NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL OPINION

Title: Biological Opinion on (1) U.S. Military Mariana Islands Training and Testing Activities; and (2) the National Marine Fisheries Service's promulgation of regulations and issuance of a letter of authorization pursuant to the Marine Mammal Protection Act for the U.S. Navy to "take" marine mammals incidental to Mariana Islands Training and Testing activities from August 2015 through August 2020

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce

Action Agencies: United States Navy (Navy) and NOAA's National Marine Fisheries Service, Office of Protected Resources Permits and Conservation Division

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Approved: Donna S Wieting

Donna S. Wieting
Director, Office of Protected Resources

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1 INTRODUCTION

The Endangered Species Act of 1973, as amended (ESA; 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 insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Federal agencies must do so in consultation with National Marine Fisheries Service (NMFS) for threatened or endangered species (ESA-listed), or designated critical habitat that may be affected by the action that are under NMFS jurisdiction (50 C.F.R. §402.14(a)). If a Federal action agency determines that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS concurs with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If an incidental take is expected, section 7(b)(4) requires NMFS to provide an incidental take statement that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts and terms and conditions to implement the reasonable and prudent measures.

The action agencies for this consultation are the U.S. Department of Defense (DoD)[1] and U.S. Coast Guard (USCG) (since the Navy is the lead action agency for this consultation, these agencies are collectively referred to as the "Navy" throughout this opinion), which undertakes military training and testing activities and NMFS's Office of Protected Resources, Permits and Conservation Division, which (1) promulgated regulations under the Marine Mammal Protection Act (MMPA) governing the U.S. Navy's "take" of marine mammals incidental to those military readiness activities which are in effect from August 2015 through August 2020 and (2) issued a Letter of Authorization (LOA) pursuant to the regulations that authorizes the U.S. Navy to "take" marine mammals incidental to those military readiness activities through August 2020.

This consultation, biological opinion, and incidental take statement, were completed in accordance with section 7(a)(2) of the statute (16 U.S.C. 1536 (a)(2)), associated implementing regulations (50 C.F.R. §§401-16), and agency policy and guidance was conducted by NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division

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[1] The Navy is the executive agent for Mariana Islands Training and Testing (MITT) activites which include all Navy, US Air Force (USAF), and US Coast Guard (USCG) activities as outlined in the MITT Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS). This Biological Opinion (BO) supports Navy, USAF, and USCG actions.

(hereafter referred to as “we”).This biological opinion (opinion) and incidental take statement were prepared by NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 C.F.R. §402.

This document represents NMFS’ opinion on the effects of these actions on endangered and threatened species and critical habitat that has been designated for those species. A complete record of this consultation is on file at NMFS’ Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The Navy has been conducting training and testing activities in the general geographic area of the Mariana Islands Training and Testing (MITT) Study Area for several decades. On June 12, 2015, NMFS issued a final biological opinion and conference report on the Navy’s proposed action to conduct Mariana Islands Training and Testing activities and the National Marine Fisheries Service’s promulgation of regulations and issuance of a letter of authorization pursuant to the Marine Mammal Protection Act for the U.S. Navy to “take” marine mammals incidental to Mariana Islands Training and Testing activities from August 2015 through August 2020. This opinion was based on information provided in the March 2014 U.S. Navy package requesting ESA consultation which included the September 2013 Mariana Islands Training and Testing Draft Environmental Impact Statement/Overseas Environmental Impact Statement (DEIS/OEIS); the Biological Evaluation; the Mariana Islands Training and Testing Endangered Species Act Section 7 Consultation Supplemental Information; NMFS Permits Division’s January 28, 2015, request for Section 7 consultation under the ESA; and the proposed Federal regulations under the MMPA specific to the proposed activities (78 FR 7050). We also considered later versions of the Final EIS/OEIS and other supplemental information as they were made available to us. Also considered were the 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.

This is a revised version of the 2015 biological opinion. Revisions to the 2015 opinion were required to address the following: 1) analysis of impacts to green sea turtles in consideration of the final rule, issued in 2016, to list 11 DPSs of green sea turtles as threatened or endangered under the ESA (81 FR 20057); 2) analysis of humpback whales in consideration of the final rule, issued in 2016, to divide the globally-listed humpback whale into 14 DPSs and list four DPSs as endangered and one as threatened (81 FR 62259); and 3) new scientific information provided by the Navy on coral coverage at Farallon de Medinilla. This opinion supersedes the 2015 biological opinion.

1.2 Consultation History

- On April 28, 2016, the Navy requested NMFS confirm the conference report on the effects of Navy training and testing activties on proposed DPSs for green sea turtles in the MITT Study Area as a final biological opinion.

- In May 2016, NMFS informed the Navy that revisions to the biological opinion would be required in order to address impacts to green sea turtles from multiple DPSs.

- On June 24, 2016, NMFS provided a revised draft biological opinion to the Navy. NMFS received comments from the Navy on the draft opinion on July 14, 2016.

- On September 8, 2016 NMFS issued a final rule to divide the globally-listed humpback whale in 14 DPSs and list four DPSs as endangered and one as threatened.

- On October 18, 2016, the Navy provided NMFS with new scientific information on coral coverage at Farallon de Medinilla.

- On February 10, 2017, NMFS provided a revised draft biological opinion to the Navy. NMFS received comments from the Navy on the draft opinion on March 6, 2017.

- On May 26, 2017, NMFS requested additional information from the Navy on the nature of ordnance use at Farallon de Medinilla. On July 27, 2017 Navy provided NMFS the requested information.

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 the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration.

This opinion addresses three interdependent actions conducted by the U.S. Navy and NMFS's Permits Division: (1) the U.S. Navy's military training and testing activities (i.e., readiness activities) conducted in the MITT Study Area; (2) NMFS's Permits Division's promulgation of regulations pursuant to the MMPA governing the U.S. Navy's "take" of marine mammals incidental to the Navy's military readiness activities from August 2015 through August 2020; and (3) NMFS's Permits Division's issuance of an LOA pursuant to the regulations that authorize the U.S. Navy to "take" marine mammals incidental to military readiness activities in the MITT Study Area through August 2020.

The purpose of the activities the U.S. Navy conducts in the MITT 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 LOA is to allow the U.S. Navy to "take" marine mammals incidental to military readiness activities in the MITT Study Area conducted through August 2020 in a manner that is consistent with the requirements of the MMPA and implementing regulations.

NMFS recognizes that while Navy training and testing requirements change over time in response to global or geopolitical events and other factors, the general types of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assumed that the activities conducted 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 2015 MITT FEIS/OEIS and MMPA rule.

The tempo of training within the MITT Study Area is subject to variation within the scope of the activities described in the Navy's MITT FEIS/OEIS and this opinion. Annual variation in the number of training events and quantities of authorized sonar systems and explosive training could occur. 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.

Notwithstanding this analysis, however, NMFS would fully take into account all of the best available science and any change in the status of the species when and if the Navy applies for a new MMPA incidental take authorization upon expiration of the five-year regulations considered in this opinion. The Navy would also need to initiate a new ESA consultation at that time.

The Navy, U.S. Air Force, U.S. Marine Corps, and U.S. Coast Guard routinely train in the action area in preparation for national defense missions. Typical training and testing activities and exercises covered in this opinion are briefly described in the following sections. Each military training activity described meets a requirement that can be traced ultimately to requirements set forth by the National Command Authority.

The Navy and other services have been conducting military readiness activities in the action area for decades. The tempo and types of training and testing activities have fluctuated because of the introduction of new technologies, the evolving nature of international events, advances in warfighting doctrine and procedures, and changes in force structure (organization of ships, weapons, and military personnel). Such developments influence changes in the frequency, duration, intensity, and location of required training and testing activities. The Navy categorizes training and testing activities into functional warfare areas called primary mission areas. Most training and testing activities analyzed in this opinion fall into the following eight primary mission areas:

- Anti-air warfare
- Strike warfare
- Anti-submarine warfare
- Mine warfare
- Amphibious warfare
- Anti-surface warfare
- Electronic warfare
- Naval special warfare

Not all activities can be categorized in one of these areas. The research and acquisition community (i.e., testing community) also categorizes some, but not all, of its testing activities under these primary mission areas. Testing activities analyzed in this opinion are categorized into the following areas:

- Life Cycle Activities
- Shipboard Protection Systems and Swimmer Defense Testing
- New Ship Construction
- Naval Research

U.S. Navy training and testing activities and annual activity levels are summarized in this opinion. Specific details regarding each mission area can be found in the *Mariana Islands Training and Testing Final Environmental Impact Statement/Overseas Environmental Impact Statement (FEIS/OEIS),* May 2015.

Also, on July 31, 2015, NMFS' Permits Division issued five-year regulations and an LOA to the U.S. Navy, pursuant to section 101(a)(5)(A) of the Marine Mammal Protection Act of 1972, as amended (MMPA; 16 U.S.C. 1361 *et seq*.), for taking marine mammals incidental to conducting training and testing activities in the MITT study area. The MMPA regulations are effective from August 2015 to August 2020.

2.1 Mariana Islands Training Activities

The following sections describe the training activities occurring in the MITT study area.

2.1.1 Anti-Air Warfare

The mission of anti-air warfare (AAW) is to destroy or reduce enemy air and missile threats (including unmanned airborne threats) and serves two purposes: to protect U.S. forces from attacks from the air and to gain air superiority. Anti-air warfare also includes providing U.S. forces with adequate attack warnings, while denying hostile forces the ability to gather intelligence about U.S. forces. Table 1 provides summaries of training activities in support of AAW.

Aircraft conduct anti-air warfare through radar search, detection, identification, and engagement of airborne threats-generally by firing anti-air missiles or cannon fire. Surface ships conduct anti-air warfare through an array of modern AAW systems such as aircraft detecting radar, naval guns linked to radar-directed fire-control systems, surface-to-air missile systems, and radar-controlled cannons for close-in point defense.

Table 1. Typical Anti-Air Warfare Training Exercises

Activity Name	Activity Description
Anti-Air Warfare	
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A]) – Large-caliber	Surface ship crews defend against threat aircraft or missiles with guns.
Gunnery Exercise (Surface-to-Air) (GUNEX [S-A]) – Medium-caliber	Surface ship crews defend against threat aircraft or missiles with guns.
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	Surface ship crews defend against threat missiles and aircraft with missiles.

2.1.2 Amphibious Warfare

The mission of amphibious warfare (AMW) is to project military power from the sea to the shore through the use of naval firepower and Marine Corps landing forces. It is used to attack a threat located on land by a military force embarked on ships. Amphibious warfare operations include small unit reconnaissance or raid missions to large-scale amphibious operations involving multiple ships and aircraft combined into a strike group. Table 2 provides summaries of training activities in support of AMW.

Amphibious warfare training ranges from individual, crew, and small unit events to large task force exercises. Individual and crew training include amphibious vehicles and naval gunfire support training. Small-unit training operations include shore assaults, boat raids, airfield or port seizures, and reconnaissance. Large-scale amphibious exercises involve ship-to-shore maneuver, naval fire support, such as shore bombardment, and air strike and close air support training.

Table 2. Typical Amphibious Warfare Training Exercises

Activity Name	Activity Description
Amphibious Warfare	
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.
Amphibious Rehearsal, No Landing	Amphibious shipping, landing craft, and elements of the Marine Air Ground Task Force rehearse amphibious landing operations without conducting an actual landing on shore.
Amphibious Assault	Forces move ashore from ships at sea for the immediate execution of inland objectives.
Amphibious Raid	Small unit forces move swiftly from ships at sea for a specific short-term mission. Raids are quick operations with as few Marines as possible.
Unmanned Aerial Vehicles Ops (UAV OPS)	Military units employ unmanned aerial vehicles to launch, operate, and gather intelligence for specified amphibious missions.

2.1.3 Strike Warfare

The mission of strike warfare (STW) is to conduct offensive attacks on land-based targets, such as refineries, power plants, bridges, major roadways, and ground forces to reduce the enemy's ability to wage war. Strike warfare employs weapons by manned and unmanned air, surface, submarine, and naval special warfare assets in support of extending dominance over enemy territory (power projection). Table 3 provides summaries of training activities in support of STW.

Strike warfare includes training of fixed wing attack aircraft pilots and aircrews in the delivery of precision-guided munitions, non-guided munitions, rockets, and other ordnance, including the high-speed anti-radiation missile, against land-based targets in all conditions. Not all strike mission training events involve dropping ordnance and instead the event is simulated with video footage obtained by onboard sensors.

Table 3. Typical Strike Warfare Training Exercises

Activity Name	Activity Description
Strike Warfare	
Combat Search and Rescue (CSAR)	CSAR units use helicopters, night vision and identification systems, and insertion and extraction techniques under hostile conditions to locate, rescue, and extract personnel.

2.1.4 Anti-Surface Warfare

The mission of anti-surface warfare (ASUW) is to defend against enemy ships or boats. In the conduct of anti-surface warfare, aircraft use cannons, air-launched cruise missiles or other precision guided munitions; ships employ torpedoes, naval guns, and surface-to-surface missiles; and submarines attack surface ships using torpedoes or submarine-launched, anti-ship cruise missiles. Table 4 provides summaries of training activities in support of ASUW.

Anti-surface warfare training includes surface-to-surface gunnery and missile exercises, air-to-surface gunnery and missile exercises, and submarine missile or torpedo launch events.

Table 4. Typical Anti-Surface Warfare Training Exercises

Activity Name	Activity Description
Anti-Surface Warfare	
Gunnery Exercise (Air-to-Surface) – Small-caliber	Fixed-wing and helicopter aircrews, including embarked personnel, use small-caliber guns to engage surface targets.
Gunnery Exercise (Air-to-Surface) – Medium-caliber	Fixed-wing and helicopter aircrews, including embarked personnel, use medium-caliber guns to engage surface targets.
Missile Exercise (Air-to-Surface) – Rocket (MISSILEX [A-S] – Rocket)	Fixed-wing and helicopter aircrews fire precision-guided and unguided rockets against surface targets.

Activity Name	Activity Description
Missile Exercise (Air-to-Surface) – Missile (MISSILEX [A-S] – Missile)	Fixed-wing and helicopter aircrews fire precision-guided missiles against surface targets.
Laser Targeting (at sea)	Fixed-winged, helicopter, and ship crews illuminate enemy targets with lasers.
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	Fixed-wing aircrews deliver bombs against surface targets.
Torpedo Exercise (Submarine-to-Surface)	Submarine attacks a surface target using exercise or live-fire torpedoes.
Missile Exercise (Surface-to-Surface) (MISSILEX [S-S])	Surface ship crews defend against threat missiles and other surface ships with missiles.
Gunnery Exercise Surface-to-Surface (Ship) – Large-caliber (GUNEX-S-S [Ship])	Ship crews engage surface targets with ship's large-caliber guns.
Gunnery Exercise Surface-to-Surface (Ship) – Small- and Medium-caliber (GUNEX-S-S [Ship])	Ship crews engage surface targets with ship's small- and medium-caliber guns.
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.
Gunnery Exercise Surface-to-Surface (Boat) (GUNEX-S-S [Boat])	Small boat crews engage surface targets with small- and medium-caliber weapons.
Maritime Security Operations (MSO)	Helicopter and surface ship crews conduct a suite of Maritime Security Operations (e.g., Vessel Search, Board, and Seizure; Maritime Interdiction Operations; Force Protection; and Anti-Piracy Operation).

2.1.5 Anti-Submarine Warfare

The mission of anti-submarine warfare is to locate, neutralize, and defeat hostile submarine threats to surface forces. Anti-submarine warfare (ASW) is based on the principle of a layered defense of surveillance and attack aircraft, ships, and submarines all searching for hostile submarines. These forces operate together or independently to gain early warning and detection, and to localize, track, target, and attack hostile submarine threats. Table 5 provides summaries of training activities in support of ASW.

Anti-submarine warfare training addresses basic skills such as detection and classification of submarines, and distinguishing between sounds made by enemy submarines and those of friendly submarines, ships, and marine life. More advanced, integrated ASW training exercises are conducted in coordinated, at-sea training events involving submarines, ships, fixed wing aircraft, and helicopters. This training integrates the full spectrum of anti-submarine warfare from detecting and tracking a submarine to attacking a target using either exercise torpedoes or simulated weapons.

Table 5. Typical Anti-Submarine Warfare Training Exercises

Activity Name	Activity Description
Anti-Submarine Warfare	
Tracking Exercise/Torpedo Exercise – Helicopter (TRACKEX/TORPEX – Helo)	Helicopter crews search, track, and detect submarines. Exercise torpedoes may be used during this event.
Tracking Exercise – Maritime Patrol Aircraft Extended Echo Ranging Sonobuoys	Maritime patrol aircraft crews search, detect and track submarines using explosive source sonobuoys or multistatic active coherent system.
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/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 – Submarine (TRACKEX/TORPEX – Sub)	Submarine crews search, detect, and track submarines and surface ships. Exercise torpedoes may be used during this event.

2.1.6 Electronic Warfare

The mission of electronic warfare (EW) is to degrade the enemy's ability to use their electronic systems, such as communication systems and radar, in order to confuse or deny them the ability to defend their forces and assets. Electronic warfare is also used to recognize an emerging threat and counter an enemy's attempt to degrade the electronic capabilities of the Navy. Table 6 provides summaries of training activities in support of EW.

Typical EW activities include threat avoidance training, signals analysis for intelligence purposes, and use of airborne and surface electronic jamming devices to defeat tracking and communications systems.

Table 6. Typical Electronic Warfare Training Exercises

Activity Name	Activity Description
Electronic Warfare	
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.

Activity Name	Activity Description
Counter Targeting – Flare Exercise (FLAREX) – Aircraft	Fixed-winged aircraft and helicopters crews defend against an attack by deploying flares to disrupt threat infrared (IR) missile guidance systems.
Counter Targeting Chaff Exercise (CHAFFEX) – Ship	Surface ships defend against an attack by deploying chaff, a radar reflective material, which disrupt threat targeting and missile guidance radars.
Counter Targeting Chaff Exercise (CHAFFEX) – Aircraft	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.

2.1.7 Mine Warfare

The mission of mine warfare (MIW) is to detect, and avoid or neutralize mines to protect Navy ships and submarines and to maintain free access to ports and shipping lanes. Mine warfare also includes offensive mine laying to gain control of, or deny the enemy access to sea space. Naval mines can be laid by ships (including purpose-built minelayers), submarines, or aircraft. Table 7 provides summaries of training activities in support of MIW.

Mine warfare neutralization (destruction) training includes exercises in which ships, aircraft, submarines, or underwater vehicles search for mines. Personnel train to destroy or disable mines by attaching and detonating underwater explosives to the mine. Other neutralization techniques involve impacting the mine with a bullet-like projectile or intentionally triggering the mine to detonate.

Table 7. Typical Mine Warfare Exercises

Activity Name	Activity Description
Mine Warfare	
Civilian Port Defense	Naval mine warfare activities conducted at various ports and harbors, in support of maritime homeland defense/security.
Mine Laying	Fixed-winged aircraft and vessel crews drop/launch non explosive mine shapes.
Mine Neutralization – Explosive Ordnance Disposal (EOD)	Personnel disable threat mines. Explosive charges may be used.
Limpet Mine Neutralization System/Shock Wave Generator	Navy divers place a small charge on a simulated underwater mine.
Submarine Mine Exercise	Submarine crews practice detecting mines in a designated area.
Airborne Mine Countermeasure (MCM) – Mine Detection	Helicopter aircrews detect mines using towed and laser mine detection systems (e.g., AN/AQS-20, Airborne Laser Mine Detection System).

Activity Name	Activity Description
Mine Countermeasure Exercise – Towed Sonar	Surface ship crews detect and avoid mines while navigating restricted areas or channels using towed active sonar.
Mine Countermeasure Exercise – Surface (SMCMEX)	Mine countermeasure ship crews detect, locate, identify, and avoid mines while navigating restricted areas or channels using active sonar.
Mine Neutralization – Remotely Operated Vehicle Sonar	Helicopter aircrews disable mines using remotely operated underwater vehicles.
Mine Countermeasure (MCM) – Towed Mine Neutralization	Ship crews and helicopter aircrews tow systems (e.g., Organic and Surface Influence Sweep, MK 104/105) through the water that are designed to disable and/or trigger mines.

2.1.8 Naval Special Warfare

The mission of naval special warfare (NSW) 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. Table 8 provides summaries of training activities in support of NSW.

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.

Table 8. Typical Naval Special Warfare Exercises

Activity Name	Activity Description
Naval Special Warfare	
Personnel Insertion/Extraction	Military personnel train for covert insertion and extraction into target areas using helicopters, fixed-wing aircraft (insertion only), small boats, and submersibles.
Direct Action (Tactical Air Control Party [TACP]/Joint Tactical Air Control)	Military personnel train for controlling of combat support aircraft; providing target designation, airspace de-confliction, and terminal control for Close Air Support. Teams also train in use of small arms and mortars.
Underwater Demolition Qualification/Certification	Navy divers conduct training and certification in placing underwater demolition charges.
Intelligence, Surveillance, Reconnaissance (ISR)	Special Warfare units train to collect and report battlefield intelligence.
Underwater Survey	Navy divers train in survey of underwater conditions and features in preparation for insertion, extraction, or intelligence, surveillance and reconnaissance activities.

2.1.9 Major Training and Other Training Activities

Major training exercises provide multi-Service and Joint participation in realistic maritime and expeditionary training that is designed to replicate the types of events and challenges that could be faced during real-world contingency operations. Major training exercises also include providing training to submarine, ship, aircraft, and special warfare forces in mission tactics, techniques, and procedures. Table 9 provides summaries of Major Training and Other Training Activities.

Table 9. Major Training and Other Training Activities

Activity Name	Activity Description
Major Training Activities	
Joint Expeditionary Exercise	A 10-day at-sea and ashore exercise which brings different branches of the United States military together in a joint environment that includes planning and execution efforts as well as military training activities at sea, in the air, and ashore. More than 8,000 personnel may participate and could include the combined assets of a Carrier Strike Group and Expeditionary Strike Group, Marine Expeditionary Units, Army Infantry Units, and Air Force aircraft.
Joint Multi-Strike Group Exercise	A 10-day at-sea and ashore exercise in which up to three Carrier Strike Groups integrated with U.S. Air Force and U.S. Marine Corps forces would conduct at-sea training and STW exercises simultaneously.
Marine Air Ground Task Force Exercise (Amphibious) – Battalion	A 10-day at-sea and shore exercise which conducts over the horizon, ship to objective maneuver for the elements of the Expeditionary Strike Group and the Amphibious Marine Air Ground Task Force. The exercise utilizes all elements of the Marine Air Ground Task Force (Amphibious), conducting training activities ashore with logistic support of the Expeditionary Strike Group and conducting amphibious landings.
Special Purpose Marine Air Ground Task Force Exercise	A 10-day at-sea and ashore exercise similar to Marine Air Ground Task Force (Amphibious) – Battalion, but task organized to conduct a specific mission (e.g., Humanitarian Assistance, Disaster Relief, Non-combatant Evacuation Operations).
Other Training Activities	
Surface Ship Sonar Maintenance	In-port and at-sea maintenance of sonar systems.
Submarine Sonar Maintenance	In-port and at-sea maintenance of sonar systems.
Small Boat Attack	Small boats or personal watercraft conduct attack activities on units afloat.
Submarine Navigation	Submarine crews locate underwater objects and ships while transiting out of port.
Search and Rescue at Sea	United States Coast Guard and military personnel train with ships, fixed wing and rotary aircraft to locate and rescue missing personnel and vessels at sea.

Activity Name	Activity Description
Precision Anchoring	Releasing of anchors in designated locations.

2.2 Mariana Islands Testing Activities

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

The individual commands within the research and acquisition community included in this opinion are Naval Air Systems Command, Naval Sea Systems Command, the Office of Naval Research, and the Naval Research Laboratory.

The Navy operates in an ever-changing strategic, tactical, and funding and time-constrained environment. Testing activities occur in response to emerging science or fleet operational needs. For example, future Navy experiments to develop a better understanding of ocean currents may be designed based on advancements made by non-government researchers not yet published in the scientific literature. Similarly, future but yet unknown Navy operations within a specific geographic area may require development of modified Navy assets to address local conditions. Such modifications must be tested in the field to ensure they meet fleet needs and requirements.

