for green sea turtles average between 109,000 and 150,000 nests per year (USFWS 2007). Nest counts specific to the endangered breeding population along the Pacific coast of Mexico (the population that may use southern California for foraging) average roughly 6,050 nests per year (USFWS 2007). Range-wide, hawksbill populations are only a fraction of their historical abundance (USFWS 2007) and nest counts throughout the Pacific are decreasing. An average of 11,000 to 12,700 hawksbill nests occur each year in the Pacific. Leatherback populations in the North Pacific Ocean, and worldwide, currently face high probabilities of extinction. The most recent estimates of leatherback nests in the Pacific Ocean total 3,172 nests per year (USFWS 2013), with the population trending downward. For loggerheads, nest counts from along the Japanese coast (the location of the primary nesting beaches for the North Pacific DPS) indicate a substantial decline in the size of the nesting population in recent decades. The most recent estimate of loggerhead nests in Japan averaged 4,133 nests per year (USFWS 2007). More recently, Seminoff et al. (2014) estimated a mean annual abundance of 43,226 loggerhead turtles off the Pacific Coast of the Baja California Peninsula, a known foraging hotspot of the species. The endangered population of olive ridley sea turtles that nests on the Pacific coast of Mexico appears to be increasing in abundance. Eguchi et al. (2007) studied the at-sea abundance of this population and estimated the annual average abundance to total 1.39 million individuals. Trends for the threatened population of olive ridley sea turtles, which includes all subpopulations that do not nest on the Pacific Coast of Mexico, are less clear. However, substantial numbers of nesting turtles are observed when compared with the other turtle species considered in this Opinion. For example, the Ostional nesting beach in Costa Rica has seen arribadas ranging from 3,564 to 476,550 egg laying females between 2006 and 2010 (Valverde et al. 2012).

In the Hawaii Range Complex, we conservatively concluded 7,273 Pacific sea turtle will experience behavioral responses during HSTT activities. We also concluded that 327 Pacific sea turtles will experience TTS due to acoustic sources during training exercises and 78 Pacific sea turtles will experience TTS during testing activities in the Hawaii Range Complex. We anticipate that 21 Pacific sea turtles will experience PTS from acoustic sources during training exercises and 5 Pacific sea turtles will experience PTS from acoustic sources during testing activities. We anticipate that 13 Pacific sea turtles will experience slight lung injury and 4 Pacific sea turtles will die due to acoustic sources during training in the Hawaii Range Complex. We apportion Pacific sea turtle mortality in the Hawaii Range Complex due to acoustic sources based on stranding network data. Because 97 percent of sea turtles in the Main Hawaiian Islands that strand are green turtles, we expect the majority of sea turtle mortality in the Hawaii Range Complex to be green turtles (3.88/year). Three percent of sea turtles that strand in the Main Hawaiian Islands are hawksbills and olive ridleys; therefore, we expect the remaining sea turtle species that will experience mortality from acoustic exposure in the Hawaii Range Complex to be hawksbills and olive ridleys (0.12/year). Because leatherbacks and loggerheads rarely strand in Hawaii (Balazs and Chaloupka 2006b), we believe the likelihood of any individuals of these species being exposed to acoustic stressors and then dying from that exposure is so small as to be discountable.

In the SOCAL Range Complex, we concluded zero green sea turtles will experience behavioral reponses during HSTT activities. We also concluded zero green turtles will experience TTS due to acoustic sources during training and 555 green turtles will experience TTS during testing activities in the SOCAL Range Complex. We anticipate one green turtle will experience PTS from acoustic sources during training activities but no green turtles will experience PTS from acoustic sources during testing activities in the SOCAL Range Complex. We anticipate no green turtles will experience slight lung injury or die due to acoustic sources during training in the SOCAL Range Complex.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Ketten 2006; 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, b) 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. 2012b; 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 in sections 6.2.6 and 6.6.1.16, 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 testing activities conducted during the remaining period of the five-year MMPA rule and into the reasonably foreseeable future are likely to cause TTS and PTS in ESA-listed sea turtles in the HSTT Action Area at the levels described above. However, as described previously in sections 6.2.3 and 6.6.1.16, there is no evidence that TTS or PTS 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 and PTS will not result in fitness consequences to individuals or the populations to which they belong.