Some testing activities are similar to training activities conducted by the fleet. For example, both the fleet and the research and acquisition community fire 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 that the torpedo meets performance specifications and operational requirements. These differences may result in different analysis and potential mitigations for the activity.

2.2.1 Naval Air Systems Command Testing Activities

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 delivered to 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.

The majority of testing and development activities (Table 10) conducted by NAVAIR are similar to fleet training activities, and many platforms (e.g., Maritime Patrol Aircraft) and systems (e.g., sonobuoys) currently being tested are already being used by the fleet or will ultimately be integrated into fleet training activities. However, some testing and development may be conducted in different locations and in a different manner than the fleet and therefore, though the potential environmental effects may be the same, the analysis for those activities may differ. Training with systems and platforms delivered to the fleet within the timeframe of this document are analyzed in the training sections of the MITT EIS/OEIS.

Table 10. Typical Naval Air Systems Command Testing Activities

Testing Event	Description	Weapons/Rounds/ Sound Source
Anti-Surface Warfare		
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 weapons system or as part of another systems integration test.	Explosive missiles
Anti-Submarine Warfare		
Anti-Submarine Warfare Torpedo Test	This event is similar to the training event torpedo exercise. The Test evaluates anti-submarine 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. Some tests from fixed-wing aircraft will involve releasing torpedoes and sonobuoys from high altitudes (approximately 25,000 feet [ft.]).	Exercise (Non-explosive) torpedoes
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (Sonobuoy)	This event is similar to the training event anti-submarine warfare TRACKEX-Maritime Patrol Aircraft. 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.	Directional Command Activated Sonobuoy System (DICASS) active sonobuoys, Improved Extended Echo Ranging (IEER) sonobuoys (2 detonations per IEER buoy), High Duty Cycle sonobuoys, various Signal Underwater Sound (SUS) devices, Multi-static Active Coherent (MAC) sonobuoys
Electronic Warfare		

Testing Event	Description	Weapons/Rounds/ Sound Source
Flare Test	Flare tests evaluate newly developed or enhanced flares, flare dispensing equipment, or modified aircraft systems against flare deployment. Tests may also train pilots and aircrew in the use of newly developed or modified flare deployment systems. Flare tests are often conducted with other test events, and are not typically conducted as standalone tests.	Chaff and flares are expended

2.2.2 Naval Sea Systems Command Testing Activities

Naval Sea Systems Command testing activities are aligned with its mission of new ship construction, life cycle support, and weapon systems development. Each major category of Naval Sea Systems Command activities is described below:

2.2.2.1 New Ship Construction Activities

Ship construction activities include testing of ship systems, and developmental and operational test and evaluation programs for new technologies and systems. At-sea testing of systems aboard a ship may include sonar, acoustic countermeasures, radars, and radio equipment. At-sea test firing of shipboard weapon systems, including guns, torpedoes, and missiles, are also conducted. Table 11 provides summaries of testing activities in support of ship construction and maintenance.

Table 11. Typical New Ship Construction Activities

Activity Name	Activity Description
New Ship Construction	
Anti-Submarine Warfare Mission Package Testing	Ships and their supporting platforms (e.g., helicopters, unmanned aerial systems) detect, localize, and prosecute submarines.
Anti-Surface Warfare Mission Package Testing	Ships and their supporting platforms (e.g., helicopters, unmanned aerial vehicles) detect, localize, and prosecute surface vessels.
Mine Countermeasure Mission Package Testing	Ships conduct mine countermeasure operations.

2.2.2.2 Life Cycle Activities

Testing activities (Table 12) are conducted throughout the life of a Navy ship to verify performance and mission capabilities. Sonar systems testing occurs pierside during maintenance, repair, and overhaul availabilities, and at sea immediately following most major overhaul periods. 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 to radar. Additionally, electromagnetic measurements of off-board electromagnetic signature are conducted for submarines, ships, and surface crafts periodically.

Table 12. Typical Life Cycle Activities in the Study Area

Activity Name	Activity Description
Life Cycle Activities	
Ship Signature Testing	Tests ship and submarine radars, electromagnetic, or acoustic signatures.

2.2.3 Other Naval Sea System Command Testing Activities

Numerous test activities and technical evaluations, in support of Naval Sea Systems Command's systems development mission, often occur in conjunction with fleet activities within the MITT Study Area. Tests within this category include, but are not limited to anti-submarine warfare and mine warfare tests using torpedoes, sonobuoys, and mine detection and neutralization systems. Pierside, swimmer detection systems will also be tested.

Unique Naval Sea Systems Command planned testing includes a kinetic energy weapon for Navy ships, which uses electromagnetic energy to propel a projectile at a surface, air, or ground target.

Table 13. Other Naval Sea System Command Testing Activities.

Activity Name	Activity Description
Anti-Surface Warfare/Anti-Submarine Warfare Testing	
Kinetic Energy Weapon Testing	A kinetic energy weapon uses stored electromagnetic energy released in a burst to accelerate a projectile. Projectiles used for testing are either non-explosive or in-air explosive munitions.
Torpedo Testing	Air, surface, or submarine crews employ live/exercise torpedoes against submarines or surface vessels.
Countermeasure Testing	Various systems (e.g., towed arrays and defense systems) are employed to detect, localize, and track incoming weapons.
At-sea Sonar Testing	At-sea testing to ensure systems are fully functional in an open ocean environment.
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.

2.2.4 Office of Naval Research and Naval Research Laboratory Testing Activities

As the Navy's Science and Technology provider, Office of Naval Research and the Naval Research Laboratory provide technology solutions for Navy and Marine Corps needs. The Office of Naval Research's missions, defined by law, are 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, the Office of Naval Research 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. The Office of Naval Research activities 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. Office of Naval Research testing is shown in Table 14.

Table 14. Typical Office of Naval Research Testing Activity in the Study Area

Activity Name	Activity Description
Office of Naval Research	
North Pacific Acoustic Lab Philippine Sea 2018–19 Experiment (Deep Water)	The experiment area encompasses international waters. The initial experiment was completed in May of 2011; an acoustic tomography array, a distributed vertical line array (DVLA), and moorings were deployed in the deep-water environment of the northwestern Philippine Sea. The acoustic tomography array and DVLA have remained in situ at the experiment site since that time, collecting oceanographic and acoustic data used to study deep-water propagation and to characterize the temperature and velocity structure in this oceanographically complex and highly dynamic region. In addition, data will be collected during two periods of intensive experimental at-sea operations in May and July of 2018. During fall 2018, data will be collected passively by remotely sensing seagliders. Research vessels, acoustic test sources, side scan sonar, ocean gliders, the existing moored acoustic tomographic array and distributed vertical line array, and other oceanographic data collection equipment will be used to collect information on the ocean environment. The final phases of the experiment will be completed during March through May 2019. The resulting analyses will aid in developing a more complete understanding of deep water sound propagation and the temperature-velocity profile of the water column in this part of the world.

2.3 Sonar, Ordnance, Targets, and Other Systems Used in Training and Testing

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 application: 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.

2.3.1 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 the listed 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. Even when the Navy is "operating continuously" while conducting ASW training, the sonar sends out a ping nominally once every 50 seconds, it does not ping continuously when it is in operation. Therefore, even when operating continuously sonar signals are only actively transmitting two percent of the time. 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.

2.3.2 Ordnance and Munitions

Most ordnance and munitions used during training 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 and testing events. Non-explosive ordnance munitions look and perform similarly to high explosive ordnance, but lack the main explosive charge.

2.3.3 Defense 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 opinion 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.

2.3.4 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 potential 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 (*Zalopus 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.

2.3.5 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 potential impacts to marine mammals:

- 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.

2.4 Proposed Training Activity Levels

The following table provides a summary of training activities (as described in Section 2.1 above) including tempo and quantities of inert and live munitions that the Navy plans to expend during training that were analyzed by the U.S. Navy. Munitions that contain explosives are shaded grey in the table to highlight activities that might have greater potential for impact to listed species.

Table 15. Training Activities Ocurring in the Action Area

Range Activity	Number of Activities (per year)	Ordnance (Number per year)	Location
Anti-Air Warfare			
Air Combat Maneuver (ACM)	4,800	None	Study Area > 12 nm from land: SUA
Air Defense Exercise (ADEX)	100	None	Study Area > 12 nm from land: SUA
Air Intercept Control (AIC)	4,800	None	Study Area > 12 nm from land: SUA
Gunnery Exercise (Air-to-Air) – Medium-caliber (GUNEX [A-A]) Medium-caliber	36	9,000 rounds	Study Area SUA > 12 nm from land
Missile Exercise (Air-to-Air) (MISSILEX [A-A])	18	36 explosive missiles	Study Area SUA > 12 nm from land
Gunnery Exercise (Surface-to-Air) – Large-caliber (GUNEX [S-A]) – Large-caliber	5	40 rounds	Study Area SUA > 12 nm from land
Gunnery Exercise (Surface-to-Air) – Medium-caliber (GUNEX [S-A]) – Medium-caliber	12	24,000 rounds	Study Area SUA > 12 nm from land
Missile Exercise (Surface-to-Air) (MISSILEX [S-A])	15	15 explosive missiles	Study Area SUA > 12 nm from land
Strike Warfare (STW)			
Combat Search and Rescue	80	None	MIRC
Bombing Exercise (Air-to-Ground) (BOMBEX [A-G])*	2,300	2,670 NEPM 6,242 explosive rounds	FDM

Range Activity	Number of Activities (per year)	Ordnance (Number per year)	Location
Gunnery Exercise (Air-to-Ground) (GUNEX [A-G])*	96	24,000 small-caliber rounds 94,150 medium-caliber rounds 17,350 explosive med.-caliber rounds 200 explosive large-caliber rounds	FDM
Missile Exercise (MISSILEX)*	85	2,000 explosive rockets 85 explosive missiles	FDM
Amphibious Warfare (AMW)			
Amphibious Rehearsal, No Landing – Marine Air Ground Task Force	12	None	Study Area and Nearshore
Amphibious Assault	6	Blanks; Simunitions	MIRC; Tinian; Guam
Amphibious Raid	6	Blanks; Simunitions	MIRC; Tinian; Guam; Rota
Anti-Surface Warfare (ASUW)			
Gunnery Exercise (Air-to-Surface) – Small-caliber (GUNEX [A-S]) – Small-caliber	242	48,040 rounds	Study Area SUA > 12 nm from land
Gunnery Exercise (Air-to-Surface) – Medium-caliber (GUNEX [A-S]) – Medium-caliber	295	36,650 (7,150 explosive)	Study Area SUA > 12 nm from land; Transit Corridor
Missile Exercise (Air-to-Surface) – Rocket (MISSILEX [A-S] – Rocket)	3	114 rockets (114 explosive)	Study Area SUA > 12 nm from land
Missile Exercise (Air-to-Surface) (MISSILEX [A-S])	20	20 explosive missiles	Study Area SUA > 12 nm from land
Laser Targeting (at sea)	600	None	Study Area SUA > 12 nm from land
Bombing Exercise (Air-to-Surface) (BOMBEX [A-S])	37	368 NEPM 184 explosive	Study Area > 50 nm from land
Torpedo Exercise (Submarine-to-Surface)	5	10 EXTORP	Study Area > 3 nm from land
Missile Exercise (Surface-to-Surface) (MISSILEX [S-S])	12	12 Missiles explosive	Study Area > 50 nm from land
Gunnery Exercise (Surface-to-Surface) Ship – Large-caliber (GUNEX [S-S] – Ship) Large-caliber	140	5,698 rounds (500 explosive)	Study Area SUA > 12 nm from land; Transit Corridor

Range Activity	**Number of Activities (per year)**		**Ordnance (Number per year)**	**Location**
Gunnery Exercise (Surface-to-Surface) Ship – Small- and Medium-caliber (GUNEX [S-S] – Ship) Small- and Medium-caliber	100		21,900 rounds (900 explosive)	Study Area SUA > 12 nm from land; Transit Corridor
Sinking Exercise (SINKEX) Representative ordnance. Actual ordnance used will vary (typically less than shown).	2		28 explosive Bombs 42 explosive Missiles 800 explosive Large-caliber rounds 2 MK-48 explosive 4 explosive Demolitions	Study Area > 50 nm from land and > 1,000 fathoms depth
Gunnery Exercise (Surface-to-Surface) Boat – Small and Medium-caliber (GUNEX [S-S] – Boat	Medium-caliber	10	2,100 (100 explosive)	Study Area SUA > 12 nm from land; Transit Corridor
	Small-caliber	40	36,000 rounds	Study Area > 3 nm from land; Transit Corridor
Maritime Security Operations (MSO)	40		200 G911 anti-swimmer grenade	Study Area; MIRC
Anti-Submarine Warfare (ASW)				
Tracking Exercise – Helicopter (TRACKEX – Helo)	62		None/ REXTORP	Study Area > 3 nm from land; Transit Corridor
Torpedo Exercise – Helicopter (TORPEX – Helo)	4		4 EXTORP	Study Area > 3 nm from land
Tracking Exercise – Maritime Patrol Advanced Extended Echo Ranging Sonobuoys	11		None	Study Area > 3 nm from land
Tracking Exercise – Maritime Patrol Aircraft (TRACKEX – Maritime Patrol Aircraft)	34		None/ REXTORP	Study Area > 3 nm from land
Torpedo Exercise – Maritime Patrol Aircraft (TORPEX – Maritime Patrol Aircraft)	4		4 EXTORP	Study Area > 3 nm from land
Tracking Exercise – Surface (TRACKEX – Surface)	CG/DDG-92 FFG-30 LCS-10		None/ REXTORP	Study Area > 3 nm from land
Torpedo Exercise – Surface (TORPEX – Surface)	3		3 EXTORP	Study Area > 3 nm from land
Tracking Exercise – Submarine (TRACKEX – Sub)	12		None	Study Area > 3 nm from land; Transit Corridor

Range Activity	**Number of Activities (per year)**	**Ordnance (Number per year)**	**Location**
Torpedo Exercise – Submarine (TORPEX – Sub)	10	40 MK-48 EXTORP	Study Area > 3 nm from land
Major Training Activities			
Joint Expeditionary Exercise	1	Note 1	Study Area; MIRC
Joint Multi-Strike Group Exercise	1	Note 1	Study Area; MIRC
Marine Air Ground Task Force Exercise (Amphibious) – Battalion	4	Note 1	Study Area to nearshore; MIRC; Tinian; Guam; Rota; Saipan; FDM
Special Purpose Marine Air Ground Task Force Exercise	2	Note 1	Study Area to nearshore; MIRC; Tinian; Guam; Rota; Saipan
Electronic Warfare (EW)			
Electronic Warfare Operations (EW Ops)	480	None	Study Area
Counter Targeting Flare Exercise (FLAREX) – Aircraft	3,200	25,600 cartridges	Study Area > 12 nm from land
Counter Targeting Chaff Exercise (CHAFFEX) – Ship	40	240 cartridges	Study Area > 12 nm from land
Counter Targeting Chaff Exercise (CHAFFEX) – Aircraft	3,200	25,600 cartridges	Study Area > 12 nm from land
Mine Warfare (MIW)			
Civilian Port Defense	1	Note 1	Mariana littorals; MIRC; Inner and Outer Apra Harbor
Mine Laying	4	480 mine shapes	MIRC Warning Areas
Mine Neutralization – Explosive Ordnance Disposal (EOD)	20	20 explosive charges	MIRC mine neutralization sites at Piti and Outer Apra Harbor site are 10 lb. NEW maximum and 20 lb NEW maximum at Agat Bay Mine Neutralization Site
Limpet Mine Neutralization System/Shock Wave Generator	40	40 charges	Mariana littorals; Inner and Outer Apra Harbor
Submarine Mine Exercise	16	n/a	Study Area; nearshore
Airborne Mine Countermeasure – Mine Detection	4	n/a	Study Area; nearshore
Mine Countermeasure Exercise – Towed Sonar (AQS-20, LCS)	4	n/a	Study Area

Range Activity	Number of Activities (per year)	Ordnance (Number per year)	Location
Mine Countermeasure Exercise – Surface (SMCMEX) Sonar (SQQ-32, MCM)	4	n/a	Study Area
Mine Neutralization – Remotely Operated Vehicle Sonar (ASQ-235 [AQS-20], SLQ-48)	4	4 explosive neutralizers	Study Area
Mine Countermeasure – Towed Mine Detection	4	n/a	Study Area
Naval Special Warfare (NSW)			
Personnel Insertion/ Extraction	240	None	MIRC; Guam; Tinian; Rota
Underwater Demolition Qualification/ Certification	30	30 explosive charges	MIRC underwater demolition sites, 20 lb. NEW maximum charge (except Piti 10 lb. NEW maximum)
Other			
Surface Ship Sonar Maintenance	42	None	Study Area > 3 nm from land; Inner Apra Harbor; Transit Corridor
Submarine Sonar Maintenance	48	None	Study Area > 3 nm from land; Inner Apra Harbor; Transit Corridor
Small Boat Attack	6	2,100 small-caliber rounds	Study Area > 3 nm from land
	12	4,000 blank rounds	Study Area
Submarine Navigation	8	None	Apra Harbor and Mariana littorals
Search and Rescue At Sea	40	None	Study Area
Precision Anchoring	18	None	Apra Harbor; Mariana Islands anchorages

Notes:
(1) Exercise is composed of various activities accounted for elsewhere within Table
(2) Discussed as an embedded training activity to CHAFFEX/FLAREX in MIRC EIS/OEIS Appendix D (Air Quality Calculations and Record of Non-Applicability).
(3) CHAFF = Chaff Exercise, EIS = Environmental Impact Statement, EOD = Explosive Ordnance Disposal, EXTORP = Exercise Torpedo, FDM = Farallon de Medinilla, FLAREX = Flare Exercise, lb. = pounds, LCS = Littoral Combat Ship, MIRC = Mariana Islands Range Complex, mm = millimeters, n/a = Not Applicable, NEPM = Non-explosive Practice Munitions, NEW = Net Explosive Weight, nm = nautical miles, OEIS = Overseas Environmental Impact Statement, REXTORP = Recoverable Exercise Torpedo, SUA = Special Use Airspace
*** Ordnance are dropped on land (FDM) but are included in this table as a very small percentage may miss or richochet and end up in the water.**

2.5 Proposed Testing Activity Levels / Naval Air Systems Command

The following table provides a summary of testing activities including tempo and quantities of inert and live munitions that the Naval Air Systems Command plans to expend during testing that were analyzed by the U.S. Navy. Munitions containing explosives are shaded grey in the table to highlight activities that may have greater potential for impacts to listed resources than inert materials.

Table 16. Proposed Naval Air Systems Command Testing Activities

Range Activity	Action		
	No. of activities (per year)	Ordnance (Number per year)	Location
Anti-Surface Warfare			
Air-to-Surface Missile Test	8	8 harpoon missiles (4 explosive)	Action Area > 50 nm from land
Anti-Submarine Warfare			
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft (Sonobuoys)	188	240 IEER/MAC 553 SUS	Action Area > 3 nm from land
Anti-Submarine Warfare Torpedo Test	40	40 EXTORP	Action Area > 3 nm from land
Broad Area Maritime Surveillance (BAMS) Testing – MQ-4C Triton	10	None	Action Area
Electronic Warfare			
Flare Test	10	None	Action Area > 3 nm from land

Notes: EIS = Environmental Impact Statement, EXTORP = Exercise Torpedo, IEER = Improved Extended Echo Ranging, MAC = Multi-static Active Coherent, nm = nautical miles, OEIS = Overseas Environmental Impact Statement, SUS = Signal Underwater Sound

2.6 Proposed Testing Activity Levels - Naval Sea Systems Command

The following table provides a summary of testing activities including tempo and quantities of inert and live munitions that the Naval Sea Systems Command plans to expend during testing that were analyzed by the U.S. Navy. Munitions containing explosives are shaded grey in the table to highlight activities that may have greater potential for impacts to listed resources than inert materials.

Table 17. Proposed Naval Sea Systems Command Testing Activities

Range Activity	Action		
	Number of activities (per year)	Ordnance (Number per year)	Location
Life Cycle Activities			
Ship Signature Testing	17	None	Action Area
Anti-Surface Warfare/Anti-Submarine Warfare Testing			
Kinetic Energy Weapon Testing	50	2,000 projectiles	MIRC > 12 nm from land
	1 event total	5,000 projectiles	
Torpedo Testing	2	20 torpedoes (8 explosive)	MIRC > 3 nm from land
Countermeasure Testing	2	56 torpedoes	Action Area
At-Sea Sonar Testing	20	None	Action Area
New Ship Construction			
ASW Mission Package Testing	33	None	Action Area
MCM Mission Package Testing	32	48 neutralizers (24 explosive)	Action Area

Range Activity	Action			
	Number of activities (per year)		Ordnance (Number per year)	Location
ASUW Mission Package Testing	Gun Testing – Small-caliber	4	2,000 rounds	Action Area; Warning Area > 12 nm from land
	Gun Testing – Medium-caliber (30 mm)	4	4,080 rounds (2,040 explosive)	
	Gun Testing – Large-caliber (57 mm)	4	5,600 rounds (3,920 in-air explosive)	
	Missile/ Rocket Testing	4	32 missiles/ rockets (16 explosive)	

Notes: IEER = Improved Extended Echo Ranging, MCM = Mine Countermeasure, MIRC = Mariana Islands Range Complex, mm = millimeters, nm = nautical miles

2.7 Office of Naval Research and Naval Research Laboratory Testing Activities

As the Department of the Navy’s Science and Technology provider, Office of Naval Research and Naval Research Laboratory provide technology solutions for Navy and Marine Corps needs. The Office of Naval Research's mission, defined by law, 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, Office of Naval Research 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. The Office of Naval Research events include research, development, test, and evaluation activities; surface processes acoustic communications experiments; shallow and deep water acoustic communications experiments; sediment acoustics experiments; shallow and deep water acoustic propagation experiments; and long-range acoustic propagation experiments.

Table 18. Proposed Naval Research Activities

Office of Naval Research	Action		
	Number of activities (per year)	Ordnance (Number per year)	Location
North Pacific Acoustic Lab Phillipine Sea 2018-19 Experiment (Deep Water)	1	n/a	Action Area

2.8 U.S. Navy Proposed Mitigation

This section summarizes the Navy's mitigation measures, that were developed in close coordination with NMFS' Office of Protected Resources based on previous consultations and lessons-learned from employment during Phase I (2010 to 2015) training and testing activities. Table 19 provides a summary of the Navy's proposed mitigation measures. These measures are also described in NMFS' proposed final rule in this opinion. The measures presented in the table are discussed in greater detail in the Final EIS/OEIS, May 2015.

Table 19. Summary of the U.S. Navy's Mitigation

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Specialized Training	Lookouts will complete the Introduction to the U.S. Navy Afloat Environmental Compliance Training Series and the U.S. Navy Marine Species Awareness Training or civilian equivalent.	The mitigation zones observed by Lookouts are specified for each Mitigation Zone Procedural Measure below.
Low-Frequency and Hull-Mounted Mid-Frequency Active Sonar during Anti-Submarine Warfare and Mine Warfare	2 Lookouts (general) 1 Lookout (minimally manned, moored, or anchored)	Sources that can be powered down: 1,000 yd. (914 m) and 500 yd. (457 m) power downs and 200 yd. (183 m) shutdown for marine mammals (hull-mounted mid-frequency and low-frequency) and sea turtles (low-frequency only). Sources that cannot be powered down: 200 yd. (183 m) shutdown for marine mammals and sea turtles.
High-Frequency and Non-Hull Mounted Mid-Frequency Active Sonar	1 Lookout	200 yd. (183 m) for marine mammals (high-frequency and mid-frequency), sea turtles (bins MF8, MF9, MF10, and MF12 only)
Improved Extended Echo Ranging Sonobuoys	1 Lookout	600 yd. (549 m) for marine mammals and sea turtles Passive acoustic monitoring conducted with Navy assets participating in the activity.
Explosive Sonobuoys using >0.5–2.5 lb. NEW	1 Lookout	350 yd. (320 m) for marine mammals and sea turtles Passive acoustic monitoring conducted with Navy assets participating in the activity.
Anti-Swimmer Grenades	1 Lookout	200 yd. (183 m) for marine mammals and sea turtles
Mine Countermeasures and Mine Neutralization using Positive Control Firing Devices	General: 1 or 4 Lookouts (NEW dependent) Diver-placed: 2 Lookouts	NEW dependent for marine mammals, sea turtles, and hammerhead sharks
Mine Neutralization Activities Using Diver-Placed Time-Delay Firing Devices	4 Lookouts	Up to 10-minute time-delay using up to 20 lb. NEW: 1,000 yd. (914 m) for marine mammals, sea turtles, and hammerhead sharks.

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Explosive and Non-Explosive Gunnery Exercises – Small- and Medium-Caliber Using a Surface Target	1 Lookout	200 yd. (183 m) for marine mammals and sea turtles
Explosive and Non-Explosive Gunnery Exercises – Large-Caliber Using a Surface Target	1 Lookout	Explosive: 600 yd. (549 m) for marine mammals and sea turtles Non-Explosive: 200 yd. (183 m) for marine mammals and sea turtles Both: 70 yd. (64 m) within 30 degrees on either side of the gun target line on the firing side for marine mammals and sea turtles Both: 350 yd. (320 m) for surveyed shallow coral reefs
Non-Explosive Missile Exercises and Explosive Missile Exercises (Including Rockets) up to 250 lb. NEW using a Surface Target	1 Lookout	900 yd. (823 m) for marine mammals and sea turtles 350 yd. (320 m) for surveyed shallow coral reefs
Explosive Missile Exercises (Including Rockets) from > 250 to 500 lb. NEW using a Surface Target	1 Lookout	2,000 yd. (1.8 km) for marine mammals and sea turtles 350 yd. (320 m) for surveyed shallow coral reefs
Bombing Exercises, Explosive and Non-Explosive	1 Lookout	Explosive: 2,500 yd. (2.3 km) for marine mammals and sea turtles Non-Explosive: 1,000 yd. (914 m) for marine mammals and sea turtles Both: 350 yd. (320 m) for surveyed shallow coral reefs
Torpedo (Explosive) Testing	1 Lookout	2,100 yd. (1.9 km) for marine mammals and sea turtles and jellyfish aggregations Passive acoustic monitoring conducted with Navy assets participating in the activity.
Sinking Exercises	2 Lookouts	2.5 nm for marine mammals and sea turtles and jellyfish aggregations. Passive acoustic monitoring conducted with Navy assets participating in the activity.

Activity Category or Mitigation Area	Recommended Lookout Procedural Measure	Recommended Mitigation Zone and Protection Focus
Vessel Movements	1 Lookout	500 yd. (457 m) for whales 200 yd. (183 m) for all other marine mammals (except bow riding dolphins)
Towed In-Water Device Use	1 Lookout	250 yd. (229 m) for marine mammals
Precision Anchoring	No Lookouts in addition to standard personnel standing watch	Avoidance of precision anchoring within the anchor swing diameter of shallow coral reefs, live hardbottom, artificial reefs, and shipwrecks Except at existing anchorages as well as at near-shore training areas around Guam and in Apra Harbor, the Navy will not conduct precision anchoring activities within the anchor swing diameter of surveyed shallow coral reefs, live hardbottom, artificial reefs, and shipwrecks.
Shallow Coral Reefs, Hardbottom Habitat, Artificial Reefs, and Shipwrecks	No Lookouts in addition to standard personnel standing watch	Except at existing anchorages as well as at near-shore training areas around Guam and in Apra Harbor, the Navy will not conduct precision anchoring within the anchor swing diameter, or explosive mine countermeasure and neutralization activities within 350 yd. (320 m) of surveyed shallow coral reefs, live hardbottom, artificial reefs, and shipwrecks. No explosive or non-explosive small-, medium-, and large-caliber gunnery exercises using a surface target, explosive or non-explosive missile exercises using a surface target, explosive and non-explosive bombing exercises, or at-sea explosive testing within 350 yd. (320 m) of surveyed shallow coral reefs

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

On April 17, 2013, NMFS’ Permits Division received an application from the U.S. Navy requesting regulations and a LOA for the take of 26 species of marine mammals incidental to Navy training and testing activities to be conducted in the MITT Study Area over 5 years. The Navy requested regulations that would establish a process for authorizing take, via a five-year LOA, of marine mammals for training and testing activities proposed to be conducted from August 2015 through August 2020.

NMFS Permits Division issued the requested regulations and an LOA on July 31, 2015.

NMFS Permits Division determined that the taking of marine mammals incidental to Navy training and testing activities to be conducted in the MITT Study Area from August 2015 to August 2020 would have a negligible impact[2] on species or stock(s) and would not have an unmitigable adverse impact on the availability of species or stock(s) for subsistence uses (since there are no such uses in the specified area). NMFS’ final rule and LOA also set forth the permissible methods of taking pursuant to such activity, and other means of effecting the least practicable adverse impact on affected species or stocks and their habitat, paying particular attention to rookeries, mating grounds, and areas of similar significance. The ESA-listed species that are authorized to be taken under the MMPA by the Navy’s activity, the dates when take is authorized occur, the permissible methods of taking, and the mitigation and monitoring measures required by the MMPA rule and LOA are all consistent with the Navy’s action described previously in this opinion so they will not be reiterated here.On November 24, 2015, NMFS Permits Divison amended Section 218.95(g)(1)(ii)(F) to authorize modification to watchstander requirements for observed behavior of marine mammals during Major Training Events in the MITT Study Area (80 FR 73556). These modifications do not alter the mitigation measures the Navy is required to implement. With these modifications, the Navy is no longer required to report individual marine mammal sighting information when mitigation is not being implemented during the MTEs. After 5 years of collecting marine mammal sighting data for all animals sighted during MTEs, NMFS and the Navy have determined that without the ability to obtain species information, this data set does not provide for any meaningful analysis beyond that which may be possible using mitigation-related observations alone. The Navy and NMFS have thoroughly investigated several potential uses for the data prior to reaching this conclusion. Additionally, this reporting requirement places an administrative burden on ship’s watch teams, which was undue given the limited value of the information collected, as was described during the Adaptive Management Process. The Navy will continue to collect marine mammal sighting data during MTEs for every instance when any form of mitigation is employed such as powering down or securing sonar, maneuvering the ship, or delaying an event—in other words, in

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[2] 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.”

instances where animals are closer to the sound source around which mitigation measures are implemented. This data is useful in supporting mitigation effectiveness analyses and also may be helpful in supporting an understanding of the frequency with which marine mammals (generally, not by species) may be encountered or detected in close proximity to a particular source (*e.g.,* where the likelihood of auditory or other injury is higher). The regulations as copied below reflect this amendment to Section 218.95(g)(1)(ii)(F). This change does not alter the scope, type, or level of activity proposed by the Navy, nor does it alter the mitigation measures the Navy is required to implement; therefore this amendment does not alter our assessment of the effects of this action.

We provide the text of the MMPA regulations below.

2.9.1 PART 218—REGULATIONS GOVERNING THE TAKING AND IMPORTING OF MARINE MAMMALS

1. The authority citation for part 218 continues to read as follow: Authority: 16 U.S.C. 1361 et seq.

2. Subpart J is added to part 218 to read as follows:

2.9.1.1 Subpart J – Taking and Importing Marine Mammals; U.S. Navy's Mariana Islands Training and Testing

§ 218.90 Specified activity and specified geographical region.

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

(b) The taking of marine mammals by the Navy is only authorized if it occurs within the MITT Study Area, which includes the MIRC and areas to the north and west. The Study Area includes established ranges, operating areas, warning areas, and special use airspace in the region of the Mariana Islands that are part of the MIRC, its surrounding seas, and a transit corridor to the Hawaii Range Complex. The Study Area also includes Navy pierside locations where sonar maintenance and testing may occur.

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

(1) Non-impulsive Sources Used During Training and Testing:
  (i) Low-frequency (LF) Source Classes:
    (A) LF4 – an average of 123 hours per year.
    (B) LF5 – an average of 11 hours per year.
    (C) LF6 – an average of 40 hours per year.
  (ii) Mid-frequency (MF) Source Classes:
    (A) MF1 – an average of 1,872 hours per year.

    - (B) MF2 – an average of 625 hours per year.
    - (C) MF3 – an average of 192 hours per year.
    - (D) MF4 – an average of 214 hours per year.
    - (E) MF5 – an average of 2,588 items per year.
    - (F) MF6 – an average of 33 items per year.
    - (G) MF8 – an average of 123 hours per year.
    - (H) MF9 – an average of 47 hours per year.
    - (I) MF10 – an average of 231 hours per year.
    - (J) MF11 – an average of 324 hours per year.
    - (K) MF12 – an average of 656 hours per year.
  - (iii) High-frequency (HF) and Very High-frequency (VHF) Source Classes:
    - (A) HF1 – an average of 113 hours per year.
    - (B) HF4 – an average of 1,060 hours per year.
    - (C) HF5 – an average of 336 hours per year.
    - (D) HF6 – an average of 1,173 hours per year.
  - (iv) Anti-Submarine Warfare (ASW) Source Classes:
    - (A) ASW1 – an average of 144 hours per year.
    - (B) ASW2 – an average of 660 items per year.
    - (C) ASW3 – an average of 3,935 hours per year.
    - (D) ASW4 – an average of 32 items per year.
  - (v) Torpedoes (TORP) Source Classes:
    - (A) TORP1 – an average of 115 items per year.
    - (B) TORP2 – an average of 62 items per year.
  - (vi) Acoustic Modems (M):
    - (A) M3 – an average of 112 hours per year.
    - (B) [Reserved]
  - (vii) Swimmer Detection Sonar (SD):
    - (A) SD1 – an average 2,341 hours per year.
- (2) Impulsive Source Detonations During Training and Testing:
  - (i) Explosive Classes:
    - (A) E1 (0.1 to 0.25 lb NEW) – an average of 10,140 detonations per year.
    - (B) E2 (0.26 to 0.5 lb NEW) – an average of 106 detonations per year.
    - (C) E3 (>0.5 to 2.5 lb NEW) – an average of 932 detonations per year.
    - (D) E4 (>2.5 to 5 lb NEW) – an average of 420 detonations per year.
    - (E) E5 (>5 to 10 lb NEW) – an average of 684 detonations per year.
    - (F) E6 (>10 to 20 lb NEW) – an average of 76 detonations per year.
    - (G) E8 (>60 to 100 lb NEW) – an average of 16 detonations per year.
    - (H) E9 (>100 to 250 lb NEW) – an average of 4 detonations per year.
    - (I) E10 (>250 to 500 lb NEW) – an average of 12 detonations per year.
    - (J) E11 (>500 to 650 lb NEW) – an average of 6 detonations per year.
    - (K) E12 (>650 to 2,000 lb NEW) – an average of 184 detonations per year.
  - (ii) [Reserved]

2.9.1.2 § 218.91 Effective dates and definitions.

(a) Regulations are effective [INSERT DATE OF FILING] through [INSERT DATE FIVE YEARS AFTER DATE OF FILING].

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

(1) *Uncommon Stranding Event (USE)* – A stranding event that takes place within an OPAREA where a Major Training Exercise (MTE) occurs and involves any one of the following:

(i) Two or more individuals of any cetacean species (not including mother/calf pairs), unless of species of concern listed in paragraph (b)(1)(ii) of this section found dead or live on shore within a 2-day period and occurring within 30 miles of one another.

(ii) A single individual or mother/calf pair of any of the following marine mammals of concern: beaked whale of any species, *Kogia* spp., Risso's dolphin, melon-headed whale, humpback whale, sperm whale, blue whale, fin whale, or sei whale.

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

(2) *Shutdown* – The cessation of active sonar operation or detonation of explosives within 14 nautical miles of any live, in the water, animal involved in a USE.

2.9.1.3 § 218.92 Permissible methods of taking.

(a) Under a Letter of Authorization (LOA) issued pursuant to § 218.97, the Holder of the Letter of Authorization may incidentally, but not intentionally, take marine mammals within the area described in § 218.90, provided the activity is in compliance with all terms, conditions, and requirements of these regulations and the appropriate LOA.

(b) The activities identified in § 218.90(c) must be conducted in a manner that minimizes, to the greatest extent practicable, any adverse impacts on marine mammals and their habitat.

(c) The incidental take of marine mammals under the activities identified in § 218.90(c) is limited to the following species, by the identified method of take:

(1) Level B Harassment for all Training and Testing Activities:

(i) Mysticetes:

(A) Blue whale (*Balaenoptera musculus*) – 140 (an average of 28 annually)

(B) Bryde's whale (*Balaenoptera edeni*) – 1,990 (an average of 398 annually)

(C) Fin whale (*Balaenoptera physalus*) - 140 (an average of 28 annually)

(D) Humpback whale (*Megaptera novaeangliae*) – 4,300 (an average of 860 annually)

(E) Minke whale (*Balaenoptera acutorostrata*) – 505 (an average of 101 annually)

(F) Sei whale (*Balaenoptera borealis*) - 1,595 (an average of 319 annually)

(G) Omura's whale (*Balaenoptera omurai*) - 515 (an average of 103 annually)

(ii) Odontocetes:

(A) Blainville's beaked whale (*Mesoplodon densirostris*) – 22,130 (an average of 4,426 annually)

(B) Bottlenose dolphin (*Tursiops truncatus*) – 3,705 (an average of 741 annually)

(C) Cuvier's beaked whale (*Ziphius cavirostris*) – 112,705 (an average of 22,541 annually)

(D) Dwarf sperm whale (*Kogia sima*) – 71,085 (an average of 14,217 annually)

(E) False killer whale (*Pseudorca crassidens*) – 2,775 (an average of 555 annually)

(F) Fraser's dolphin (*Lagenodelphis hosei*) – 12,860 (an average of 2,572 annually)

(G) Gingko-toothed beaked whale (*Mesoplodon ginkgodens*) – 19,485 (an average of 3,897 annually)

(H) Killer whale (*Orcinus orca*) – 420 (an average of 84 annually)

(I) Longman's beaked whale (*Indopacetus pacificus*) – 9,620 (an average of 1,924 annually)

(J) Melon-headed whale (*Peponocephala electra*) – 10,425 (an average of 2,085 annually)

(K) Pantropical spotted dolphin (*Stenella attenuata*) – 64,055 (an average of 12,811 annually)

(L) Pygmy killer whale (*Feresa attenuata*) – 525 (an average of 105 annually)

(M) Pygmy sperm whale (*Kogia breviceps*) – 27,895 (an average of 5,579 annually)

(N) Risso's dolphin (*Grampus griseus*) – 2,525 (an average of 505 annually)

(O) Rough-toothed dolphin (*Steno bredanensis*) – 9,095 (an average of 1,819 annually)

(P) Short-finned pilot whale (Globicephala macrorhynchus) – 9,075 (an average of 1,815 annually)

(Q) Sperm whale (*Physeter macrocephalus*) – 2,530 (an average of 506 annually)

(R) Spinner dolphin (*Stenella longirostris*) – 2,945 (an average of 589 annually)

(S) Striped dolphin (*Stenella coerulealba*) – 16,490 (an average of 3,298 annually)

(2) Level A Harassment for all Training and Testing Activities:

(i) Odontocetes:

(A) Dwarf sperm whale (*Kogia sima*) – 205 (an average of 41 annually)

(B) Pygmy sperm whale (*Kogia breviceps*) – 75 (an average of 15 annually)

2.9.1.4 § 218.93 Prohibitions.

Notwithstanding takings contemplated in § 218.92 and authorized by an LOA issued under §§ 216.106 and 218.97 of this chapter, no person in connection with the activities described in § 218.90 may:

(a) Take any marine mammal not specified in § 218.92(c);

(b) Take any marine mammal specified in § 218. 92(c) other than by incidental take as specified in § 218.92(c);

(c) Take a marine mammal specified in § 218.92(c) if such taking results in more than a negligible impact on the species or stocks of such marine mammal; or

(d) Violate, or fail to comply with, the terms, conditions, and requirements of these regulations or an LOA issued under §§ 216.106 and 218.97.

2.9.1.5 § 218.94 Mitigation.

(a) When conducting training and testing activities, as identified in § 218.90, the mitigation measures contained in the LOA issued under §§ 216.106 and 218.97 of this chapter must be implemented. These mitigation measures include, but are not limited to:

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

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

(ii) Lookouts positioned in aircraft or on boats will, to the maximum extent practicable and consistent with aircraft and boat safety and training and testing requirements, comply with the observation objectives described above in § 218.94 (a)(1)(i).

(iii) Lookout measures for non-impulse sound:

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

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

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

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

(iv) Lookout measures for explosives and impulse sound:

(A) Aircraft conducting IEER sonobuoy activities and explosive sonobuoy exercises will have one lookout.

(B) Surface vessels conducting anti-swimmer grenade activities will have one lookout.

(C) During general mine countermeasure and neutralization activities using up to a 20-lb net explosive weight detonation (bin E6 and below), vessels greater than 200 ft (61 m) will have two lookouts, while vessels less than 200 ft (61 m) or aircraft will have one lookout.

(D) Mine neutralization activities involving positive control diver-placed charges using up to a 20-lb net explosive weight detonation will have two lookouts. The divers placing the charges on mines will report all marine mammal sightings to their supporting small boat or Range Safety Officer.

(E) When mine neutralization activities using diver-placed charges with up to a 20-lb net explosive weight detonation are conducted with a time-delay firing device, four lookouts will be used. Two lookouts will be positioned in each of two small rigid hull inflatable boats. When aircraft are used, the pilot or member of the aircrew will serve as an additional lookout. The divers placing the charges on mines will report all marine mammal sightings to their supporting small boat or Range Safety Officer.

(F) Surface vessels or aircraft conducting small- or medium-caliber gunnery exercises against a surface target will have one lookout.

(G) Aircraft conducting missile exercises (including rockets) against surface targets will have one lookout.

(H) Aircraft conducting bombing exercises will have one lookout.

(I) During explosive torpedo testing, one lookout will be used and positioned in an aircraft.

(J) During sinking exercises, two lookouts will be used. One lookout will be positioned in an aircraft and one on a surface vessel.

(K) Surface vessels conducting explosive and non-explosive large-caliber gunnery exercises will have one lookout.

(v) Lookout measures for physical strike and disturbance:

(A) While underway, surface ships will have at least one lookout.

(B) During activities using towed in-water devices, that are towed from a manned platform, one lookout will be used.

(C) Non-explosive small-, medium-, and large-caliber gunnery exercises using a surface target will have one lookout.

(D) Non-explosive bombing exercises will have one lookout.

(E) Aircraft conducting non-explosive missile exercises against a surface target will have one lookout.

(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-impulse 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 during the activity) if any visually detected marine mammals are within 1,000 yd (914 m) of the source (i.e., the bow).

(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 during the activity) if any detected marine mammals are sighted within 500 yd (457 m) of the source.

(C) The Navy shall ensure that low-frequency and hull-mounted mid-frequency active sonar transmissions (for sources that can be turned off during the activity) are ceased if any visually detected marine mammals are within 200 yd (183 m) of the sonar dome. Active transmission will recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone; (2) 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; (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes; (4) the ship has transited more than 2,000 yd. (1.8 kilometers [km]) beyond the location of the last sighting; or (5) 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).

(D) If the source is not able to be powered down during the activity (e.g., low-frequency sources within bins LF4 and LF5), mitigation will involve ceasing active transmission if a marine mammal is sighted within 200 yd. (183 m). Active transmission will recommence if any one of the following conditions is met: (1) the animal is observed existing the mitigation zone; (2) 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; (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes; or (4) the ship has transited more than 400 yd. (366 m) beyond the location of the last sighting.

(E) With the exception of activities involving platforms operating at high altitudes, when marine mammals are visually detected, the Navy shall ensure that high-frequency and non-hull-mounted mid-frequency active sonar transmission (for sources that can be turned off during the activity) is ceased if any visually detected marine mammals are within 200 yd (183 m) of the source. Active transmission will recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) 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, (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes for an aircraft-deployed source, (4) the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a vessel-deployed source, (5) the vessel or aircraft has repositioned itself more than 400 yd. (366 m) away from the location of the last sighting, or (6) the vessel concludes that dolphins are deliberately closing in to ride the vessel's bow wave (and there are no other marine mammal sightings within the mitigation zone).

(iv) Mitigation zones for explosive and impulse sound:

(A) A mitigation zone with a radius of 600 yd (549 m) shall be established for IEER sonobuoys (bin E4). Mitigation would include pre-exercise aerial observation and passive acoustic monitoring, which would begin 30 minutes before the first source/receiver pair detonation and continue throughout the duration of the exercise. The pre-exercise aerial observation would include the time it takes to deploy the sonobuoy pattern (deployment is conducted by aircraft dropping sonobuoys in the water). Explosive detonations would cease if a marine mammal is sighted within the mitigation zone. Detonations would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes.

Passive acoustic monitoring would be conducted with Navy assets, such as sonobuoys, already participating in the activity. These assets would only detect vocalizing marine mammals within the frequency bands monitored by Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot provide locations of these animals. Passive acoustic detections would be reported to lookouts posted in aircraft and on vessels in order to increase vigilance of their visual observation.

(B) A mitigation zone with a radius of 350 yd (320 m) shall be established for explosive sonobuoys using 0.5-2.5 lb net explosive weight (bin E3). Mitigation would include pre-exercise aerial monitoring during deployment of the field of sonobuoy pairs (typically up to 20 minutes) and continuing throughout the duration of the exercise within a mitigation zone of 350 yd (320 m) around an explosive sonobuoy. Explosive detonations would cease if a marine mammal is sighted within the mitigation zone. Detonations would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes.

Passive acoustic monitoring would also be conducted with Navy assets, such as sonobuoys, already participating in the activity. These assets would only detect vocalizing marine mammals within the frequency bands monitored by Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot provide locations of these animals. Passive acoustic detections would be reported to lookouts posted in aircraft in order to increase vigilance of their visual observation.

(C) A mitigation zone with a radius of 200 yd (183 m) shall be established for anti-swimmer grenades (bin E2). Mitigation would include visual observation from a small boat immediately before and during the exercise within a mitigation zone of 200 yd (183 m) around an anti-swimmer grenade. Explosive detonations would cease if a marine mammal is sighted within the mitigation zone. Detonations would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes, or (4) the activity has been repositioned more than 400 yd (366 m) away from the location of the last sighting.

(D) A mitigation zone ranging from 350 yd (320 m) to 800 yd (732 m), dependent on charge size and if the activity involves the use of diver-placed charges, shall be established for mine countermeasure and neutralization activities using positive control firing devices. Mitigation zone distances are specified for charge size in Table 7 of the preamble.

During general mine countermeasure and neutralization activities, mitigation would include visual observation from one or more small boats or aircraft beginning 30 minutes before, during, and 30 minutes after (when helicopters are not involved in the activity) or 10 minutes before, during, and 10 minutes after (when helicopters are involved in the activity) the completion of the exercise within the mitigation zones around the detonation site.

For activities involving diver-placed charges, visual observation would be conducted by either two small boats, or one small boat in combination with one helicopter. Boats would position themselves near the mid-point of the mitigation zone radius (but always outside the detonation plume radius and human safety zone) and travel in a circular pattern around the detonation location. When using two boats, each boat would be positioned on opposite sides of the detonation location, separated by 180 degrees. If used, helicopters would travel in a circular pattern around the detonation location.

For both general and diver-placed positive control mine countermeasure and neutralization activities, explosive detonations will cease if a marine mammal is sighted within the mitigation zone. Detonations will recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) 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, (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes, when helicopters are not involved in the activity or (4) the mitigation zone has been clear from any additional sightings for a period of 10 minutes when helicopters are involved in the activity.

(E) A mitigation zone with a radius of 1,000 yd (914 m) shall be established for mine countermeasure and neutralization activities using diver-placed time-delay firing devices (bin E6). Mine neutralization activities involving diver-placed charges would not include time-delay longer than 10 minutes. Mitigation would include visual observation from small boats or aircraft commencing 30 minutes before, during, and until 30 minutes after the completion of the exercise within a mitigation zone of 1,000 yd (914 m) around the detonation site. During activities using time-delay firing devices involving up to a 20 lb net explosive weight charge, visual observation will take place using two small boats. Fuse initiation would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes.