Acoustic stressors associated with the Navy's activities in the HSTT Action Area also have the ability to cause slight lung injury in ESA-listed sea turtles in the HSTT Action Area. As described previously in section 6.6.1.16, although slight lung injuries could temporarily affect the fitness of affected individuals by reducing their respiration rate, these effects are expected to stop once the injury has healed. Therefore, a temporary disruption of behaviors or fitness levels resulting from slight lung injury is not expected to substantially impact individual turtles when considering their overall lifetime fitness. Because we do not expect fitness consequences to individual turtles, we do not expect impacts to populations composed of those individual turtles.

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 in the Hawaii and SOCAL Range Complexes. 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 Section 6.2.7, 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 potential for population level impacts resulting from lethal exposure of sea turtles to acoustic stressors is considered below, along with lethal vessel strike.

Based on our analysis of sea turtle vessel strike, we expect one green sea turtle to be struck by the Navy each year (0.5 green turtles/year in SOCAL + 0.09 green turtles/year in Hawaii = 0.59 green turtles/year; rounded up to 1 green turtle/year) continuing into the reasonably foreseeable future. Our analysis did not indicate vessel strike of any other sea turtles in the HSTT Action Area is reasonably certain to occur. Therefore, vessel strike of all sea turtle species, with the exception of green turtles, is discountable. For the purposes of the Incidental Take Statement, strikes of green turtles from SOCAL and the Hawaii Range Complex are combined. However, for the purposes of our jeopardy determination below, strikes of green turtles in SOCAL (the endangered population from Mexico's Pacific coast breeding colony) are considered separately from strikes of green turtles in the Hawaii Range Complex (the threatened range-wide population).

Summing expected lethal take of green turtles from acoustic exposure and vessel strike, we expect four green turtles (3.88 green turtles/year from acoustics + 0.09 green turtles/year from vessel strike = 4 green turtles/year) each year to die in Hawaii (from the threatened range-wide population), and 1 green turtle to die annually in SOCAL (from the endangered population of Mexico's Pacific coast). Summing expected lethal take of green turtles over a 10-year period[10], we expect 40 green turtles from the threatened range-wide population to die and 10 green turtles from the endangered population from Mexico's Pacific coast breeding colony to die as a result of Navy training and testing activities. We expect a combined total of 0.12 hawksbill or olive ridley

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[10] We used a 10-year period to be consistent with the 10-year period used to analyze effects to marine mammals.

sea turtles to die in Hawaii each year, totaling 1.2 individuals over a 10 year period. Split evenly between the two species, this is equal to 0.06 hawksbill and 0.06 olive ridley sea turtle mortlities per year in Hawaii, or 1 (0.6 rounded up to 1) of each species over 10 years.

Death of an individual sea turtle would have a direct fitness consequence to the individual leading to lost reproductive potential that the individual might contribute to the population or sub-population. This lost reproductive potential will vary depending on the sex (male or female) and maturity of the individual. The death of a male would have less of an effect on the population than the loss of a female. Loss of a sexually mature female will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years.

As stated previously, the threatened green sea turtle population (the population that includes turtles in Hawaii) is estimated to lay 109,000 nests annually. On average green turtles nest every 2, 3 or 4 years, and lay 3.3 nests each year they breed (USFWS 2012a). Conservatively assuming that most turtles nest every 2 years, this equates to a likely total population size of approximately 66,000 females (if we assume the turtles nest every 4 years, the total population size is approximately 132,000 females). Removing 4 females from this population in one year would reduce the reproductive potential of this population by 0.006 percent. Removing 40 females from this population over 10 years would reduce the reproductive potential of this population by 0.06 percent. In either scenario, we do not consider this an appreciable reduction in the numbers of female green turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of the threatened population of green sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population.

The 1998 recovery plan for the threatened U.S. Pacific population of green sea turtles identifies disease and parasites as the only major threat occurring within the waters around Hawaii and sets criteria for the delisting of this entity. This stressor is not associated with HSTT activities. Delisting requires identifying regional stocks to source beaches, maintenance of stocks above 5,000 nesting females annually, nesting populations at source beaches are stable or increasing for 25 years, maintenance of healthy foraging areas, increases of foraging populations, completion of all priority one tasks, and the finalization of management plans and international agreements. However, we do not have sufficient information to assign sea turtles killed by Navy activities to individual nesting populations. Because of this, we considered the entirety of the listed entity, and concluded the removal of 4 green sea turtles annually, or 40 over 10 years, from acoustic stressors and vessel strike and continuing at this rate into the reasonably foreseeable future is not an appreciable reduction in the numbers or the reproductive capability of the species.

As stated previously, the endangered green turtle population (the population that includes turtles in San Diego Bay) is estimated to lay 6,050 nests annually. On average green turtles nest every 2, 3, or 4 years, and lay 3.3 nests each year they (USFWS 2012a). Conservatively assuming that most turtles nest every 2 years, this equates to a likely total population size of approximately 3,666 females (if we assume the turtles nest every 4 years, the total population size is approximately 7,333 females). Removing 1 female from this population in one year would reduce the reproductive potential of this population by 0.03 percent. Removing 10 females from this population over a 10-year period would reduce the reproductive potential of this population by 0.27 percent. In either scenario, we do not consider this to be an appreciable reduction in the numbers of female green turtles or the reproductive rate of the population due to the reproductive potential of sea turtles (e.g., large clutch size, nesting frequency), either on an annual basis or continuing into the reasonably foreseeable future. We conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the threatened population of green sea turtles and this level of mortality will not impact the survival or recovery of this population.

The 1998 recovery plan for the endangered east Pacific population of green sea turtles identifies entanglement and ingestion of debris and vessel collisions as the only major threats occurring within the waters of the U.S. west coast. Of these, only vessel collision is associated with HSTT activities. The recovery plan also sets criteria for the delisting of this entity. Delisting requires identifying regional stocks to source beaches, maintenance of stocks above 5,000 nesting females annually, nesting populations at source beaches are stable or increasing for 25 years, maintenance of healthy foraging areas, increases of foraging populations, completion of all priority one tasks, and the finalization of management plans and international agreements. However, we do not have sufficient information to assign sea turtles killed by Navy activities to individual nesting populations. Because of this, we considered the entirety of the listed entity, and concluded the removal of 1 green sea turtle annually, or 10 over 10 years, from vessel strike is not an appreciable reduction in its numbers or its reproductive capability.

As stated previously, an average of between 11,000 and 12,700 hawksbill nests are estimated to occur each year in the Pacific. On average hawksbill turtles nest every 2 or 3 years, and lay 4.5 nests each year they (USFWS 2012b). Conservatively assuming that most turtles nest every 2 years and assuming the lower estimate of the number of nests annually, this equates to a likely total population size of approximately 4,889 females (if we assume the turtles nest every 3 years, the total population size is approximately 9,778 females). Removing 1 female from this population in 10 years (0.6 turtles in 10 years, rounded to 1 individual) would reduce the reproductive potential of this population by 0.02 percent. We do not consider this to be an appreciable reduction in the numbers of female hawksbill turtles or the reproductive rate of the population, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of this population of hawksbill sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population.

The 1998 recovery plan for the U.S. Pacific populations of hawksbill sea turtles identifies increased human presence (primarily thorugh impacts to nesting beaches) and nest predation as the only major threats occurring in Hawaii and sets criteria for the delisting the species. HSTT activities are not expected to exacerbate these threats. Delisting requires identifying regional stocks to source beaches; maintenance of stocks above 1,000 nesting females annually; nesting populations at source beaches are stable or increasing for 25 years; maintenance of healthy foraging areas; increases of foraging populations; completion of all priority one tasks; and the finalization of management plans; formal cooperative relationships with revional sea turtle management programs; and international agreements. However, we do not have sufficient information to assign sea turtles killed by Navy activities to individual nesting populations. Because of this, we considered the entire Pacific Ocean population (which is a component of the rangewide, circumglobal, listed entity), and concluded the removal of 1 hawksbill sea turtle over 10 years, from acoustic stressors, continuing into the reasonably foreseeable future, is not an appreciable reduction in its numbers or its reproductive capability.

The olive ridley is thought to be the most abundant sea turtle in the world. For example, as stated previously, the Ostional nesting beach in Costa Rica saw arribadas ranging between 3,564 and 476,550 egg laying females between 2006 and 2010 (Valverde et al 2012). The individuals nesting at this location represent a subset of the worldwide threatened population (all individuals that do not nest on the Pacific coast of Mexico). Considering only the number of egg laying females at this one beach, removing 1 female from this population in 10 year (0.6 turtles in 10 years, rounded to 1 individual) would reduce the reproductive potential of this population by 0.03 percent. We do not consider this to be an appreciable reduction in the numbers of female olive ridley turtles or the reproductive rate of individuals from this nesting location, either on an annual basis or continuing into the reasonably foreseeable future. The impact of this removal on the threatened worldwide population would be even smaller. We conclude that this level of mortality is not an appreciable reduction in the numbers or reproductive rate of the threatened population of olive ridely sea turtles, either on an annual basis or continuing into the reasonably foreseeable future. Because we do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of olive ridley sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population. The same argument can be made for the endangered population of olive ridleys nesting off the Pacific coast of Mexico since recent at-sea abundance estimates for this population have totaled over one million individuals.