Survey boats would position themselves near the mid-point of the mitigation zone radius (but always outside the detonation plume radius/human safety zone) and travel in a circular pattern around the detonation location. One lookout from each boat would look inward toward the detonation site and the other lookout would look outward away from the detonation site. When using two small boats, each boat would be positioned on opposite sides of the detonation location, separated by 180 degrees. If available for use, helicopters would travel in a circular pattern around the detonation location.

(F) 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). Mitigation would include visual observation from a vessel or aircraft immediately before and during the exercise within a mitigation zone of 200 yd (183 m) around the intended impact location. Vessels would observe the mitigation zone from the firing position. When aircraft are firing, the aircrew would maintain visual watch of the mitigation zone during the activity. Firing would cease if a marine mammal is sighted within the mitigation zone. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes for a firing aircraft, (4) the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a firing vessel, or (5) the intended target location has been repositioned more than 400 yd (366 m) away from the location of the last sighting.

(G) 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). Mitigation would include visual observation from a ship immediately before and during the exercise within a mitigation zone of 600 yd (549 m) around the intended impact location. Ships would observe the mitigation zone from the firing position. Firing would cease if a marine mammal is sighted within the mitigation zone. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes.

(H) A mitigation zone with a radius of 900 yd (823 m) around the deployed target shall be established for missile exercises involving aircraft firing up to 250 lb net explosive weight using and a surface target (bin

E9). When aircraft are firing, mitigation would include visual observation by the aircrew or supporting aircraft prior to commencement of the activity within a mitigation zone of 900 yd (823 m) around the deployed target. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes or 30 minutes (depending on aircraft type).

(I) A mitigation zone with a radius of 2,000 yd (1.8 km) shall be established for missile exercises involving aircraft firing >250 to 500 lb net explosive weight using and a surface target (bin E10). When aircraft are firing, mitigation would include visual observation by the aircrew prior to commencement of the activity within a mitigation zone of 2,000 yd (1.8 km) around the intended impact location. Firing would cease if a marine mammal is sighted within the mitigation zone. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes or 30 minutes (depending on aircraft type).

(J) A mitigation zone with a radius of 2,500 yd (2.3 km) shall be established for bombing exercises (bin E12). Mitigation would include visual observation from the aircraft immediately before the exercise and during target approach within a mitigation zone of 2,500 yd (2.3 km) around the intended impact location. Bombing would cease if a marine mammal is sighted within the mitigation zone. Bombing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes.

(K) A mitigation zone with a radius of 2,100 yd (1.9 km) shall be established for torpedo (explosive) testing (except for aircraft operating at high altitudes) (bin E11). Mitigation would include visual observation by aircraft immediately before, during, and after the exercise within a mitigation zone of 2,100 yd (1.9 km) around the intended impact location. Firing would cease if a marine mammal is sighted within the mitigation zone. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes or 30 minutes (depending on aircraft type).

In addition to visual observation, passive acoustic monitoring would be conducted with Navy assets, such as passive ships sonar systems or sonobuoys, already participating in the activity. Passive acoustic observation would be accomplished through the use of remote acoustic sensors or expendable sonobuoys, or via passive acoustic sensors on submarines when they participate in the proposed action. These assets would only detect vocalizing marine mammals within the frequency bands monitored by Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot provide locations of these animals. Passive acoustic detections would be reported to the lookout posted in the aircraft in order to increase vigilance of the visual observation and to the person in control of the activity for their consideration in determining when the mitigation zone is free of visible marine mammals.

(L) A mitigation zone with a radius of 2.5 nautical miles around the target ship hulk shall be established for sinking exercises (bin E12). Mitigation would include aerial observation beginning 90 minutes before the first firing, visual observations from vessels throughout the duration of the exercise, and both aerial and vessel observation immediately after any planned or unplanned breaks in weapons firing of longer than 2 hours. Prior to conducting the exercise, the Navy would review remotely sensed sea surface temperature and sea surface height maps to aid in deciding where to release the target ship hulk.

The Navy would also monitor using passive acoustics during the exercise. Passive acoustic monitoring would be conducted with Navy assets, such as passive ships sonar systems or sonobuoys, already participating in the activity. These assets would only detect vocalizing marine mammals within the frequency bands monitored by Navy personnel. Passive acoustic detections would not provide range or bearing to detected animals, and therefore cannot provide locations of these animals. Passive acoustic detections would be reported to lookouts posted in aircraft and on vessels in order to increase vigilance of their visual observation. Lookouts will also increase observation vigilance before the use of torpedoes or unguided ordnance with a net explosive weight of 500 lb or greater, or if the Beaufort sea state is a 4 or above.

The exercise would cease if a marine mammal is sighted within the mitigation zone. The exercise would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes. Upon sinking the vessel, the Navy would conduct post-exercise visual observation of the mitigation zone for 2 hours (or until sunset, whichever comes first).

(M) 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 conducted from a ship. Firing would cease if a marine mammal is sighted within the mitigation zone. Firing would recommence if any one of the following conditions is met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, (3) the mitigation zone has been clear from any additional sightings for a period of 30 minutes, or (4) the vessel has repositioned itself more than 140 yd (128 m) away from the location of the last sighting.

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

(A) 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) 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 non-explosive small-, medium-, and large-caliber gunnery exercises using a surface target. Mitigation would include visual observation immediately before and during the exercise within a mitigation zone of 200 m around the intended impact location. Firing would cease if a marine mammal is visually detected within the mitigation zone. Firing would recommence if any one of the following conditions are met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes for a firing aircraft, (4) the mitigation zone has been clear from any additional sightings for a period of 30 minutes for a firing vessel, or (5) the intended target location has been repositioned more than 400 yd (366 m) away from the location of the last sighting and the animal's estimated course direction.

(B) A mitigation zone of 1,000 yd (914 m) shall be established for non-explosive bombing exercises. Mitigation would include visual observation from the aircraft immediately before the exercise and during target approach within a mitigation zone of 1000 yd (914 m) around the intended impact location. Bombing would cease if a marine mammal is visually detected within the mitigation zone. Bombing would recommence if any one of the following conditions are met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes.

(C) When aircraft are firing, a mitigation zone of 900 yd (823 m) around the deployed target shall be established for non-explosive missile exercises (including rockets) using a surface target. Firing would cease if a marine mammal is visually detected within the mitigation zone. Firing would recommence if any one of the following conditions are met: (1) the animal is observed exiting the mitigation zone, (2) the animal is thought to have exited the mitigation zone based on its course and speed and the relative motion between the animal and the source, or (3) the mitigation zone has been clear from any additional sightings for a period of 10 minutes or 30 minutes (depending on aircraft type).

(3) Stranding Response Plan:

(i) The Navy shall abide by the letter of the "Stranding Response Plan for Major Navy Training Exercises in the MITT Study Area," to include the following measures:

(A) Shutdown Procedures – When an Uncommon Stranding Event (USE - defined in § 218.91) occurs during a Major Training Exercise (MTE) in the MITT Study Area, the Navy shall implement the procedures described below.

> (*1*) The Navy shall implement a shutdown (as defined § 218.91) when advised by a NMFS Office of Protected Resources Headquarters Senior Official designated in the MITT 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.

(*5*) Within 72 hours of NMFS notifying the Navy of the presence of a USE, the Navy shall provide available information to NMFS (per the MITT 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.

2.9.1.6 § 218.95 Requirements for monitoring and reporting.

(a) As outlined in the MITT 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 § 218.90 is thought to have resulted in the mortality or injury of any marine mammals, or in any take of marine mammals not identified in § 218.91.

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

(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

(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 MITT Monitoring Program Report – (1) The Navy shall submit an annual report describing the implementation and results of the MITT Monitoring Program, described in § 218.95. Data standards will be consistent to the extent appropriate 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 MITT Monitoring Program shall, at a minimum, provide the same marine mammal observation data required in § 218.95. (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 multiple 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. The report shall be submitted either 90 days after the calendar year, or 90 days after the conclusion of the monitoring year date to be determined by the Adaptive Management process.

(f) Sonar Exercise Notification – The Navy shall submit to NMFS (specific contact information to be provided in the LOA) either an electronic (preferably) or verbal report within 15 calendar days after the completion of any major exercise indicating:
(i) Location of the exercise.
(ii) Beginning and end dates of the exercise.
(iii) Type of exercise.

(g) Annual MITT Exercise and Testing Report – 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 a detailed report 3 months after the anniversary of the date of issuance of the LOA. The detailed annual report 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 report will be based on the accumulation of data from the current year's report and data collected from previous reports. The detailed report shall contain information identified in § 218.95 (e)(1-5).

(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 Multi-Strike Group Exercise (Valiant Shield).
    (B) Joint Expeditionary Exercise
(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 information for each sighting when mitigation occurred during each MTE:
        (*1*) Date/time/location of sighting.
        (*2*) Species (if not possible, indication of whale/dolphin).
        (*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 to 500 yd, 500 to 1,000 yd, 1,000 to 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 animal'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 animal closing to bow ride, paralleling course/speed, floating on surface and not swimming, etc.) and if any calves present.

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

(iv) 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, 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).

(*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 SINEKX and net explosive weight (NEW) of each weapon and the combined 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 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; and
        (C) Improved Extended Echo-Ranging System (IEER)/sonobuoy summary, including:
            (*1*) Total expended/detonated rounds (buoys).
            (*2*) Total number of self-scuttled IEER rounds.
            (3) 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.
            (4) 5-year Close-out Exercise and Testing Report – This report will be included as part of the 2020 annual exercise or testing report. This report will provide the annual totals for each sound source bin with a comparison to the annual allowance and the 5-year total for each sound source bin with a comparison to the 5-year allowance. Additionally, if there were any changes to the sound source allowance, this report will include a discussion of why the change was made and include the analysis to support how the change did or did not result in a change in the 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.

2.10 NMFS' Issuance of Letters of Authorization (LOA) Pursuant to the MMPA Regulations

Subsequent to promulgation of the MMPA Rule, NMFS' Permits Division issued an LOA for Navy training and testing activities for the five year period (August 2015 through August 2020). The LOA set forth:

(1) Permissible methods of incidental taking;

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

(3) Requirements for mitigation, monitoring and reporting.

Issuance of the LOA was dependent on a determination that the total number of marine mammals taken by the activity as a whole would have no more than a negligible impact on the affected species or stock of marine mammals. Incidental take of marine mammals authorized by the LOA is also provided in the MMPA Rule and thus may not exceed the levels published in the Rule and assessed in this opinion. Mitigation requirements are the same as those described in the MMPA rule as provided in the previous section and as assessed in this opinion. Any unforeseen changes in the incidental take authorization above what is authorized in the MMPA Rule and LOA, would require amending the Rule and reinitiation of section 7 consultation pursuant to the ESA.

Section 7(b)(4)(C) of the ESA provides that if an endangered or threatened marine mammal is involved, the taking must first be authorized by Section 101(a)(5) of the MMPA. Accordingly, the terms of this incidental take statement and the exemption from Section 9 prohibitions of the ESA become effective only upon the issuance of MMPA authorization (i.e, five year regulations and LOA) to take the marine mammals in this Opinon. This authorization was issued July 31, 2015. Absent such authorization, the incidental take statement would be inoperative for marine mammals.

2.11 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 CFR 402.02). The action area encompasses the MITT Study Area, transit corridor and area outsides of the study area where direct and indirect effects of stressors from Navy training and testing activities could be experienced.

The MITT action area is composed of the established ranges (at-sea ranges and land based training areas on Guam and CNMI), operating areas, and special use airspace in the region of the Mariana Islands that are part of the Mariana Islands Range Complex (MIRC) and its surrounding seas, and includes a transit corridor.[3] The MITT action area does not include the territorial waters of Palau or Micronesia. The transit corridor is outside the geographic boundaries of the MIRC and is a direct route across the high seas for Navy ships in transit between the MIRC and the HRC. The Proposed Action also includes pierside sonar maintenance and testing alongside Navy piers located in Inner Apra Harbor. The MITT action area is depicted in Figure 1.

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[3] Vessel transit corridors are the routes typically used by Navy ships to traverse from one area to another. The route depicted in Figure 1 is a direct route between the MIRC and the HRC, making it a quick and fuel-efficient transit. The depicted transit corridor is notional and may not represent actual routes used. Actual routes navigated are based on a number of factors including, but not limited to, weather and training; however, the corridor represents the environment potentially impacted by the Proposed Action.

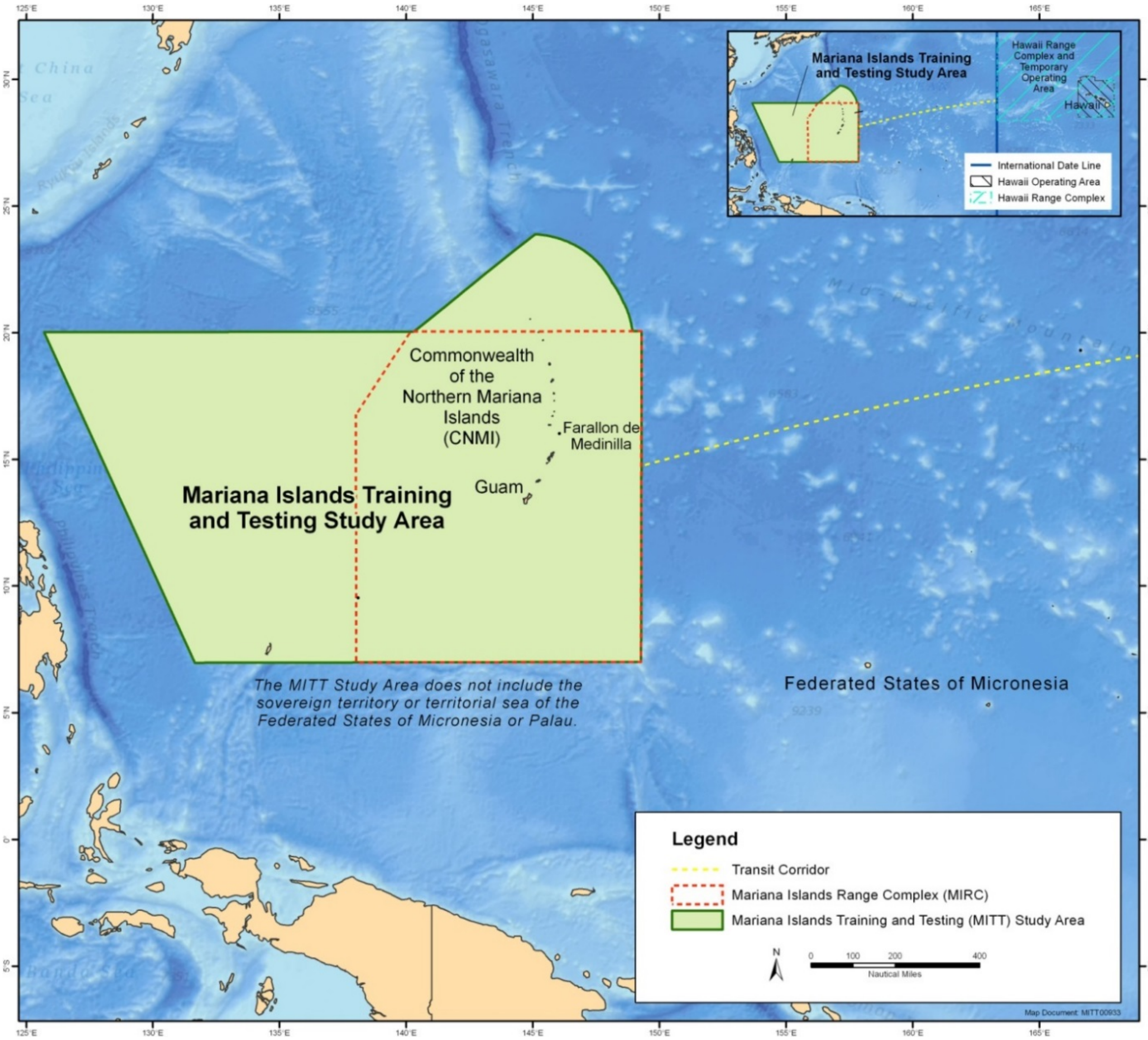

Figure 1. Mariana Islands Action Area with the Mariana Islands Range Complex and notional transit corridor.

A range complex is a designated set of specifically bounded geographic areas that encompasses a water component (above and below the surface) and airspace, and may encompass a land component where training and testing of military platforms, tactics, munitions, explosives, and electronic warfare systems occurs. Range complexes include established ocean operating areas and special use airspace, which may be further divided to provide better control of the area and activities for safety reasons.

The MITT action area includes the MIRC land training areas and at-sea operating areas that were previously addressed in the MIRC EIS/OEIS (May 2010) with modifications to the special use air space that were addressed in the MIRC Airspace Environmental Assessment (EA)/Overseas EA (OEA) (U.S. Department of the Navy 2013), and the seaward extension of the northern and western edges of the MIRC study area, the transit corridor, and Navy pierside locations in the Apra Harbor Naval Complex.

2.11.1 Mariana Islands Range Complex

The MIRC includes land training areas, ocean surface and subsurface areas, and special use airspace. These areas extend from the waters south of Guam to north of Pagan (CNMI), and from the Pacific Ocean east of the Mariana Islands to the Philippine Sea to the west, encompassing 501,873 square nautical miles (nm^2) of open ocean.

2.11.1.1 Special Use Airspace and Air Traffic Controlled Assigned Airspace

The MIRC is anticipated to include approximately 70,000 nm^2 of special use airspace and Air Traffic Control Assigned Airspace (ATCAA; once Federal Aviation Administration [FAA] rule-making and non-rule making airspace changes are complete[4]). As depicted in Figure 2 and Figure 3, this airspace is almost entirely over the ocean (except ATCAA 6 and W-13A) and includes warning areas, ATCAAs, and restricted areas. Warning Area (W)-517 and W-12 include approximately 11,769 nm^2 of special use airspace; W-11 (A/B) is approximately 10,467 nm^2 of special use airspace, and W-13 (A/B/C) is approximately 13,752 nm^2 of special use airspace. The ATCAAs of the MIRC account for more than 28,750 nm^2 of airspace and includes ATCAA 5 and ATCAA 6 (Figure 2).

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[4] The MIRC Airspace EA/OEA tiered off from the MIRC EIS/OEIS; the Navy analyzed the potential impacts of redesignating ATCAAs in the MITT Study Area with Warning Areas and expanding the special use airspace around FDM. In that EA/OEA, no new training or testing events were proposed. The EA/OEA concluded that no significant impacts to the environment would occur as a result of the airspace redesignation and expansion. The FAA has rule-making and non-rule making authority for the airspace redesignation and expansion, and the MIRC Airspace EA/OEA supported the FAA in its rule-making and non-rule making process to establish special use airspace which is anticipated to be established in the summer of 2017.

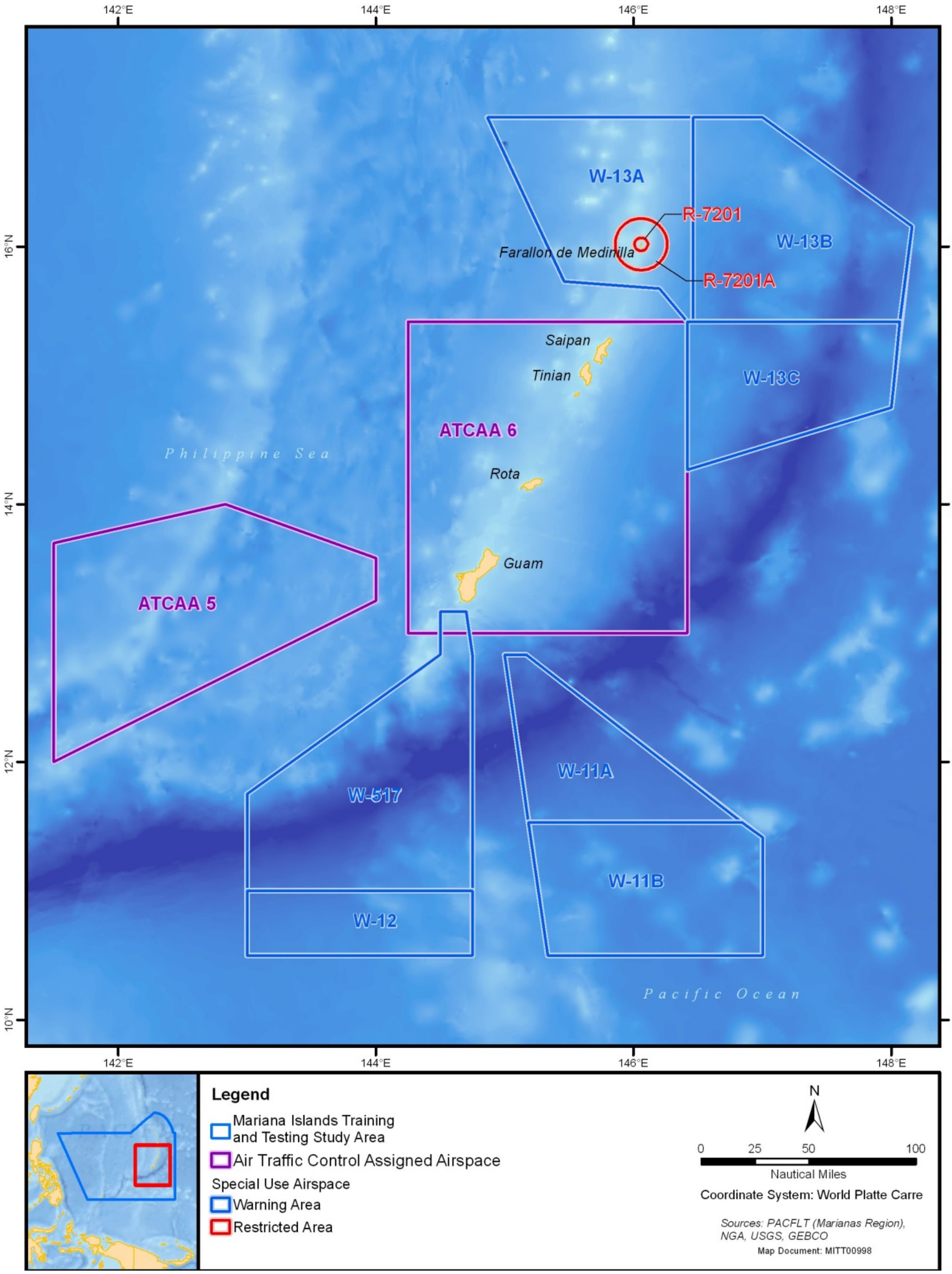


Figure 2. Mariana Islands Range Complex Airspace.