The 1998 recovery plan for the U.S. Pacific populations of olive ridley sea turtles identifies no major threats occurring in Hawaii and sets criteria for the delisting the species. Delisting requires identifying regional stocks to source beaches, stability in the number of nesting females over 10 years, increases of foraging populations, and the finalization of management plans and international agreements. However, we do not have sufficient information to assign sea turtles killed by Navy activities to individual nesting populations. Because of this, we considered just a single nesting beach population (which is a component of the rangewide, circumglobal, listed entity), and concluded the removal of 1 olive ridley sea turtle over 10 years, from acoustic stressors, continuing into the reasonably foreseeable future, is not an appreciable reduction in its numbers or its reproductive capability.

The 1998 recovery plan for the U.S. Pacific populations of leatherback sea turtles identifies fisheries bycatch as the only major threat occurring in Hawaii and 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. We do not have sufficient information to assign leatherback a specific number of takes from Navy acoustic stressors; however, any leatherbacks affected 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.

The North Pacific DPS of loggerhead sea turtles does not have a recovery plan; therefore no recovery criteria have been established, nor have major threats and their remedies specific to loggerhead recovery been established. We do not have sufficient information to assign North Pacific loggerheads a specific number of takes from Navy acoustic stressors; however, any loggerheads affected 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 supporting information from a recovery plan suggesting temporary effects from acoustic stressors would impede loggerhead recovery, we do not expect sub-lethal loggerehead sea turtle take to impede recovery of this species.

Based on our analysis in this Opinion, we conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of ESA-listed sea turtles in the wild by reducing the reproduction, numbers, or distribution of these species. We also conclude that effects from ongoing U.S. Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of the recovery of ESA-listed sea turtles in the wild by reducing the reproduction, numbers, or distribution of these species. That is, non-impulsive acoustic stressors will not have fitness consequences at the individual level. Some impulsive acoustic stressors and vessel strike will kill small numbers of turtles. However, neither the lethal or sub-lethal effects of

Navy activities will result in an appreciable reduction in reproductive capability at the population or range-wide level. 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 CONCLUSION

During the consultation, we reviewed the current status of endangered blue, fin, western north Pacific gray, humpback, sei, sperm, north Pacific right, and MHI insular false killer whales, endangered Hawaiian monk seals, threatened Gualalupe fur seals, endangered green, hawksbill, loggerhead, olive ridley, and leatherback turtles, endangered southern California steelhead trout, black abalone, and white abalone. We also reviewed the current status of the Eastern Pacific DPS of scalloped hammerhead shark that was recently listed under the ESA.

Our consultation evaluated the effects on the ESA-listed species listed above in the context of the current *Environmental Baseline* and *Cumulative Effects* of U.S. Navy military readiness activities (training exercises and testing activities) in the Hawaii-SOCAL Training and Testing Action Area for the remainder of the five-year period (December 2013 through December 2018) and continuing into the reasonably foreseeable future, the National Marine Fisheries Service's Permit and Conservation Division's promulgation of a rule under the Marine Mammal Protection Act (MMPA) on the taking of marine mammals incidental to the Navy's proposed military readiness activities, and the subsequent issuance of two letters of authorization pursuant to the MMPA rule.

It is NMFS' opinion that the U.S. Navy's military readiness activities in the Hawaii-SOCAL Training and Testing Action Area, the Permits Division's promulgation of the MMPA rule and subsequent issuance of LOAs pursuant to that rule, are likely to adversely affect, but would not appreciably reduce the likelihood of survival or recovery of endangered blue, fin, western North Pacific gray, humpback, sei, sperm, or MHI insular false killer whales, Hawaiian monk seals, threatened Guadalupe fur seals, endangered green, hawksbill, loggerhead, olive ridley, and leatherback turtles and therefore we conclude would not jeopardize these species.

We find that these same proposed actions are not likely to adversely affect endangered Southern California steelhead trout, endangered black abalone, white abalone, north Pacific right whales, or the Eastern Pacific DPS of scalloped hammerhead shark.

We also find that the actions are not likely to result in the destruction or adverse modification of critical habitat that has been designated for endangered or threatened species in the HSTT Action Area.

We believe these conclusions are valid for the remainder of the five-year period of the MMPA rule and continuing into the reasonably forseeable future beyond the five-year period, assuming that the type, amount and extent of training exercises and testing activities do not exceed levels assessed in this biological and conference opinion and the status of the species affected by these actions does not change substantially from what was assessed in this Opinion.

This opinion also concludes that the NMFS' issuance of the rule and two letters of authorization (LOAs) pursuant to the MMPA rule as assessed in this Opinion for respective training and testing activities to take marine mammals through December 2018, incidental to the U.S. Navy's testing and training exercises are likely to adversely affect ESA-listed species. The levels of take associated with acoustic stressors to ESA-listed marine mammals assessed in this Opinion are identical to those assessed in the MMPA rule. However, take associated with mortality of ESA-listed whale species resulting from vessel strike is lower in this Opinion than in the MMPA rule. This Opinion analyzed vessel strikes of ESA-listed whales that reasonably would be expected based on historical Navy strike data and relative whale abundances (see section 6.6.2). In contrast, the number of whale strikes published in the MMPA rule represents an improbable worst-case scenario that substantially exceeds actual strike rates and is not reasonably expected to occur. The ESA requires that take of marine mammals specified in an incidental take statement be authorized under the MMPA. Because take of marine mammals from acoustic stressors assessed in this Opinion is identical to the MMPA rule and LOAs, and take from vessel strikes assessed in this Opinion is equal to or below that authorized in the MMPA rule and LOAs, all take of marine mammals specified in this Opinion and incidental take statement has also been authorized under the MMPA[11].

8 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. 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. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the Marine Mammal Protection Act, there is a definition of what is referred to as Level B harassment: "any act of pursuit, torment, or annoyance which... has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering but which does not have the potential to injure a marine mammal or marine mammal stock in the wild" 16 U.S.C. §1362(18)(A)(ii). For this consultation, we interpret

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[11] *In the litigation challenging the MMPA rule and LOAs, the court entered an order on March 31, 2015, finding that NMFS's negligible impact and least practicable adverse impact determinations were arbitrary and capricious. Under the court's scheduling order, appropriate relief is to be addressed in future proceedings. In the event the MMPA rule and LOAs are vacated, the provisions of the Incidental Take Statement pertaining to listed marine mammals would become operative only upon issuance of new MMPA take authorization or if Navy testing and training activities in the HSTT Action Area are exempted from the requirements of the MMPA pursuant to 16 USC 1371(f).*

"harassment" using the MMPA training exercises and testing definition of harassment to marine mammals. For other species, specifically sea turtles, we apply "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. 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 and, when marine mammals are involved, the take is authorized under the MMPA.

8.1 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" may be used if we cannot assign numerical limits for animals that would be incidentally taken during the course of an action (51 FR 19953). The amount of take resulting from the Navy's activities and evaluated in this Opinion was estimated based on the best information available.

Here we summarize the anticipated take from annual and non-annual training exercises by species as proposed by the Navy in the HSTT Action Area and the interdependent actions of issuance of a five-year regulation, and LOAs by NMFS' Permits Division to authorize take of marine mammals pursuant to the MMPA. Below we provide numeric estimates for each species for which we could develop such estimates.

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[12]. The levels of take associated with acoustic stressors to ESA-listed marine mammals specified in this incidental take statement are identical to those assessed in the current MMPA rule (December 24, 2013 through December 24, 2018) and LOAs. However, take associated with mortality of ESA-listed whale species resulting from vessel strike is lower in this incidental take statement than in the current MMPA rule. The Opinion analyzed vessel strikes of ESA-listed whales that reasonably would be expected based on historical Navy strike data and relative whale abundances (see section 6.6.2). In contrast, the number of whale strikes published in the MMPA rule represents an improbable worst-case scenario that substantially exceeds actual strike rates and is not reasonably expected to occur. Because take of marine mammals from acoustic stressors specified

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[12] *In the litigation challenging the MMPA rule and LOAs, the court entered an order on March 31, 2015, finding that NMFS's negligible impact and least practicable adverse impact determinations were arbitrary and capricious. Under the court's scheduling order, appropriate relief is to be addressed in future proceedings. In the event the MMPA rule and LOAs are vacated, the provisions of the Incidental Take Statement pertaining to listed marine mammals would become operative only upon issuance of new MMPA take authorization or if Navy testing and training activities in the HSTT Action Area are exempted from the requirements of the MMPA pursuant to 16 USC 1371(f).*

in this incidental take statement is identical to the MMPA rule and LOAs, and take from vessel strikes specified in this incidental take statement is equal to or below that authorized in the MMPA rule and LOAs, all take of marine mammals specified in this incidental take statement has also been authorized under the MMPA.