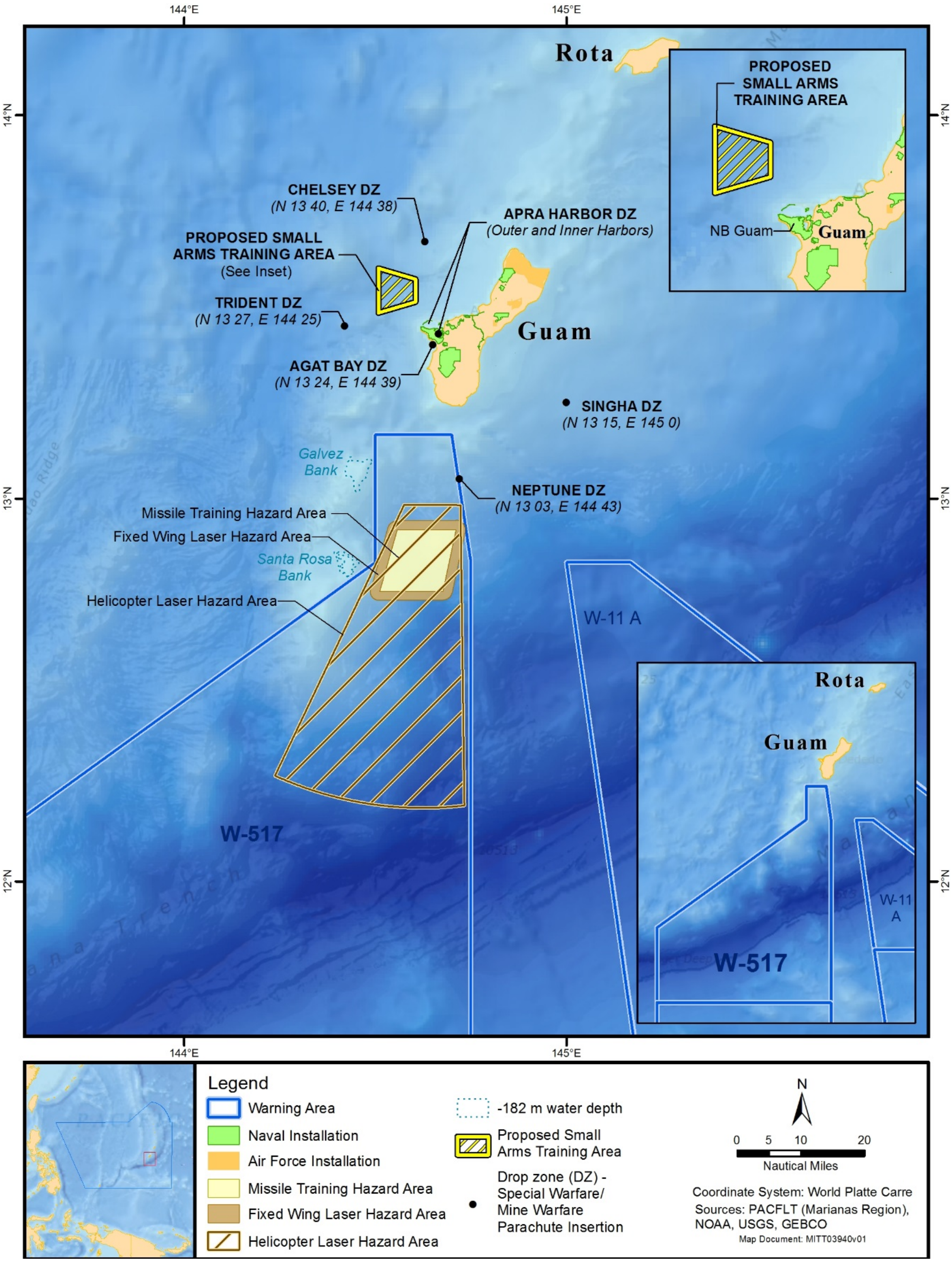


Figure 3. Mariana Islands Range Complex Airspace and Warning Area 517.

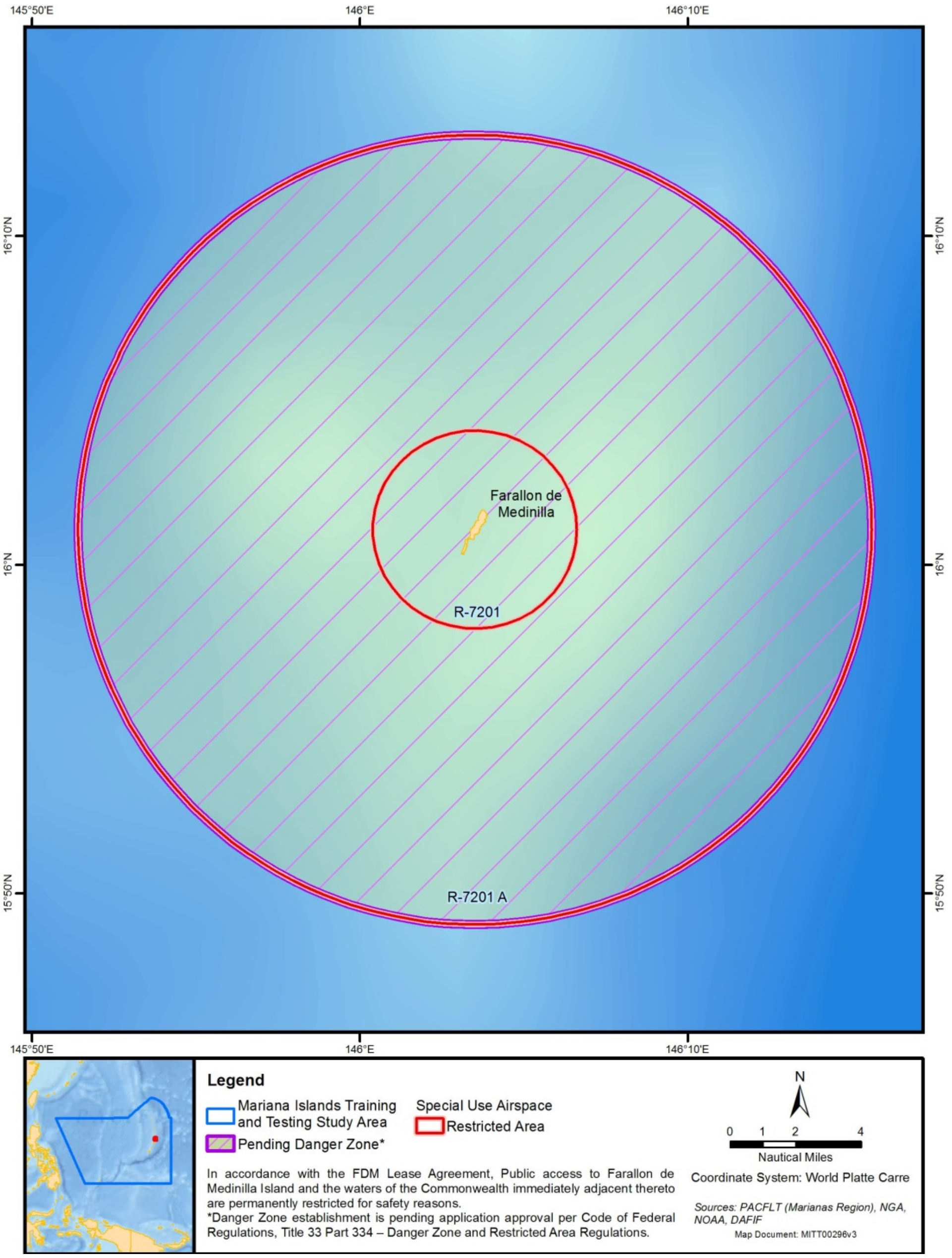
The restricted area airspace over or near land areas within the MIRC makes up 452 nm^2 of special use airspace and includes restricted areas (R)-7201 and R-7201A which extends in a 12 nm radius around Farallon de Medinilla (FDM) (Figure 4).

2.11.1.2 Sea and Undersea Space

The MIRC includes the sea and undersea space from the ocean surface to the ocean floor. The MIRC includes designated sea and undersea space training sites to include designated drop zones, underwater demolition and floating mine exclusion zones, danger zones associated with live fire ranges, and training areas associated with military controlled beaches, harbors, and littoral areas.

W-517 (Figure 3) is special use airspace where the sea space underneath is also restricted from public access during hazardous training and testing events. Portions of the Marianas Trench Marine National Monument, established in January 2009 by Presidential Proclamation under the authority of the Antiquities Act (16 U.S. Code §§431–433), lie within the MIRC. The prohibitions required by the Proclamation do not apply to activities and exercises of the Armed Forces (including those carried out by the U.S. Coast Guard).

The MIRC Airspace EA/OEA proposed and analyzed a Danger Zone around FDM. The Army Corps of Engineers has rule-making authority for Danger Zone establishment. The pending Danger Zone rule for FDM extends out 12 nm from a center point on FDM and over a range hazard area of approximately 452 nm^2 (Figure 2 and Figure 4).



In accordance with the FDM Lease Agreement, Public access to Farallon de Medinilla Island and the waters of the Commonwealth immediately adjacent thereto are permanently restricted for safety reasons.

*Danger Zone establishment is pending application approval per Code of Federal Regulations, Title 33 Part 334 – Danger Zone and Restricted Area Regulations.

Sources: PACFLT (Marianas Region), NGA, NOAA, DAFIF

Map Document: MITT00296v3

Figure 4. Farallon de Medinilla Restricted Area 7201, 7201A, and Danger Zone.

2.11.1.3 Land

Commander Joint Region Marianas provides executive level installation management support to all DoD components and tenants through assigned regional installations on Guam and the Commonwealth of the Northern Mariana Islands in support of training and testing in the Marianas, including coordination with Northern Mariana Islands Commonwealth Port Authority for logistic and operational support of DoD aircraft and vessels; acts as the interface between the Navy and the civilian community; ensures compliance with all environmental laws and regulations, safety procedures, and equal opportunity policy; and performs other functions and tasks as assigned. While land based activities are not part of this consultation, a description of installations on Guam and the CNMI are provided for informational purposes only.

2.11.1.3.1 Guam

The Navy has control of approximately 28 square miles (mi.^{2}) (72.5 square kilometers [km^2]) of land in noncontiguous properties on Guam. There are five Navy annexes: Main Base (which includes Apra Harbor Naval Complex and Main Base/Polaris Point) (Figure 5), Naval Base Guam Munitions Site (Figure 6); Hospital Annex/Nimitz Hill, Naval Base Guam Telecommunications Site (Figure 7), and Naval Base Guam Barrigada (Figure 8).

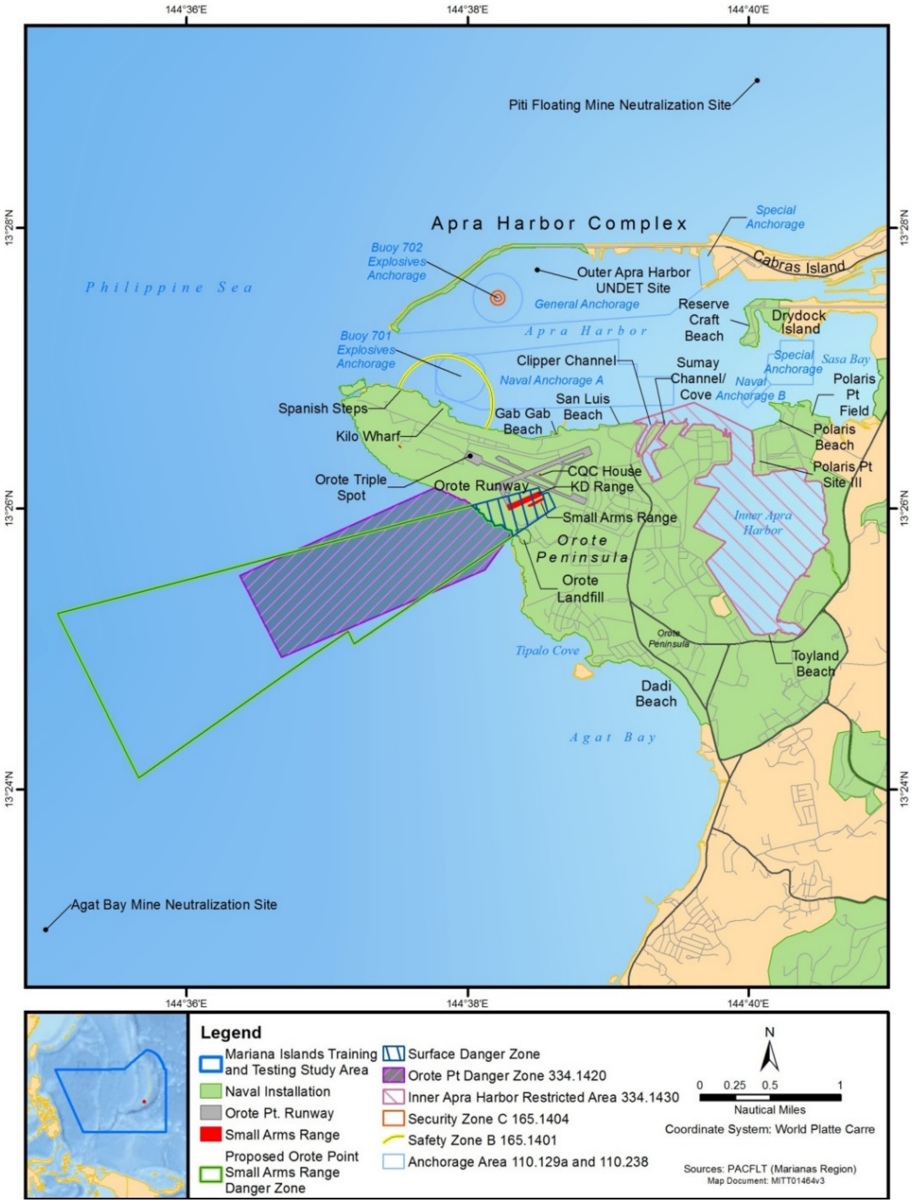


Figure 5. Apra Harbor Naval Complex (Main Base) and Main Base/Polaris Point.

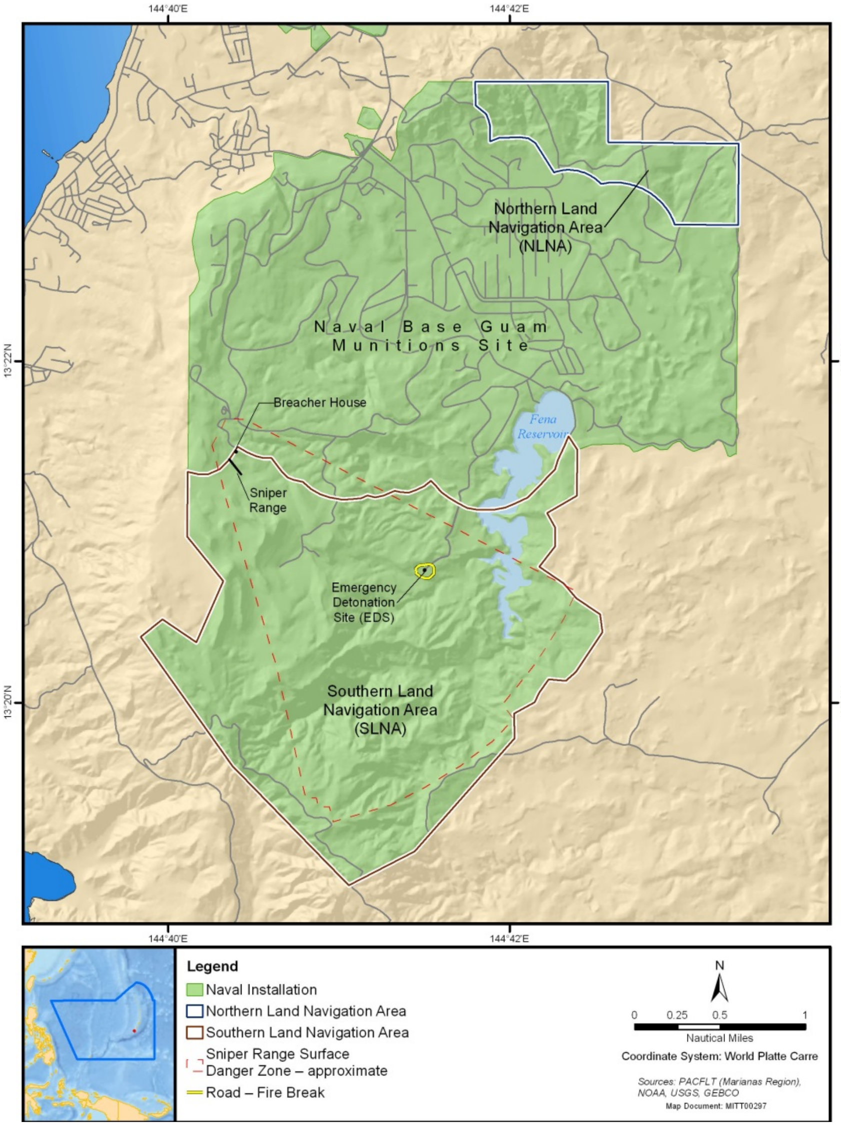


Figure 6. Naval Base Guam Munitions Site

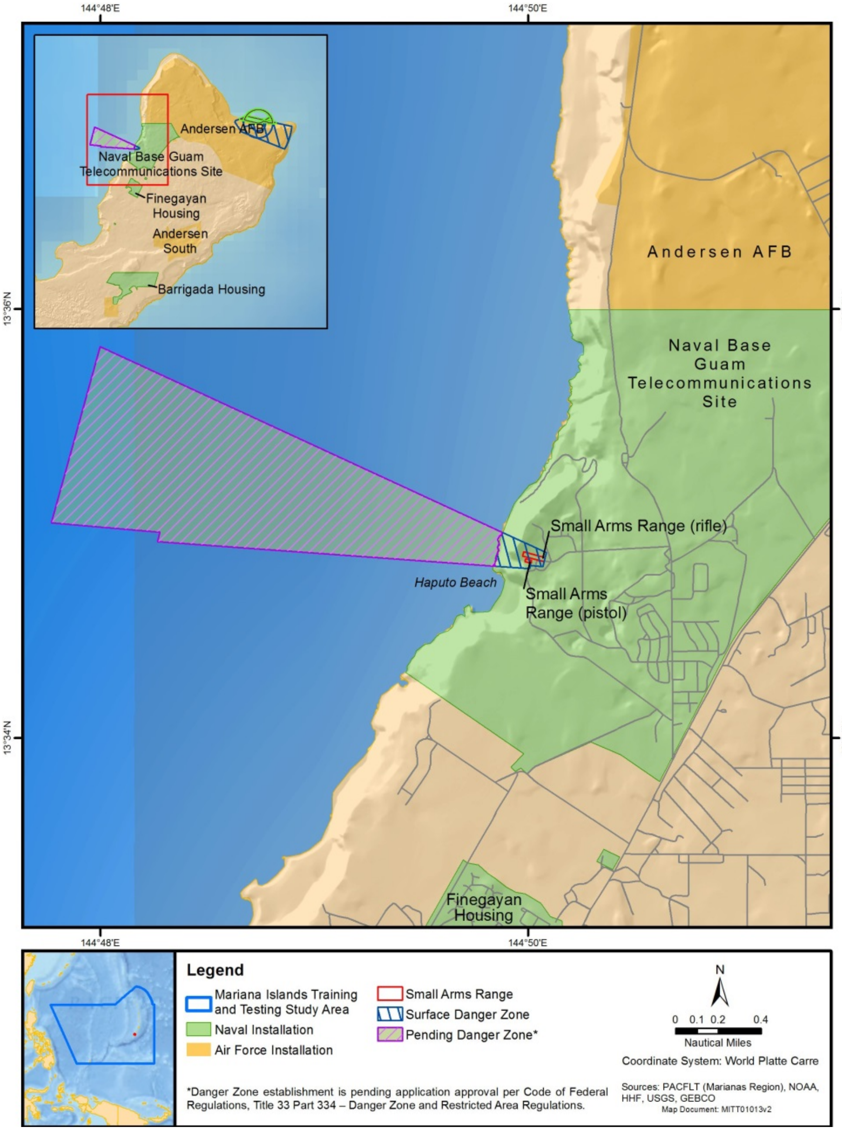


Figure 7. Naval Base Guam Telecommunications Site

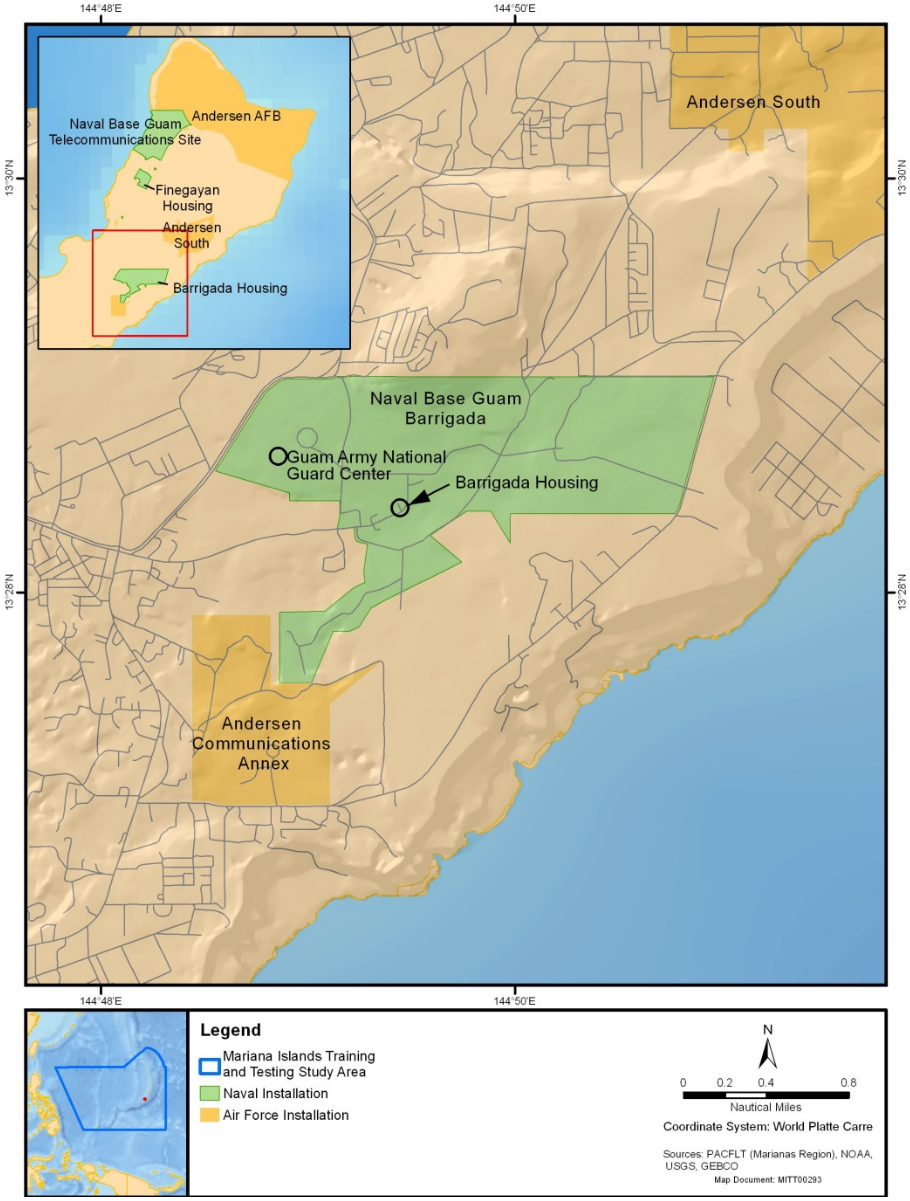


Figure 8. Naval Base Guam Barrigada

Andersen Air Force Base, one of the largest U.S. Air Force airfields, is located in the northern portion of the island of Guam. Andersen Air Force Base includes the main base and Northwest Field which covers 24.5 mi.^{2} (63.5 km^2), Andersen South 3.2 mi.^{2} (8.3 km^2), and Andersen Barrigada Annex 0.7 mi.^{2} (1.8 km^2) (Figure 9).