For marine mammal take from acoustic stressors, the total five-year take referenced below is for the five-year (December 24, 2013 through December 24, 2018) period of the current MMPA Rule and LOAs. For marine mammal take from ship strikes, we specify take numbers for the remainder of the five-year period of the current MMPA Rule and LOAs. Any take from vessel strikes in excess of the levels specified in this incidental take statement will require reinitiation of consultation.

8.1.1 **Training**

The anticipated take incidental to annual and non-annual training exercises in a given year where all possible activities are carried out in that year is provided below. For marine mammals, we provide the annual take and take from non-annual training (not occurring every year) from acoustic stressors and a total for the five-year period (December 24, 2013 through December 24, 2018) in Table 78. As such, take from non-annual activities would be less than the sum of the total of each year given a scenario where all annual and all non-annual activities take place in that year. For sea turtles only annual take is provided.

Table 78. Take of ESA-listed marine mammals incidental to training activities, the Marine Mammal Protection Act Regulation, and issuance of the Letter of Authorization pursuant to the MMPA regulations.

ESA-Listed Species	Training Activities				
	Acoustic Stressors				Vessel Strike
	Harass (Behavioral & Temporary Threshold Shift)	Harm (Permanent Threshold Shift)	Harm (GI Tract, Slight Lung Injury, Other)	Mortality	Injury or Mortality
Cetaceans					
Fin Whale	Up to 1719 per year; Not to exceed 8,531 total in 5 years	0	0	0	Up to two ESA-listed whales during the remainder of the five-year period of the MMPA rule not to exceed
Blue Whale	Up to 4,325 per year; Not to exceed 21,559 total in 5 years	0	0	0	
Humpback Whale	Up to 9,273 per year; Not to exceed 46,365 total in 5 years	0	0	0	
Sei Whale	Up to 630 per year; Not to exceed 2,996 total in 5 years	0	0	0	

Sperm Whale	Up to 3,332 per year; Not to exceed 15,920 total in 5 years	0	0	0	one of each species[1,4]
Western North Pacific Gray Whale	Up to 10 per year; Not to exceed 50 total in 5 years	0	0	0	0
Main Hawaiian Island Insular False Killer Whale	Up to 49 per year; Not to exceed 220 total in 5 years	0	0	0	0

ESA-Listed Species	**Training Activities**				
	Acoustic Stressors				**Vessel Strike**
	Harass (Behavioral & Temporary Threshold Shift)	**Harm (Permanent Threshold Shift)**	**Harm (GI Tract, Slight Lung Injury, Other)**	**Mortality**	**Injury or Mortality**
Pinnipeds					
Hawaiian Monk Seal	Up to 1,292 per year; Not to exceed 6,334 total in 5 years	0	0	0	0
Guadalupe Fur Seal	Up to 2,603 per year; Not to exceed 13,015 total in 5 years	0	0	0	0
Sea Turtles					
Green Sea Turtle[2]	0[3]	0	0	0	1 / yr
Pacific Sea Turtles[3]	Behavioral: 7,144 / yr TTS: 327 / yr	21 / yr	13 / yr	4 / yr	0

[1] The Navy cannot always determine the species of whale that is struck. Incidental take of large whales from vessel strike will be exceeded if more than two ESA-listed large whale species , not to exceed one of each species, are struck during the remainder of the five-year period of the MMPA Rule. If the Navy cannot determine the identity of a large whale species struck by a vessel, the agencies shall assume for purposes of this ITS that the species is a listed species. Within four weeks of the strike, NMFS and the Navy shall confer to determine the listed species most likely involved based on the best available information.

[2] Green turtle take in the SOCAL Range as of the HSTT Action Area

[3] The Pacific sea turtle category including green, hawksbill, loggerhead, olive ridley, and leatherback turtles in the Hawaii Range Complex and Transit Corridor areas of the HSTT Action Area

[4] Vessel Strike of cetaceans, pinnipeds, and sea turtles was not assessed separately for testing activities.

8.1.2 Testing

The anticipated take incidental to annual testing activities and for the five-year period is provided in Table 79.

Table 79. Take Authorized Incidental to Annual Testing Activities, Issuance of the Marine Mammal Protection Act Regulation, and Issuance of the Letter of Authorization pursuant to those MMPA regulations.

ESA-Listed Species	Testing Activities				
	Acoustic Stressors				Vessel Strike
	Harass (Behavioral & Temporary Threshold Shift)	Harm (Permanent Threshold Shift)	Harm (GI Tract, Slight Lung Injury, Other)	Mortality	Injury or Mortality
Cetaceans					
Blue Whale	Up to 428 per year; Not to exceed 2,140 total in 5 years	0	0	0	Vessel Strike of cetaceans, pinnipeds, and sea turtles was not assessed separately for testing activities. Refer to Table 78
Fin Whale	Up to 225 per year; Not to exceed 1,125 total in 5 years	0	0	0	
Humpback Whale	Up to 927 per year; Not to exceed 4,635 total in 5 years	0	0	0	
Sei Whale	Up to 51 per year; Not to exceed 255 total in 5 years	0	0	0	
Western North Pacific Gray Whale	Up to 2 per year; Not to exceed 10 total in 5 years	0	0	0	
Sperm Whale	Up to 263 per year; Not to exceed 1,315 total in 5 years	0	0	0	
Main Hawaiian Island Insular False Killer Whale	Up to 4 per year; Not to exceed 20 total in 5 years	0	0	0	
Pinnipeds					
Hawaiian Monk Seal	Up to 358 per year; Not to exceed 1,790 total in 5 years	0	0	0	
Guadalupe Fur Seal	Up to 269 per year; Not to exceed 1,345 total in 5 years	0	0	0	

ESA-Listed Species	**Testing Activities**				
	Acoustic Stressors				**Vessel Strike**
	Harass (Behavioral & Temporary Threshold Shift)	**Harm (Permanent Threshold Shift)**	**Harm (GI Tract, Slight Lung Injury, Other)**	**Mortality**	**Injury or Mortality**
Sea Turtles					
Green Sea Turtle[1]	555[3] / yr	1 / yr	0	0	See Table 78
Pacific Sea Turtles[2]	Behvaioral: 129 / yr TTS: 78 / yr	5 / yr	0	0	

[1] **Green turtle take in the SOCAL Range as of the HSTT Action Area**

[2] **The Pacific sea turtle category including green, hawksbill, loggerhead, olive ridley, and leatherback turtles in the Hawaii Range Complex and Transit Cooridor areas of the HSTT Action Area**

8.2 Effects of the Take

In this biological 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.

8.3 Reasonable and Prudent Measures

The measures described below are non-discretionary, must be undertaken by NMFS Permits and Conservation Division and the U.S. Navy so they become binding conditions of any permit or Letter of Authorization issued to the U.S. Navy, in order for the exemption in section 7(o)(2) to apply. The NMFS Permits and Conservation Division and the U.S. Navy have a continuing duty to regulate the activity covered by this Incidental Take Statement. If NMFS Permits and Conservation Division (1) fails to require the U.S. Navy to adhere to the Terms and Conditions of the Incidental Take Statement through enforceable terms that are added to the permit or grant document, and/or (2) fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, NMFS Permits and Conservation Division and the U.S. Navy must report the progress of the action and its impact on the species to NMFS ESA Interagency Cooperation division as specified in the incidental take statement (50 CFR §402.14(i)(3)). NMFS believes the following reasonable and prudent measure is necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

8.3.1 **Monitoring**

As discussed in Section 8.1 of 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 HSTT 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.

The Navy is required under the final MMPA rule and the reasonable and prudent measures in this Opinion to report activity levels (including sonar hours and the type and number of explosives used) to NMFS. As such, we established a term and condition of this Incidental Take Statement, requiring the Navy to report to NMFS any exceedance of activity levels or planned testing or training events specified above 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.

The take levels specified above also include four annual sea turtle mortalities from explosives (based on Navy modeling) and one annual sea turtle mortality resulting from a vessel strike. As discussed in the preceding Opinion, in contrast to vessel strikes involving ESA-listed cetaceans, it is very difficult to detect the occurrence of a vessel strike involving a sea turtle. Even with regional stranding network data, it is exceedingly difficult to attribute injury or mortality to Navy training or testing activities given multiple use of harbors, coastal zones, and offshore areas by Navy and non-Navy entities. Although we believe that Navy monitoring may detect some sea turtle mortalities resulting from the use of explosives, most mortalities likely go undetected. Therefore, for these forms of take as well, we must again rely on estimated take associated with levels of activities and any opportunistic observations of sea turtle mortalities during or following testing or training activities as measurements of take and a trigger for reinitiation of consultation. We are aware of no other feasible or available alternative means of determining when estimated levels of these forms of take are exceeded.