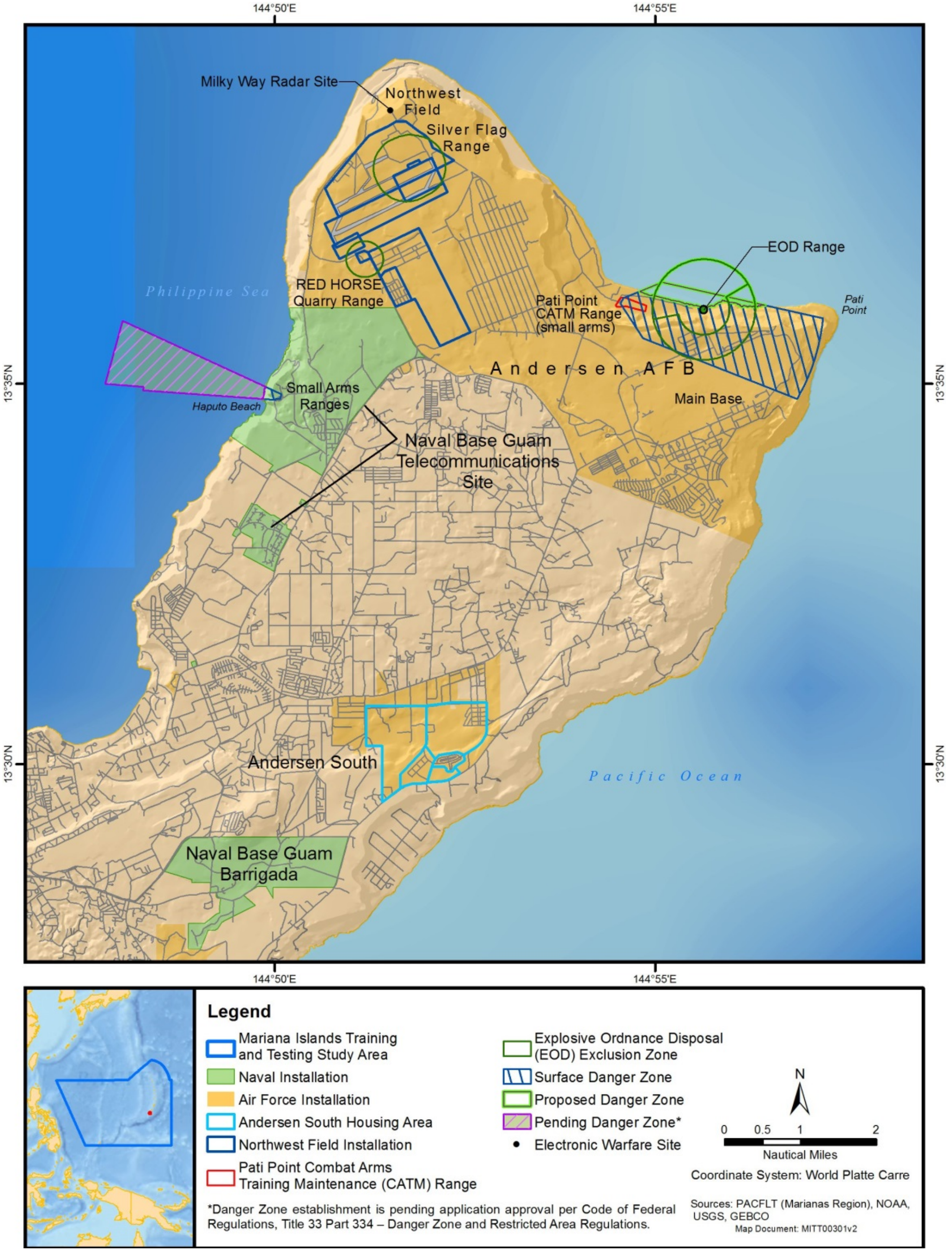
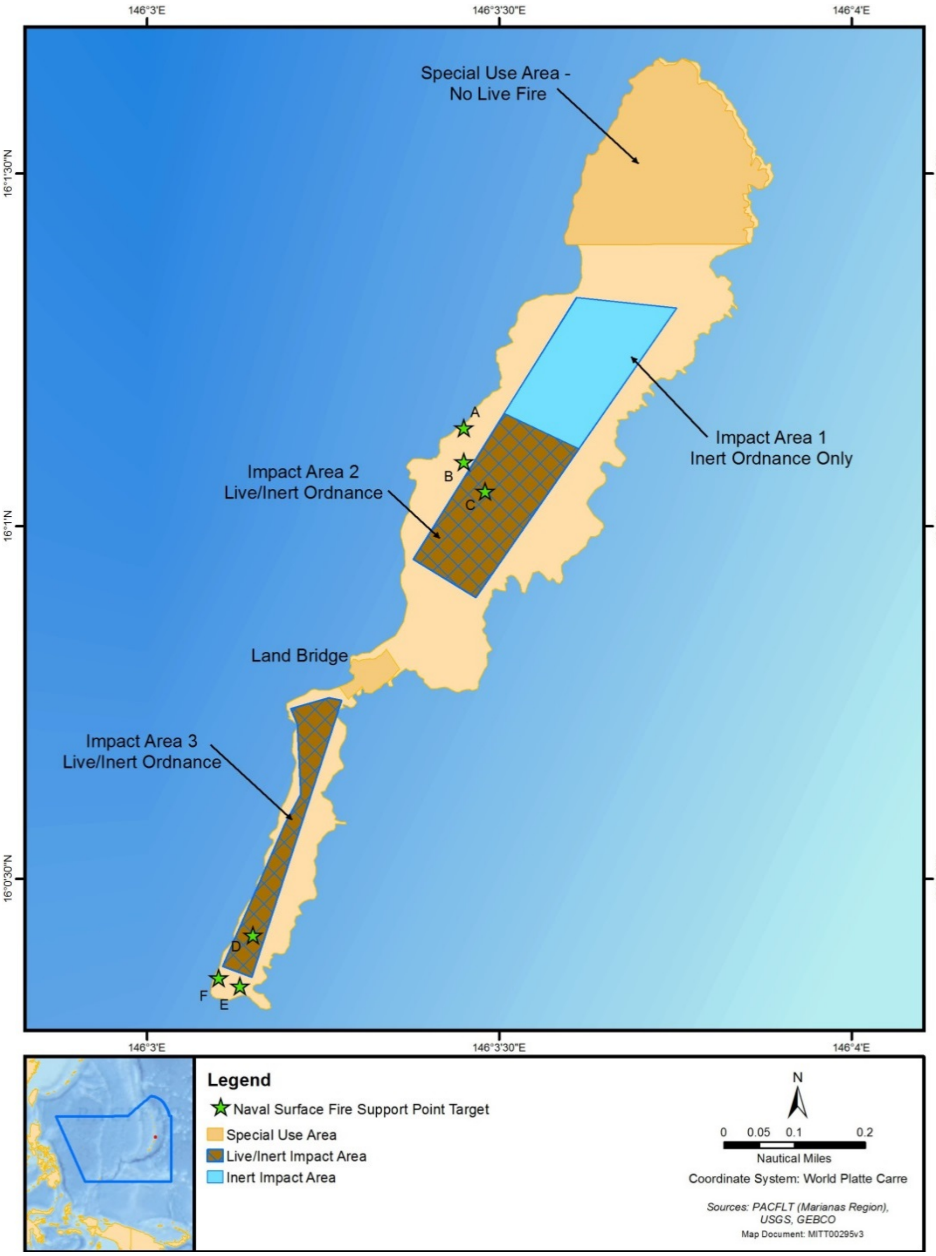


Figure 9. Andersen Air Force Base

2.11.1.3.2 Commonwealth of the Northern Mariana Islands

No DoD personnel are permanently stationed in the CNMI, with the exception of a U.S. Army Reserve unit located on Saipan.

Farallon de Medinilla (FDM). FDM is a rocky and uninhabited island, approximately 1.7 miles (mi.) (2.7 kilometer [km]) long and 0.3 mi. (0.5 km) wide (Figure 10). The DoD leases FDM for use as a live and inert gunnery, missile, and bombing range.



Sources: PACFLT (Marianas Region), USGS, GEBCO

Figure 10. Farallon de Medinilla

Tinian. Tinian has a land area of approximately 39 mi.^{2} (101 km^2). The DoD leases approximately 15,347 contiguous acres (6,210.7 hectares) of northern Tinian (the Military Lease Area) for field training (Figure 11). The Military Lease Area is further divided into the Exclusive Military Use Area and the Leaseback Area.

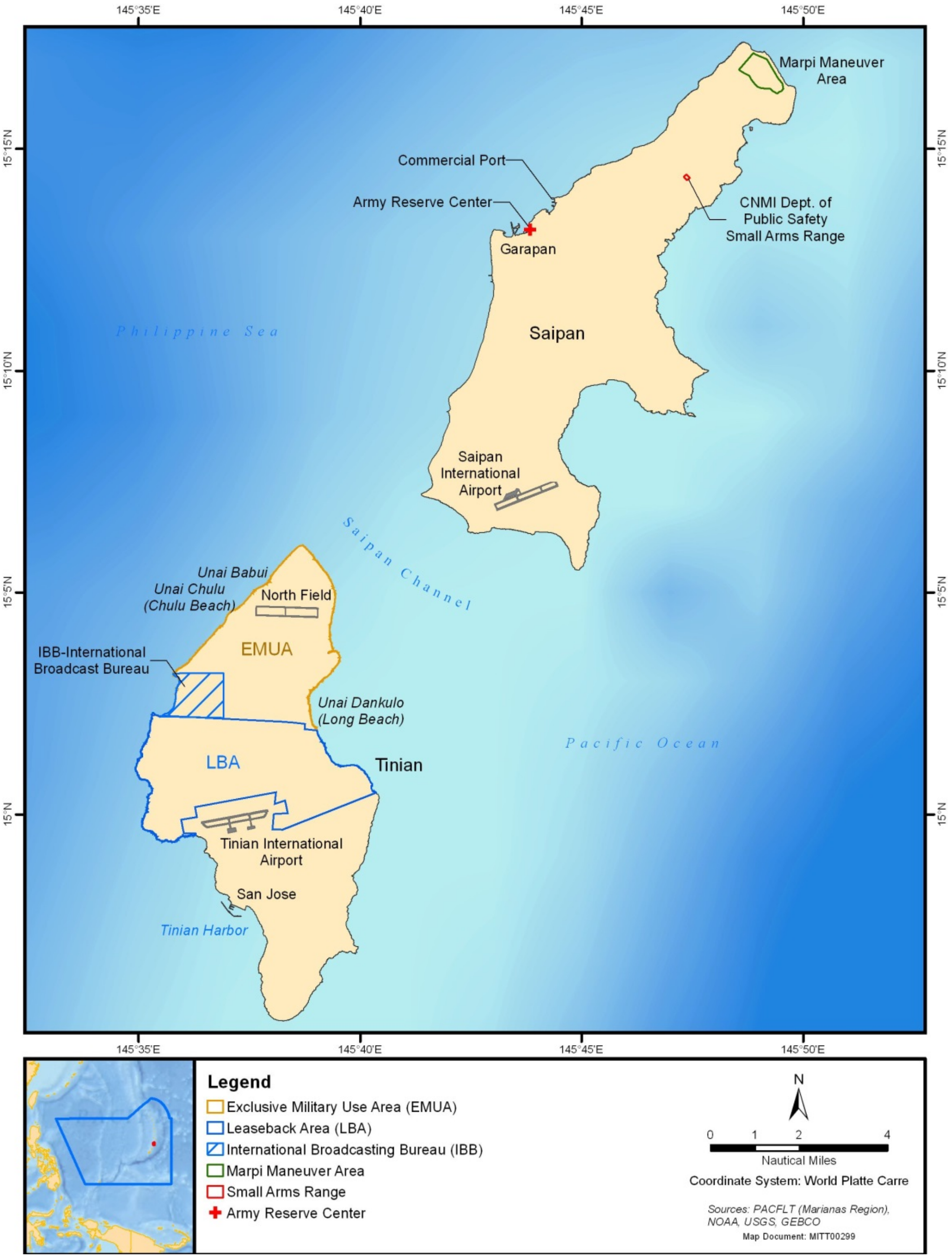


Figure 11. Tinian and Saipan

Saipan. Approximately 0.28 mi.^{2} (0.73 km^2) on Tanapag Harbor (commercial port) is leased by the DoD**.** The Army Reserve center is located in Garapan (Figure 11).

Rota. Rota is approximately 11 mi. (17.7 km) long and 3 mi. (4.8 km) wide (Figure 12). Training on Rota is scheduled with Joint Region Marianas and coordinated with Rota officials for proposed training areas and activities. Training activities conducted on Rota typically include special warfare training and combat search and rescue training.

There are no DoD ranges or training areas on Rota. Limited airfield, expeditionary, and special warfare training on Rota is arranged by Joint Region Marianas, and coordinated with CNMI and Rota authorities.

Coordinate System: World Platte Carre

Sources: USGS, GEBCO
Map Document: MITT00298v2

Figure 12. Rota

2.11.2 Ocean Operating Areas Outside the Bounds of the Mariana Islands Range Complex

In addition to the MIRC, the MITT action area includes the area to the north of the MIRC that is within the Exclusive Economic Zone of the Commonwealth of the Northern Mariana Islands and the areas to the west of the MIRC (Figure 1). The MITT action area also includes a transit corridor, which is a direct route between the MIRC and the Hawaii Range Complex.

Although not part of any defined range complex, the transit corridor is important to the Navy in that it provides adequate air, sea, and undersea space in which vessels and aircraft conduct training and some sonar maintenance and testing while in transit.

The transit corridor is defined by a great circle route (e.g., shortest distance) between the MIRC and the HRC. While in transit and along the corridor, vessels and aircraft would, at times, conduct basic and routine unit level training such as gunnery and sonar training as long as the training does not interfere with the primary objective of reaching their intended destination. Ships also conduct sonar maintenance, which includes active sonar transmissions.

Effects of Navy training and testing activities with the portion of the transit corridor that lies within the Hawaii-Southern California Training and Testing (HSTT) action area were analyzed separately in the HSTT biological opinion. This biological opinion addresses training and testing activities along the transit corridor outside of the action area defined for HSTT (in and around the Hawaiian Islands).

2.11.3 Pierside Locations and Apra Harbor

The action area includes pierside locations in the Apra Harbor Naval Complex where surface ship and submarine sonar maintenance testing occur. For purposes of this BO, pierside locations include channels and routes to and from the Navy port in the Apra Harbor Naval Complex, and associated wharves and facilities within the Navy port and shipyard (Figure 5).

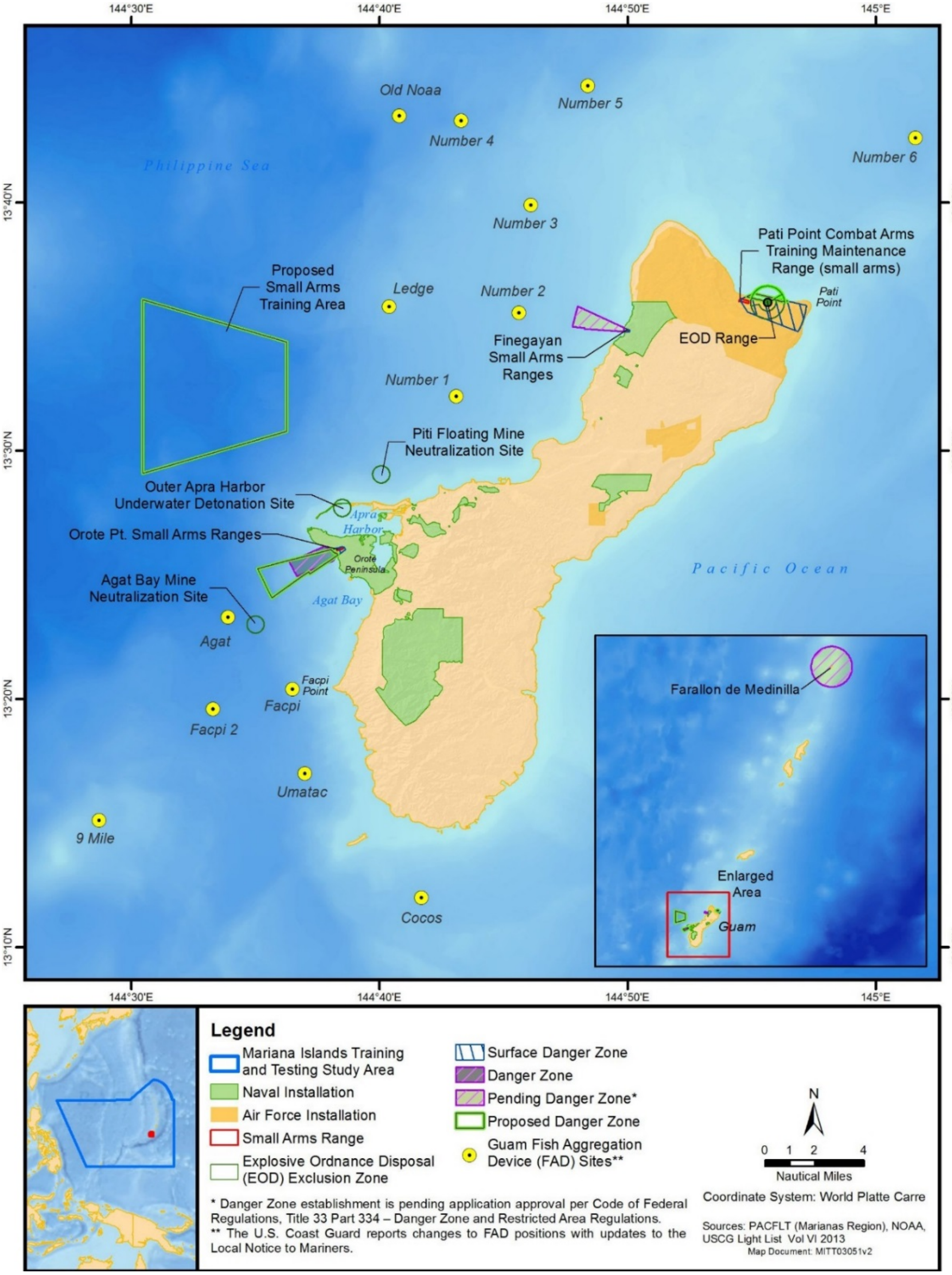
2.11.4 Nearshore Training and Testing Areas

The following table and figure describe the nearshore training and testing activities in MITT.

Table 20. Summary of Nearshore Training and Testing Activities

Nearshore Training and Testing Areas	Description
Pacific Ocean off Orote Point, Apra Harbor, Island of Guam, Mariana Islands; Small Arms Firing Range	Used for Small Arms Training. Down range Surface Danger Zone extends out over the nearshore waters of Guam off Orote Point.
Finegayan Small Arms Range	Used for small arms training. Down range Surface Danger Zone extends out over the nearshore waters of Guam off Haputo Point and overlays part of the “Small Arms Safety Drop Zone” shown on NOAA Chart 81048, Guam.
Pati Point Combat Arms Training Maintenance Small Arms Range	Used for small arms training. Down range Surface Danger Zone extends out over the nearshore waters of Guam off Pati Point.
Small Arms Firing Area	An area used by surface vessel crews to conduct small arms training. This firing area is over water west of Guam, beyond 3 nm of Guam and within territorial waters, and within a Navy “Firing Danger Area” charted on NOAA Chart 81048, Guam.
Agat Bay Mine Neutralization Site	Used by divers training to conduct underwater detonations (UNDETs). The Exclusion Zone has a minimum 640-meter (m) radius and is located beyond 3 nm of Guam and within territorial waters.
Piti Point Mine Neutralization Site	Used by divers training to conduct UNDETs. The Exclusion Zone has a minimum 640 m radius and is located within 3 nm of Guam.
Apra Harbor UNDET Site	Used by divers training to conduct UNDETs. The Exclusion Zone has a minimum 640 m radius over water, and is located within Apra Harbor. The Glass Breakwater forms the northern edge of Exclusion Zone.

Nearshore Training and Testing Areas	Description
Pati Point Explosive Ordnance Disposal Range	Land site used by the Air Force to dispose of ordnance. The Exclusion Zone extends partially out of Guam off Pati Point.



* Danger Zone establishment is pending application approval per Code of Federal Regulations, Title 33 Part 334 – Danger Zone and Restricted Area Regulations.
** The U.S. Coast Guard reports changes to FAD positions with updates to the Local Notice to Mariners.

Figure 13. Nearshore Training and Testing Areas

2.12 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 and interdependent actions outside the scope of this consultation.

3 OVERVIEW OF NMFS' ASSESSMENT FRAMEWORK

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

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

Section 7 assessment involves the following steps:

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

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

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

4) We identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. We also consider whether the action "may affect" designated critical habitat. This is our exposure analysis.

5) We evaluate the available evidence to determine how individuals of those ESA-listed species are likely to respond given their probable exposure. We also consider how the action may affect designated critical habitat. This is our response analyses.

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

7) The adverse modification analysis considers the impacts of the proposed action on the essential habitat features and conservation value of designated critical habitat.

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

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

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

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

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

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

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

3.1 Evidence Available for the Consultation

To conduct these analyses, we considered all lines of evidence available through published and unpublished sources that represent evidence of adverse consequences or the absence of such consequences. A considerable body of scientific information on anthropogenic sounds and their effect on marine mammals and other marine life has become available. NMFS status reviews for listed species also provide information on the status of the species including their resiliency, population trends and specific threats to recovery that contributes to our *Status of Listed Resources*, *Environmental Baseline*, and Risk Analyses.

To comply with our obligation to use the best scientific and commercial data available, we conducted electronic literature searches throughout the consultation, including within NMFS Office of Protected Resources' electronic library (using *EndNote* ® software). We examined the literature that was cited in the submittal documents and any articles we collected through our electronic searches. The U.S. Navy provided NMFS with a draft and final EIS/OEIS on training and testing that are conducted in the action area. We also evaluated the Navy's annual and comprehensive major training exercise and monitoring reports to assess effectiveness of mitigation and actual take incidental to actual training and testing activity levels where feasible.

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

3.1.1 The 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.

3.1.1.1 *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. Modeled effects from NAEMO were used to support the U.S. Navy's analyses in the MITT EIS/OEIS, mitigation strategies, Biological Evaluations, and MMPA incidental take authorization applications. We have verified the 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).

3.1.1.1.1 Overview

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

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

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

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 four 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 five 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 action area.

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 MITT action area, sound may propagate beyond the boundary of the action area. Any exposures occurring outside the boundary of the action area are counted as if they occurred within the action area boundary or within the action area for this opinion.

3.1.1.1.2 Model Assumptions

There are limitations to the data used in the NAEMO, and the results must be interpreted within 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 (Kastelein et al. 2009; Mooney et al. 2008; Popov and Supin 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.

3.1.2 Navy Marine Species Density Database

There is no single source of density data for every geographic area, marine species, and season of the year because of the costs, resources, and effort that would be required to provide sufficient survey coverage to estimate density. Therefore, to characterize the marine species density for large affected areas such as the MITT action area, the Navy compiled data from several sources, resulting in the Pacific Navy Marine Species Density Database (NMSDD) Technical Report dated September 2013.

For estimating densities of marine mammals and sea turtles in the MITT action area, the Navy selected representative areas (Figure 14) in the MITT action area including pelagic and nearshore environments and the transit corridor to represent the range of different habitats that could occur along the corridor, including both island (Wake Island) and open ocean habitats.

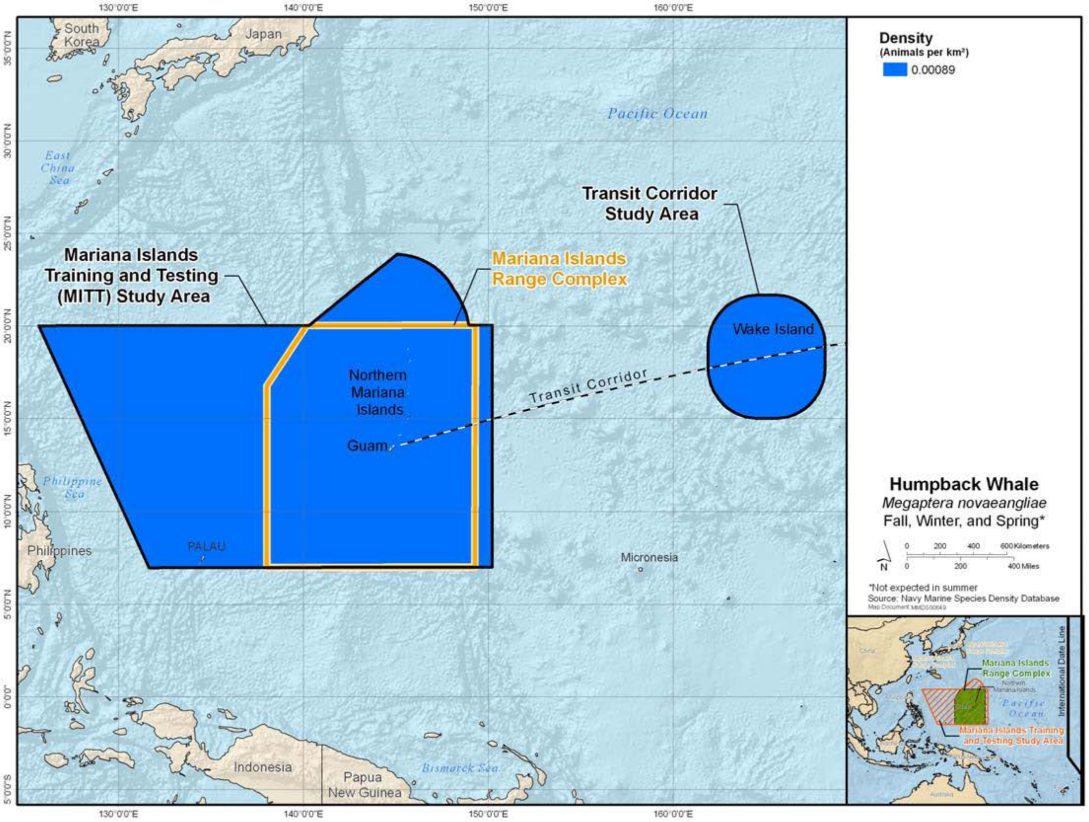


Figure 14. Example of Resprentative Areas Selected for Estimating Marine Species Densities

3.1.3 Differences Between Navy Modeling Considered in Previous Biological Opinions and this Biological Opinion

Phase I (roughly 2008 through 2012) of the Navy’s at-sea environmental planning and permitting effort including MIRC addressed U.S. Navy training and testing activities in a number of separate documents. Different modeling processes were used to estimate the effects of sound on marine species incidental to military readiness activities. Phase II (roughly 2013 through 2021) methodology eliminates the varying modeling processes by utilizing a standard model, NAEMO, for all acoustic effects analyses.

The first step earlier in the Phase I modeling process involved propagation modeling. For sonars, the Comprehensive Acoustic System Simulation (CASS)/Gaussian Ray Bundle (GRAB) model was used. Explosive sources were analyzed using either Reflection and Refraction in Multilayered Ocean/Ocean Bottoms with Shear Wave Effects (REFMS) or a modified version of CASS/GRAB. Phase II modeling retains some of the Phase I features, such as use of the same propagation model (i.e., CASS/GRAB), for developing tonal source footprints. Phase II uses

REFMS exclusively for explosive propagation and includes the addition of the Range-Dependent Acoustic Model (RAM) to model non-explosive impulsive sources (i.e., airguns).

For Phase I, footprints were created for each active source used in an activity, and the movements of the source were modeled over the operating area. Only one source type was modeled at a time. Unlike Phase I, NAEMO has the capability to simultaneously run multiple sources during a scenario, affording a more realistic depiction of the potential effects of an activity. For example, transmissions emitted by a surface combatant with its hull-mounted sonar, a helicopter with its dipping sonar, a torpedo's homing sonar, and the countermeasures discharged by the targeted submarine can be modeled simultaneously.

Although the acoustic propagation was modeled in three dimensions during Phase I analyses, in some cases, the three-dimensional (3-D) footprint was collapsed into a two-dimensional (2-D) acoustic footprint by utilizing the maximum received level, irrespective of the depth, at each range step. In other areas, a volumetric, 3-D footprint was developed to allow for variations in animal depth. For Phase II analyses, the 3-D acoustic propagation field was maintained throughout the analysis process.

Phase I distributed marine species uniformly in the respective density cells over the area being modeled. The animals were distributed in two dimensions, except in locations where data for species-specific dive profiles were available. In those areas, the animals were distributed in 3-D. In the 2-D distribution, all animals within the range of the maximum energy field would be affected, while in the volumetric approach, effects depended on where the animals were in the water column in relation to the propagation pattern. In Phase II, data on species-specific habitat preference, podding behavior, and dive profiles were taken into account and used to distribute individual animals in the model. An animat, or virtual representation of a marine animal, serves as a dosimeter, recording the energy received from all active sources during a scenario, resulting in the cumulative effects of all sources being accounted for when the impacts are analyzed.

Another difference between Phase I and Phase II modeling involves the environmental data used during propagation modeling. Phase II incorporates bathymetry into the propagation modeling process for non-impulsive sources and non-explosive impulsive (i.e., airgun) sources; Phase I used flat-bottomed bathymetry. Flat-bottom bathymetry will continue to be used in Phase II for all impulsive sources, as it was in Phase I. Futhermore, Phase II uses range-dependent sound speeds, wind speed, and bottom properties.

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

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

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

- Best available science on species' behavior,
- Number of platforms (i.e., aircraft, vessels) used during specific activities,
- Ability to detect specific species, and
- Ability to observe the mitigation zone around different platforms during different activities.

3.1.4.1 Animal avoidance behavior

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

3.1.4.2 Mitigation

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

The post-model acoustic effect analysis quantification process is summarized in Table 21. In brief, the mitigation effectiveness score for an event is multiplied by the estimated sightability of each species to quantify the number of animals that were originally modeled as a mortality (explosives only) or injury (all sound-producing activities) exposure but would, in reality, be observed by Lookouts or shore-based observers prior to or during a sound-producing activity. Observation of marine mammals prior to or during a sound-producing event would be followed by stop or delay of the sound-producing activity, which would reduce actual marine mammal sound exposures. The Navy only quantitatively adjusted model-predicted effects within the range to mortality (explosives only) and injury (all sound-producing activities). Despite employing the required mitigation measures during an activity that will also reduce some TTS exposures, the Navy did not quantitatively adjust the model-predicted TTS effects or other predicted behavioral effects as a result of implemented mitigation. The total model-predicted number of animals affected is not reduced by the post-model mitigation analysis, since all reductions in mortality and injury effects are then added to and counted as TTS effects.

The steps of the post-model analysis are briefly summarized in Table 21 and presented in the order they are expected to occur during an actual training or testing activity, which is also the order in which they were mathematically considered in the post-model analysis. When feasible for a given activity, mitigation begins prior to the actual production of underwater sound (e.g., 10–30 minutes, dependent upon platform, prior to most sonar and explosive activities); therefore, mitigation effectiveness is applied in the post-model analysis before animal avoidance is quantified.

Table 21. Post Model Acoustic Impact Analysis Process

Is the Sound Source Sonar/Other Active Acoustic Source or Explosives?	
Sonar and Other Active Acoustic Sources	Explosives
S-1. Is the activity preceded by multiple vessel activity or hovering helicopter?	E-1. Is the activity preceded by multiple vessel activity or hovering helicopter?
Species sensitive to human activity (i.e., beaked whales) are assumed to avoid the activity area, putting them out of the range to Level A harassment. Model-estimated PTS to these species during these activities are unlikely to actually occur and, therefore, are considered to be TTS (animal is assumed to move into the range of potential TTS).	Species sensitive to human activity (i.e., beaked whales) are assumed to avoid the activity area, putting them out of the range to mortality. Model-estimated mortalities to these species during these activities are unlikely to actually occur and, therefore, are considered to be injuries (animal is assumed to move into the range of potential injury).

The activities preceded by multiple vessel movements or hovering helicopters are listed in Tables 3.4-14 and 3.4-15 in Section 3.4.4.1.2 (Avoidance Behavior and Mitigation Measures as Applied to Sonar and Other Active Acoustic Sources) in the FEIS.	The activities preceded by multiple vessel movements or hovering helicopters are listed in Section 3.4.4.2.2 (Avoidance Behavior and Mitigation as Applied to Explosives) in the FEIS.
S-2. Can Lookouts observe the activity-specific mitigation zone (see Chapter 5) up to and during the sound-producing activity?	E-2. Can Lookouts observe the activity-specific mitigation zone (see Chapter 5) up to and during the sound-producing activity?
If Lookouts are able to observe the mitigation zone up to and during a sound-producing activity, the sound-producing activity would be halted or delayed if a marine mammal is observed and would not resume until the animal is thought to be out of the mitigation zone (per the mitigation measures in Chapter 5). Therefore, model-estimated PTS exposures are reduced by the portion of animals that are likely to be seen [Mitigation Effectiveness (1, 0.5, or 0) x Sightability, g(0)]. Any animals removed from the model-estimated PTS are instead assumed to be TTS (animal is assumed to move into the range of TTS). The g(0) value is associated with the platform (vessel or aircraft) with the dedicated Lookout(s). For activities with lookouts on both platforms, the higher g(0) is used for analysis. The Mitigation Effectiveness values are provided in Section 3.4.4.1.2 (Avoidance Behavior and Mitigation Measures as Applied to Sonar and Other Active Acoustic Sources) in the FEIS.	If Lookouts are able to observe the mitigation zone up to and during an explosion, the explosive activity would be halted or delayed if a marine mammal is observed and would not resume until the animal is thought to be out of the mitigation zone (per the mitigation measures in Chapter 5). Therefore, model-estimated mortalities and injuries are reduced by the portion of animals that are likely to be seen [Mitigation Effectiveness (1, 0.5, or 0) x Sightability, g(0)]. Any animals removed from the model-estimated mortalities or injuries are instead assumed to be injuries or behavioral disturbances, respectively (animals are assumed to move into the range of a lower effect). The g(0) value is associated with the platform (vessel or aircraft) with the dedicated Lookout(s). For activities with lookouts on both platforms, the higher g(0) is used for analysis. The g(0) values are provided in Table 3.4-8. The Mitigation Effectiveness values are provided in Table 3.4-21 in Section 3.4.2.2 (Avoidance Behavior and Mitigation as Applied to Explosives) in the FEIS.

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

The Navy incorporated the data from two Finneran studies (2010 and 2011), in coordination with other scientific literature, to develop auditory weighting functions and "weighted" thresholds for auditory criteria. A summary of the findings from the two papers is provided below as well as an explanation of how the Navy incorporated the results of these papers into the weighting functions and thresholds used to support the MITT EIS/OEIS and this opinion. We address the likely reasons that takes for ESA species declined when comparing the take estimates for MIRC to the MITT, despite the fact that one might have anticipated the application of the two recent Finneran papers (Finneran 2010) (Finneran 2011) would have resulted in increased estimated takes overall. The Navy and NMFS determined it was appropriate to use a new Navy model—NAEMO—that is considered the best available information and incorporates the new weighting functions and thresholds. NAEMO also incorporates a number of other significant changes and enhancements compared with the SAIC model used in MIRC. This paper surmises that the changes in density data and distributions incorporated in NAEMO was likely the main factor that influenced an overall reduction of take estimates for ESA species between MIRC and the MITT estimates used in this opinion.

Finneran and Schlundt (2010)

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

Finneran and Schlundt (2011)

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

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

The criteria used for predicted acoustic impacts to marine mammals are described in section 6.2.14 of this opinion. On August 4, 2016, NMFS released its Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (new Guidance). This new Guidance established new thresholds and associated weighting functions for predicting auditory injury, or permanent threshold shift (PTS) and temporary threshold shift (TTS). NMFS uses acoustic thresholds to help quantify "take" and as part of more comprehensive effects analyses under several statutes, including the ESA. In the August 4, 2016, Federal Register notice announcing the new Guidance (81 FR 51694), NMFS explained the approach it would take during a transition period, during which we will balance the need to consider this new best available science with the fact that some applicants have already committed time and resources to the development of analyses based on our previous thresholds and have constraints that preclude the recalculation of take estimates, as well as consideration of where the action is in the agency's decision-making "pipeline." In that notice, we included a non-exhaustive list of factors that would inform the most appropriate approach for considering the new Guidance, including: how far in the process the application or prospective application has progressed; when the activity is scheduled to begin or other timing constraints; the complexity of the analyses and the cost and practicality of redoing them; the temporal and spatial scope of anticipated effects; and the relative degree to which the new Guidance is expected to affect the results of the acoustic impact analyses.

In developing the new Guidance, NMFS compiled, interpreted, and synthesized scientific information currently available on the effects of anthropogenic sound on marine mammals, including a recent Technical Report by Dr. James Finneran (U.S. Navy-SPAWAR Systems Center Pacific) that proposed new weighting functions and thresholds for predicting the onset of both PTS and temporary threshold shifts (TTS) in marine mammals (Finneran, 2016). The methodologies presented within this paper (and in NMFS' new Guidance) build upon the methodologies used to develop the criteria applied within the 2015 MITT biological opinion and MMPA authorization (Finneran and Jenkins 2012), and incorporate relevant auditory research made available since 2012 (e.g., Kastelein *et al*., 2012a; Kastelein *et al*., 2012b; Finneran and Schlundt, 2013; Kastelein *et al*., 2013a; Kastelein *et al*., 2013b; Popov *et al*., 2013; Kastelein *et al*., 2014a; Kastelein *et al*., 2014b; Popov *et al*., 2014; Finneran *et al*., 2015; Kastelein *et al*., 2015a; Kastelein *et al*., 2015b; Popov *et al*., 2015). In light of limited data at the time, Finneran and Jenkins (2012) presented a conservative approach to development of auditory weighting functions. In 2016, with the benefit of newly-available data, Finneran was able to synthesize a wide range of auditory data, including newly-available studies, to predict refined auditory weighting functions and corresponding TTS and PTS thresholds across the complete hearing ranges of functional hearing groups. At the time of the issuance of the MITT final rule and biological opinion in 2015, the Technical Guidance had not been issued. Therefore, the Navy did not directly use the new auditory weighting functions and PTS/TTS criteria in its acoustic modeling for this biological opinion. However, the underlying science contained within Finneran (2015) (upon which NMFS' new Guidance is based) has been addressed qualitatively within the applicable sections of this biological opinion.

In addition to the fact that it was possible to address the new Guidance adequately without remodeling, it would have been impractical for the Navy to re-model its proposed action based on the new Guidance for this reinitiated consultation.[5] NMFS Permit Division promulgated regulations and issued a letter of authorization pursuant to the MMPA for the Navy to "take" marine mammals incidental to MITT activities on July 31, 2015. Prior to issuance of the rule and LOA, the Navy committed substantial time and resources to the development of acoustic analyses based on previous guidance. It took approximately ten months for the Navy to collect necessary information on MITT activities (e.g., number of sonar hours expected for each source bin, etc).The initial round of MITT modeling took approximately six months to complete, there was a follow on to that modeling that was delivered about five months later. Additionally, it is worth noting that modeling for Phase III MITT activities is scheduled to begin this year. In short, the underlying science contained within Finneran (2015) (upon which NMFS' new Guidance is based) has been addressed qualitatively within the applicable sections of this biological opinion, and the exposure estimates were not remodeled because a qualitative assessment of the new Guidance and the activities showed that it was not necessary in order to support the analysis, in addition to being impractical.

If the new Guidance were applied to the MITT acoustic effects analysis, the predicted numbers of PTS/TTS could change for some functional hearing groups. For the sonar exposure estimates, if the new Guidance was quantitatively applied to the MITT effects analysis and new modeling conducted, predicted numbers of PTS and/or TTS would change to some small degree (even if only by fractions of a take). However, because the new Guidance relies on much of the same data as the auditory criteria used in the Navy's modeling, these changes would not be substantial, and would result in a reduction in the predicted impacts to the species evaluated in this opinion. Onset PTS and TTS thresholds for non-impulsive sound (sonar) for cetaceans are lower (*i.e.*, are more conservative) in Finneran and Jenkins 2012 (used in the Navy's modeling for Phase II MITT

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[5] Further, we note that this reinitated consultation revises the 2015 biological opinion to address the following: 1) analysis of impacts to green sea turtles in consideration of the final rule, issued in 2016, to list 11 DPSs of green sea turtles as threatened or endangered under the ESA (81 FR 20057); 2) analysis of humpback whales in consideration of the final rule, issued in 2016, to divide the globally-listed humpback whale into 14 DPSs and list four DPSs as endangered and one as threatened (81 FR 62259); and 3) new scientific information provided by the Navy on coral coverage at Farallon de Medinilla. None of these revisions are related to acoustic modeling or the enumeration of marine mammal takes in the form of temporary or permanent threshold shift which was the subject of the new Guidance.

activities) compared to the new Guidance, while updated auditory weighting functions for most marine mammal hearing groups have changed minimally in the new Guidance. This means that the predicted ranges to PTS and TTS from non-impulsive sources for ceteaceans evaluated in this opinion would change only minimally if NMFS' new Guidance were quantitatively applied and new modeling conducted, and that estimated numbers of takes of ESA-listed cetaceans resulting in PTS and TTS from sonar are in fact larger in this opinion than would be expected if the Navy's activities were re-modeled using the new Guidance.

For explosives, the criteria in the new Guidance would be expected to result in a larger range to effects for many explosive bins, but we would not expect any differences in take estimates for the species evaluated in this opinion to be significant. For example, during the recently completed (April 2017) consultation on U.S. Navy training activities in the Gulf of Alaska (NMFS 2017), the Navy was able to reprocess anticipated ranges to effects for Level A harassment (PTS), and subsequently ranges to effects for TTS and behavioral exposures, based on the new Guidance to assess if the new impulsive criteria could result in any additional species-specific takes from explosives. The conclusion from that analysis was that the new impulsive criteria would not change previous species-specific quantities of impulsive PTS, TTS, or behavioral exposures for any ESA-listed cetaceans, and the mitigation zones implemented during training activities using explosives remained sufficiently protective (i.e., revised range to effects for PTS, even when larger, remain encompassed by the mitigation zones for all explosive types and hearing groups). We would expect similar results if the Technical Guidance was applied to the Phase II MITT acoustic effects analysis because the types of activities conducted (i.e., explosive sources) and the mitigation measures implemented by the Navy in MITT are similar to those in the Gulf of Alaska. This information suggests that the new criteria and information presented in Finneran 2016 and the recently released Guidance do not provide a substantially different picture of the environmental impacts as to alter our conclusions.

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

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

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

3.3 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).

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

For "species" (the entity that has been listed as endangered or threatened, not the biological species concept), we are concerned about whether the number of populations that are likely to experience "significant" reductions in viability (= increases in their extinction probabilities) and the nature of any reductions in viability are likely to have "significant" consequence for the viability (= probability of demographic, ecological, or genetic extinction) of the "species" those population 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") of the entire area contained in the designation.

3.4 "Take" and Threatened Species

ESA section 9(a) take prohibitions (16 U.S.C. 1538(a)(1)(B)) apply to all species listed as endangered. In the case of threatened species, section 4(d) of the ESA leaves it to the Secretary's discretion whether and to what extent to extend the statutory 9(a) ''take'' prohibitions, and directs the agency to issue regulations it considers necessary and advisable for the conservation of the species. At this time, NMFS has not issued section 4(d) rules for the threatened coral species or for the threatened scalloped hammerhead shark – Indo-Pacific DPS. Therefore, the take prohibitions of section 9(a) of the ESA have not been extended to these species. However, consistent with *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we did assess the amount or extent of take to threatened species that is anticipated incidental to Navy training and testing activiites and included this information in the ITS. At such time a 4(d) rule is issued for these species, NMFS and the Navy may need to reinitiate consultation to amend the incidental take statement.

4 STATUS OF LISTED RESOURCES

This section identifies the ESA-listed species that potentially occur within the action area that may be affected by Mariana Islands training and testing activities. 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 22, along with their regulatory status.

Table 22. ESA-listed Species That May be Affected by MITT Activities

Species	ESA Status	Critical Habitat	Recovery Plan	Recent 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	?
Humpback Whale (*Megaptera novaeangliae*) – Western North Pacific DPS	E – 81 FR 62259	-- --	55 FR 29646	?*
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	-- --	76 FR 43985	?
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	-- --	75 FR 81584	?
Sea Turtles				
Green Turtle (*Chelonia mydas*) – Central North Pacific DPS	T – 81 FR 20057	-- --	63 FR 28359	↑**
Green Turtle (*Chelonia mydas*) – East Indian-West Pacific DPS	T – 81 FR 20057	-- --	63 FR 28359	↕**
Green Turtle (*Chelonia mydas*) – Central West Pacific DPS	E – 81 FR 20057	-- --	63 FR 28359	?**
Hawksbill Turtle (*Eretmochelys imbricata*)	E – 35 FR 8491	-- --	63 FR 28359	↓
Leatherback Turtle (*Dermochelys coriacea*)	E – 61 FR 17	-- --	63 FR 28359	↓
Loggerhead Turtle (*Caretta caretta*) – North Pacific DPS	E – 76 FR 58868	-- --	63 FR 28359	↓
Olive Ridley Turtle (*Lepidochelys olivacea*)	T – 43 FR 32800	-- --	63 FR 28359	↕
Fishes				
Scalloped hammerhead shark (*Sphyrna lewini*) – Indo-West Pacific DPS	T – 79 FR 38213	-- --	-- --	?
Corals‡				

Species	ESA Status	Critical Habitat	Recovery Plan	Recent Trend†
Acropora globiceps	T – 79 FR 53851	-- --	-- --	?
Acropora retusa	T – 79 FR 53851	-- --	-- --	?
Seriatopora aculeata	T – 79 FR 53851	-- --	-- --	?

† Species status trends from the FY 2013 to 2014 Report to Congress (NMFS 2015b): ↑ increasing, ↓ decreasing, ↕ mixed, ? unknown, - stable.

*Trend determined by information presented in the Final Rule to identify 14 DPSs of humpback whales and revise the species-wide listing.

**Trend determined by information presented in the Final Rule to list 11 DPSs of green sea turtles as threatened and endangered in the ESA.

‡NMFS determined neither *Acropora tenella nor Pavona diffluens* occur in the action rea.

Table 23 summarizes the stressor categories and potential for effects as assessed by the Navy. Our rationale for concurrence with Navy determinations of may affect, not likely to adversely affect, are summaried in Section 4.1 below. Species that are not likely to be adversely affected by any of the stressors in Table 23 were not carried forward in our effects analysis. Species that were likely to be adversely affected by any of the stressor categories are carried forward in our effects analysis.

Table 23. Summary of Navy ESA Effect Determinations and NMFS Final Determinations

Stressor	Navy Determinations		NMFS Final Determinations
	Likely to adversely affect	Not likely to adversely affect	Concurrence with Navy Determination (Yes or No)
Acoustic Stressors			
Non-implusive Acoustic Stressors - Sonar and Other Active Acoustic Sources	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Leatherback Turtle	Olive Ridley Turtle, Corals	Yes
Impulsive Acoustic Stressors - Explosives	Green Turtle, Hawksbill Turtle, Scalloped Hammerhead Shark	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, **Corals***	**Yes for all determinations except corals. *We determined that impulsive acoustic stressors would be likely to adversely affect *Acropora globiceps*.**

Stressor	**Navy Determinations**		**NMFS Final Determinations**
	Likely to adversely affect	**Not likely to adversely affect**	**Concurrence with Navy Determination (Yes or No)**
Weapons Firing, Launch, and Impact Noise	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes
Aircraft Noise	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle. Scalloped Hammerhead Shark	Yes
Vessel Noise	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes
Energy Stressors			
Electromagnetic Devices	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes
Physical Disturbance and Strike			

Stressor	**Navy Determinations**		**NMFS Final Determinations**
	Likely to adversely affect	**Not likely to adversely affect**	**Concurrence with Navy Determination (Yes or No)**
Vessel Strike	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Corals, Scalloped Hammerhead Shark,	Yes
In-Water Devices	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Corals, Scalloped Hammerhead Shark	Yes
Military Expended Materials	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, **Corals***, Scalloped Hammerhead Shark	**Yes for all determinations except corals. *We determined that military expended materials would be likely to adversely affect corals.**
Entanglement			
Fiber Optic Cables and Guidance Wires	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes

Stressor	Navy Determinations		NMFS Final Determinations
	Likely to adversely affect	Not likely to adversely affect	Concurrence with Navy Determination (Yes or No)
Decelerators/ Parachutes	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes
Ingestion			
Munitions	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Scalloped Hammerhead Shark	Yes
Military Expended Materials Other Than Munitions	--	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Scalloped Hammerhead Shark	Yes
Secondary Stressors			
Explosives, Metals, Chemicals, Disease, Parasites, Sedimentation	**--**	Humpback Whale, Blue Whale, Fin Whale, Sei Whale, Sperm Whale, Green Turtle, Hawksbill Turtle, Loggerhead Turtle, Olive Ridley Turtle, Leatherback Turtle, Corals, Scalloped Hammerhead Shark	Yes

-- indicates no species with that determination.