Included as a term and condition of this incidental statement is the requirement that the Navy immediately (or as soon as operational security considerations allow) notify NMFS if a dead or seriously injured sea turtle is observed during Navy testing or training activities that is potentially attributable to these activities including but not limited to the use of explosives and vessel strike. If the Navy observes one or more dead or seriously injured sea turtles in each of two consecutive years during Navy testing and training activities that is potentially attributable to these activities, the Navy shall contact NMFS to determine if reinitiation is required. Additionally, the Navy shall annually request sea turtle stranding data from the regional NMFS stranding office and will review any data provided by the regional office to determine if any unusual stranding events or increases in stranding rates may be attributed to Navy training and testing. If the Navy identifies an increase in sea turtle stranding data potentially associated with Navy training and testing activities, the Navy shall contact NMFS to determine if reinitiation is required.

8.3.2 Reporting

The U.S. 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 HSTT 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.

8.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, as amended, NMFS’ Permits and Conservation Division and the U.S. Navy must comply with the following terms and conditions, which implement the reasonable and prudent measure described above and outline the reporting requirements required by the section 7 regulations (50 CFR 402.14(i)). The terms and conditions described below are non-discretionary, and NMFS Permits and Conservation Division and the U.S. Navy must comply with them in order to implement the reasonable and prudent measures (50 CFR 402.14). NMFS Permits and Conservation Division and the U.S. Navy has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this incidental take statement (50 CFR 402.14). If the following terms and conditions are not complied with, the protective coverage of section 7(o)(2) will likely lapse.

1. The U.S. Navy shall implement all mitigation and monitoring measures as proposed in their 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.

2. The U.S. Navy shall immediately (or as soon as operational security considerations allow) notify NMFS if a dead or seriously injured sea turtle is observed during Navy testing or training activities that is potentially attributable to these activities, including but not limited to the use of explosives and vessel strike. If the Navy observes one or more dead or seriously injured sea turtles in each of two consecutive years during Navy testing and training activities that is potentially attributable to these activities, the Navy shall contact NMFS to determine if reinitiation is required.

3. NMFS’ Permits Division shall ensure that all mitigation and monitoring measures as prescribed by the final rule and LOAs, and as described in this Opinion, are implemented by the U.S. Navy.

4. The U.S. Navy and NMFS’ Permits Division shall compile and summarize annual monitoring and exercise reports and describe interactions with ESA-listed species and designated critical habitat.

5. If the U.S. Navy cannot determine the identity of a large whale species struck by a vessel, the Navy and NMFS shall assume for purposes of this ITS that the species is an ESA-listed species. Within four weeks of a strike, NMFS and the Navy shall confer to determine which species is most likely involved based on the best available information.

6. The U.S. Navy must report to NMFS any exceedance of activity levels or planned testing or training events specified in the preceding Opinion and in the final MMPA rule before the exceedance occurs (if operational security considerations allow), or as soon as operational security considerations allow after the relevant activity is conducted.

7. The Navy shall annually request sea turtle stranding data from the regional NMFS stranding office and will review any data provided by the regional office to determine if any unusual stranding events or increases in stranding rates may be attributed to Navy training and testing. If the Navy identifies an increase in sea turtle stranding data potentially associated with Navy training and testing activities, the Navy shall contact NMFS to determine if reinitiation is required.

9 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a action on listed species or critical habitat, to help implement recovery plans, or to develop information.

We recommend the following actions to further minimize or avoid adverse effects to ESA-listed species:

1. The Navy should coordinate with NMFS to improve sea turtle abundance and density estimates for the HSTT Action Area.

2. The Navy should explore methods to better quantify behavioral take of sea turtles not rising to the level of TTS.

3. The Navy should explore methods to better quantify the risk of vessel strike to sea turtles.

10 REINITIATION OF CONSULTATION

This concludes this reinitiated formal consultation on the U.S. Navy’s training and testing activities in the HSTT Action Area during the remaining period of the MMPA Rule (December 2013 through December 2018) and NMFS Permits and Conservation Division’s issuance of regulations pursuant to the Marine Mammal Protection Act that allow for the issuance of one or more letters of authorization to authorize the “take” of marine mammals in association with those activities, and NMFS Permits and Conservation Division issuance of two letters of authorization to the U.S. Navy pursuant to the MMPA regulations. 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.

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