4.1 Species Not Considered Further in this Opinion

NMFS uses two criteria to identify the ESA-listed species that are not likely to be adversely affected by the proposed action, as well as the effects of activities that are interrelated to or interdependent with the Federal agency's proposed action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species. If we conclude that an ESA-listed species are not likely to be exposed to the proposed activities, we must also conclude that the species is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species that are exposed to a potential stressor but are likely to be unaffected by the exposure are also not likely to be adversely affected by the proposed action. We applied these criteria to the species ESA-listed in Table 22 and we summarize our results below.

An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat. Beneficial effects are usually discussed when the project has a clear link to the ESA-listed species or its specific habitat needs and consultation is required because the species may be affected.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect. That means the ESA-listed species may be expected to be affected, but not harmed or harassed.

Discountable effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did impact a listed species),but it is very unlikely to occur.

4.1.1 Olive Ridley Sea Turtle

Only one record of an olive ridley sea turtle has been documented for Guam or the CNMI, allegedly captured near Saipan (Pritchard 1977). Olive ridley sea turtles are rare in the MITT action area. For the purposes of modeling effects to species within the action area, the Navy has estimated the olive ridley sea turtle abundance at 0.000001 animals per km^2. The U.S. Navy determined that stressors resulting from sonar and other active acoustic sources, explosives, weapons firing/launch/impact noise, aircraft noise, vessel noise, electromagnetic devices, vessels and in-water devices, fiber optic cables/wires/parachutes, and military expended materials may affect, but are not likely to adversely affect olive ridley sea turtles due to very low potential for co-ocurrence of individuals and specific stressors that could result in "take."

Exposure of olive ridley sea turtles to acoustic stressors, including non-impulsive (e.g. sonar) and impulsive sources (e.g. explosives), is anticipated to be rare. Navy modeling predicts 12 exposures at greater than 120 dB from non-impulsive acoustic stressors related to training activities and 15 exposures from non-impulsive acoustic stressors related to testing activities.

None of these exposures are anticipated to rise to the level of take (≥ 175 dB re 1 μPa root mean square). Therefore, the potential effect of non-impulsive acoustic stressors on olive ridley sea turtles is insignificant. Navy modeling also predicts no exposures of olive ridley sea turtles to impulsive acoustic stressors greater than 120 dB. Therefore, the potential for impulsive acoustic stressors to impact olive ridley sea turtles is discountable.

Similarly, all other stressors associated with Navy training and testing activities in the MITT action area are not anticipated to adversely affect olive ridley sea turtles. As documented further in section 6.7, the potential effects of noise from munition firing, launch, and impact, noise from vessels and aircraft, are insignificant to ESA-listed sea turtles, including olve ridleys. Olive ridley sea turtles would be need to be within 198 m of electromagnetic devices to detect them and potentially respond behaviorally. Given the rarity of this species in the action area and the infrequency in which these devices are used, it is not reasonable to expect olive ridley sea turtles to be exposed to electromagnetic devices (i.e., exposed within 198 m). Additionally, due to the rarity of this species in the action area and the Navy's mitigation measures to avoid collisions with protected species while vessels are underway, we believe the likelihood of a Navy vessel striking an olive ridley sea turtle is discountable. Similarly, due the rarity of this species, the low speed of non-vessel in-water devices, and the ability of olive ridley sea turtles to move, the likelihood for a non-vessel in-water devices to collide with an olive ridley sea turtle is so low as to be considered discountable. Also, as documented further in section 6.7, the likelihood of fiber optic cables, guidance wires, and decelerators/parachutes affecting ESA-listed sea turtles, including olive ridleys, is so low as to be discountable. Finally, the ingestion of small and medium caliber projectiles and other military expended materials is not likely to occur because olive ridley sea turtles are rare in the action area and most expendable materials are used over deeper waters where this species is not expected to forage. Therefore, the potential for olive ridley sea turtles to ingest expended materials is discountable.

For the reasons stated above, the effect of Navy stressors on olive ridley sea turtles are either discountable or insignificant. We conclude that Navy training and testing activities in the MITT action area are not likely to adversely affect olive ridley sea turtles and this species is not considered further in this opinion.

4.1.2 Acropora retusa

Colonies of *Acropora retusa* have short, thick finger-like branches. Branches look rough and spiky because radial corallites are variable in length. Colonies are typically brown or green in color. *Acropora retusa* is easily confused with other digitate *Acropora* species. Though it can be difficult to distinguish from many other *Acropora* species (Doug Fenner, personnel communication to Stephen H. Smith; July 20, 2016), *Acropora retusa* has some identifying features. It can be distinguished from *Acropora globiceps*, *Acropora gemmifera*, *Acropora monticulosa*, *Acropora macrostomata*, and most other digitate *Acropora* by the rough, spiky branch surfaces due to variable length radial corallites. It can be distinguished from *Acropora* *humilis* and *Acropora cophodactyla* by the small, tubular, axial corallite. It can be distinguished from *Acropora branchi* by thicker branches when *Acropora branchi* has branches at all (79 FR 53852).

4.1.2.1 Distribution and Abundance

Acropora retusa is distributed from the Red Sea and the Indian Ocean to the central Pacific. Veron (2014) reported that this species is confirmed in 23 of his 133 Indo-Pacific ecoregions, and strongly predicted to be found in an additional 21. Wallace (1999b) reports its occurrence in five of her 29 Indo-Pacific areas, many of which are larger than Veron's ecoregions. Richards (2009) estimated its range at 68 million km^2. Within U.S. waters, this species is confirmed in American Samoa, Guam, and the PRIA. (http://www.fpir.noaa.gov/PRD/prd_coral.html).

Veron (2014) reports that *Acropora retusa* occupied 0.5 percent of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.21 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as ''rare.'' Overall abundance was described as ''common in South Africa, rare elsewhere.'' Veron did not infer trends in abundance from these data. According to the Final Rule (79 FR 53852), *Acropora retusa's* absolute abundance is at least millions of colonies at the time of listing.

Within the Mariana Islands, *Acropora retusa* is confirmed in Guam, but not in CNMI (NMFS 2015 "Listed Corals in the Indo-Pacific: *Acropora retusa*, 4 p."). On Guam, a recent review of available coral survey data from numerous sites around the island showed *Acropora retusa* at only one location. Several surveys were conducted within Apra Harbor, but the species was not found there (NMFS/PIRO/HCD Guam coral database, 2015).

4.1.2.2 Habitat

Acropora retusa inhabits shallow water upper reef slopes and tidal pools. It has been reported from depths of 1 to 5 m (79 FR 53852) and down to 11 m (D. Fenner, personal communication, 2015).

4.1.2.3 Status and trends

Acropora retusa was listed as threatened on September 10, 2014 (79 FR 53851). The abundance of *A. retusa* has likely declined over the past 50 to 100 years although a precise quantification is not possible based on the limited species specific information. The species is highly susceptible to ocean warming, and is susceptible to coral disease, ocean acidification, trophic effects of fishing, nutrients over-enrichment, and predation, all of which have increased in the past 50 to 100 years, and continue to increase throughout much of its range. *Acropora retusa*'s geographic distribution extends from the Red Sea and Indian Ocean to the central Pacific. Its depth range is at least zero to 5 meters in upper reef slopes, reef flats, and adjacent habitats. Its absolute abundance is at least millions of colonies across its range. While spatial variability of threats such as ocean warming and acidification across the species' range moderates vulnerability because many colonies are either not exposed to threats or do not negatively respond to a threat at any given point in time, the threats are increasing and will continue to increase in the foreseeable future, thus the species is expected to continue to decline. However, there is inadequate data to quantify current and future population status and trends on any spatial scale, let alone across the range of the species (79 FR 53852).

4.1.2.4 Conclusion

Acropora retusa is only known to occur at a single location within the MITT action area. This location is in Fouha Bay in southwestern Guam, away from MITT activities. As such, it is unlikely MITT activities will co-occur with any life stage (i.e., adult colonies, planktonic eggs, sperm, embryos, larvae) of *Acropora retusa*. Therefore, the likelihood of MITT activities affecting *Acropora retusa* is so low as to be discountable. We conclude that MITT activities are not likely to adversely affect *Acropora retusa* and this species is not considered further in this opinion.

4.1.3 *Seriatopora aculeata*

Colonies of *Seriatopora aculeata* have pencil-thick, short, branches which taper sharply at the end, usually in fused clumps. Colonies are pink or cream in color. *S. aculeata* can easily be confused with *Seriatopora stellata*, which also has branches which taper sharply at the end, however, in *S. stellata* corallites on branch sides are in rows and raised, while on *S. aculeata* they are not (79 FR 53852).

4.1.3.1 Distribution and Abundance

Seriatopora aculeata is distributed from Australia, Fiji, Indonesia, Japan, Papua New Guinea, and Madagascar to the Marshall Islands. Veron reports that *S. aculeata* is confirmed in 19 of his 133 Indo-Pacific ecoregions, and strongly predicted to be found in an additional seven. Within U.S. waters, this species is confirmed in Guam and CNMI (http://www.fpir.noaa.gov/PRD/prd_coral.html). According to the Final Rule (79 FR 53852) *S. aculeata*'s absolute abundance is at least millions of colonies at the time of listing.

Veron (2014) reports that *S. aculeata* occupied 10.3 percent of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.70 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as ''common,'' and overall abundance was described as ''uncommon.'' Veron did not infer trends in abundance from these data.

Within the Mariana Islands, *S. aculeata* is confirmed in both Guam and CNMI (NMFS 2015 "Listed Corals in the Indo-Pacific: *Seriatopora aculeata*, 4 p."). On Guam, a recent review of available coral survey data from numerous sites around the island showed *S. aculeata* at two locations around the island. Several surveys were conducted within Apra Harbor, but the species was not found there (NMFS/PIRO/HCD Guam coral database, 2015). In CNMI, coral survey data shows *S. aculeata* on reef slopes at numerous sites around Saipan (Douglas Fenner, personal communication, 2015).

4.1.3.2 Habitat

S. aculeata generally inhabits the reef slope and back-reef, including but not limited to upper reef slopes, mid-slope terraces, lower reef slopes, reef flats, and lagoons. Around Guam it is known to occur along exposed seaward reef slopes with good water circulation and low sedimentation, although it is also found in areas with high sedimentation rates. It is reported from depths of 3 to 40 m (79 FR 53852).

4.1.3.3 Status and trends

S. aculeata was listed as threatened on September 10, 2014 (79 FR 53851). The abundance of *S. aculeata* has likely declined over the past 50 to 100 years although a precise quantification is not possible based on the limited species specific information (79 FR 53852). The species is highly susceptible to ocean warming, and is susceptible to disease, ocean acidification, trophic effects of fishing, sedimentation, nutrients, sea-level rise, predation, and collection and trade, all of which have increased in the past 50 to 100 years, and continue to increase throughout much of its range. *S. aculeata* is distributed from Madagascar to the Marshall Islands. Its depth range is 3 to 40 meters, and it occurs in a broad range of habitats on the reef slope and back-reef. Its absolute abundance is at least millions of colonies across its range. While spatial variability of threats such as ocean warming and acidification across the species' range moderates vulnerability because many colonies are either not exposed to threats or do not negatively respond to a threat at any given point in time, the threats are increasing and will continue to increase in the foreseeable future, thus the species is expected to continue to decline. However, there is inadequate data to quantify current and future population status and trends on any spatial scale, let alone across the range of the species (79 FR 53852).

4.1.3.4 Conclusion

Seriatopora aculeata has only been documented in a few locations within the MITT action area. These locations are not areas where Navy activities that could affect ESA-listed corals are concentrated (e.g., FDM or Apra Harbor). As such, it is unlikely MITT activities will co-occur with any life stage (i.e., adult colonies, planktonic eggs, sperm, embryos, larvae) of *Seriatopora aculeata*. Therefore, the likelihood of MITT activities affecting *Seriatopora aculeata* is so low as to be discountable. We conclude that MITT activities are not likely to adversely affect *Seriatopora aculeata* and this species is not considered further in this opinion.

4.2 Listed Species Considered Further in this Opinion

This opinion examines the status of each species that would be affected by the proposed action. The status is determined by the level of risk that the ESA-listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. The species status section helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02. We present information on the diving and social behavior of the different species because that behavior helps determine whether aerial and ship board surveys are likely to detect each species. We also summarize information on the vocalizations and hearing of the different species because that background information lays the foundation for our assessment of the how the different species are likely to respond to sounds produced by the Navy's training exercises and testing activities. Then we summarize information on the threats to the species and the species' status given those threats to provide points of reference for the jeopardy determinations we make later in this opinion. That is, we rely on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on this NMFS Web site: [http://www.nmfs.noaa.gov/pr/species/index.htm].

4.2.1 Blue Whale

The blue whale, *Balaenoptera musculus* (Linnæus 1758), is a cosmopolitan species of baleen whale. It is the largest animal ever known to have lived on Earth: adults in the Antarctic have reached a maximum body length of about 33 m (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 1998c).

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. Blue whales may reach 70 to 80 years of age (COSEWIC 2002a; Yochem and Leatherwood 1985b).

4.2.1.1 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. 1999; 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).

4.2.1.2 Population Structure

For this and all subsequent species, the term "population" refers to groups of individuals whose patterns of increase or decrease in abundance over time are determined by internal dynamics (births resulting from sexual interactions between individuals in the group and deaths of those individuals) rather than external dynamics (immigration or emigration). This definition is a reformulation of definitions articulated by Futuymda (1986) and Wells and Richmond (1995) and is more restrictive than those uses of 'population' that refer to groups of individuals that co-occur in space and time but do not have internal dynamics that determine whether the size of the group increases or decreases over time (see review by Wells and Richmond 1995). The definition we apply is important to section 7 consultations because such concepts as 'population decline,' 'population collapse,' 'population extinction,' and 'population recovery' apply to the restrictive definition of 'population' but do not explicitly apply to alternative definitions. As a result, we do not treat the different whale "stocks" recognized by the International Whaling Commission or other authorities as populations unless those distinctions were clearly based on demographic criteria. We do, however, acknowledge those "stock" distinctions in these narratives.

At least three subspecies of blue whales have been identified based on body size and geographic distribution (*B. musculus intermedia*, which occurs in the higher latitudes of the Southern Oceans, *B. m. musculus*, which occurs in the Northern Hemisphere, and *B. m. brevicauda* which occurs in the mid-latitude waters of the southern Indian Ocean and north of the Antarctic convergence), but this consultation will treat them as a single entity. Readers who are interested in these subspecies will find more information in Gilpatrick et al. (1997), Kato et al. (1995), Omura et al. (1970), and Ichihara (1966).

Until recently, blue whale population structure had not been tested using molecular or nuclear genetic analyses (Reeves et al. 1998). A recent study by Conway (2005) suggested that the global population could be divided into four major subdivisions, which roughly correspond to major ocean basins: eastern North and tropical Pacific Ocean, Southern Indian Ocean, Southern Ocean, and western North Atlantic Ocean. The eastern North/tropical Pacific Ocean subpopulation includes California, western Mexico, western Costa Rica, and Ecuador, and the western North Pacific Ocean subpopulation (including blue whales in the MITT action area) (Conway 2005). Genetic studies of blue whales occupying a foraging area south of Australia (most likely pygmy blue whales) have been found to belong to a single population (Attard et al. 2010). For this opinion, blue whales are treated as four distinct populations as outlined by Conway (2005).

Blue whales occur widely throughout the North Pacific. Acoustic monitoring has recorded blue whales off Oahu and the Midway Islands, although sightings or strandings in Hawaiian waters have not been reported (Barlow et al. 1997a; Northrop et al. 1971; Thompson and Friedl 1982a). Nishiwaki (1966) notes blue whale occurrence among the Aleutian Islands and in the Gulf of Alaska, but until recently, no one has sighted a blue whale in Alaska for some time, despite several surveys (Carretta et al. 2005; Forney and Brownell Jr. 1996b; Leatherwood et al. 1982c; Stewart et al. 1987), possibly supporting a return to historical migration patterns (Anonmyous. 2009).

Blue whales are thought to summer in high latitudes and move into the subtropics and tropics during the winter (Yochem and Leatherwood 1985b). Minimal data suggest whales in the western region of the North Pacific may summer southwest of Kamchatka, south of the Aleutians, and in the Gulf of Alaska, and winter in the lower latitudes of the western Pacific (Sea of Japan, the East China, Yellow, and Philippine seas) and less frequently in the central Pacific, including Hawaii (Carretta et al. 2005; Stafford 2003a; Stafford et al. 2001a; Watkins et al. 2000), although this population is severely depleted or has been extirpated (Gilpatrick and Perryman. 2009). However, acoustic recordings made off Oahu showed bimodal peaks of blue whales, suggesting migration into the area during summer and winter (McDonald and Fox 1999; Thompson and Friedl 1982a).

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

4.2.1.3 Reproduction

Gestation takes 10 to 12 months, followed by a 6 to 7 month nursing period. Sexual maturity occurs at 5 to 15 years of age and calves are born at 2 to 3 year intervals (COSEWIC 2002a; NMFS 1998b; Yochem and Leatherwood 1985b). Recent data from illegal Russian whaling for Antarctic and pygmy blue whales support sexual maturity at 23 m and 19 to 20 m, respectively (Branch and Mikhalev 2008). The mean intercalving interval in the Gulf of California is roughly two and a half years (Sears et al. 2014). Once mature, females return to the same areas where they were born to give birth themselves (Sears et al. 2014).

4.2.1.4 Movement

Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999; Reeves et al. 2004). Blue whales migrate toward the warmer waters of the subtropics in fall to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a). 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).

4.2.1.5 Feeding

Data indicate that some summer feeding takes place at low latitudes in upwelling-modified waters, and that some whales remain year-round at either low or high latitudes (Clarke and Charif 1998a; Hucke-Gaete et al. 2004; Reilly and Thayer 1990; Yochem and Leatherwood 1985b). One population feeds in California waters from June to November and migrates south in winter/spring (Calambokidis et al. 1990; Mate et al. 1999). Prey availability likely dictates blue whale distribution for most of the year (Burtenshaw et al. 2004a; Clapham et al. 1999; Sears 2002 as cited in NMFS 2006a). The large size of blue whales requires higher energy requirements than smaller whales and potentially prohibits fasting Mate et al. (1999). Krill are the primary prey of blue whales in the North Pacific (Kawamura 1980; Yochem and Leatherwood 1985b). Blue whales typically occur alone or in groups of up to five animals, although larger foraging aggregations of up to 50 have been reported including aggregations mixed with other rorquals such as fin whales (Corkeron et al. 1999; Shirihai 2002). While feeding, blue whales show slowed and less obvious avoidance behavior then when not feeding (Sears et al. 1983 as cited in NMFS 2005c). Barlow (2003) reported mean group sizes of 1.0 to 1.9 during surveys off California, Oregon, and Washington.

4.2.1.6 Diving and Social Behavior

Blue whales spend more than 94 percent of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5 to 20 times at 12 to 20 sec intervals before a deep dive of 3 to 30 min (Croll et al. 1999a; Leatherwood et al. 1976; Maser et al. 1981; Yochem and Leatherwood 1985a). 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 2005c). 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.

4.2.1.7 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. (2006a) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0 to 78.7 Hz. Reported source levels are 180 to 188 dB re 1μPa, but may reach 195 dB re 1μPa (Aburto et al. 1997b; Clark and Gagnon 2004; Ketten 1998; McDonald et al. 2001a). Samaran et al. (2010) estimated Antarctic blue whale calls in the Indian Ocean at 179 ± 5 dB re 1 μPa_{rms} -1 m in the 17 to 30 Hz range and pygmy blue whale calls at 175± 1 dB re 1 μPa_{rms} -1 m in the 17 to 50 Hz range.

As with other baleen whale vocalizations, blue whale vocalization function is unknown, although numerous hypotheses exist (maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources) (Edds-Walton 1997; Payne and Webb. 1971; Thompson et al. 1992). Intense bouts of long, patterned sounds are common from fall through spring in low latitudes, but these also occur less frequently while in summer high-latitude feeding areas. Short, rapid sequences of 30 to 90 Hz calls are associated with socialization and may be displays by males based upon call seasonality and structure. The low-frequency sounds produced by blue whales can, in theory, travel long distances, and it is possible that such long-distance communication occurs (Edds-Walton 1997; Payne and Webb. 1971). The long-range sounds may also be used for echolocation in orientation or navigation (Tyack 1999).

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

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

Blue whale songs consist of repetitively patterned sounds produced over time spans of minutes to hours, or even days (Cummings and Thompson 1971; McDonald et al. 2001b). The songs are divided into pulsed/tonal units, which are continuous segments of sound, and phrases, which are repeated combinations of 1 to 5 units (Mellinger and Clark 2003; Payne and McVay 1971). A song is composed of many repeated phrases. Songs can be detected for hundreds, and even thousands of kilometers (Stafford et al. 1998), and have only been attributed to males (McDonald et al. 2001b; Oleson et al. 2007a). Worldwide, songs are showing a downward shift in frequency (Mcdonald et al. 2009). For example, a comparison of recordings from November 2003 and November 1964 and 1965 reveals a long-term shift in the frequency of blue whale calling near San Nicolas Island. In 2003, the spectral energy peak was 16 Hz compared to ~22.5 Hz in 1964 and 1965, illustrating a more than 30 percent shift in call frequency over four decades (McDonald et al. 2006b). McDonald et al. (2009) observed a 31 percent downward frequency shift in blue whale calls off the coast of California, and also noted lower frequencies in 7 of the world’s 10 known blue whale songs originating in the Atlantic, Pacific, Southern, and Indian Oceans. Many possible explanations for the shifts exist, but none have emerged as the probable cause.

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

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. 2007c). In the SOCAL Range Complex region, D calls are produced in highest numbers during the late spring and early summer, and in diminished numbers during the fall, when A-B song dominates blue whale calling (Hildebrand et al. 2011; Hildebrand et al. 2012; Oleson et al. 2007d).

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

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (Ketten 1997a; 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. 2001b) (Croll et al. 2001c; Oleson et al. 2007d; Stafford and Moore 2005b). In terms of functional hearing capability, blue whales belong to the low-frequency group, which have a hearing range of 7 Hz to 22 kHz (Southall et al. 2007b).

Nevertheless, recent studies indicate that blue whales can hear and respond to sounds in the mid-frequency range. Nineteen controlled exposure experiments were conducted on blue whales during the Southern California-10 behavorial response study (Southall et al. 2011a) and 13 in the Southern California-11 behavorial response study (Southall et al. 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 et al. 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. 2013). However, reactions were temporary and were not consistent across individuals based on received sound levels alone. Results were likely the result of a complex interaction between sound exposure factors such as proximity to sound source and sound type (mid-frequency sonar simulation vs. pseudo-random noise), environmental conditions, and behavioral state. Surface feeding whales did not show a change in behavior during controlled exposure experiments, but deep feeding and non-feeding whales showed temporary reactions that often quickly abated after sound exposure. Distances of the sound source from the whales during controlled exposure experiments were sometimes less than a mile. Melcón et al. (2012) tested whether MFA sonar and other anthropogenic noises in the mid-frequency band affected the “D-calls” produced by blue whales in the Southern California Bight. The likelihood of an animal calling decreased with the increased received level of mid-frequency sonar, beginning at a sound pressure level of approximately 110 to 120 dB re 1 µPa. It is not known whether the lower rates of calling actually indicated a reduction in feeding behavior or social contact since the study used data from remotely deployed, passive acoustic monitoring buoys.

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, blue whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.2.14.

4.2.1.8 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.

Estimates of blue whale abundance in the North Pacific are uncertain. Prior to whaling, Gambell (1976) reported there may have been as many as 4,900 blue whales. Blue whales were hunted in the Pacific Ocean, where 5,761 were killed from 1889 to 1965 (Perry et al. 1999). This estimate does not account for under-reporting by Soviet whalers, who took approximately 800 more individuals than were reported (Ivashchenko et al. 2013). The IWC banned commercial whaling in the North Pacific in 1966, although Soviet whaling continued after the ban. Although blue whale abundance has likely increased since its protection in 1966, the possibility of unauthorized harvest by Soviet whaling vessel, incidental ship strikes, and gillnet mortalities make this uncertain. Punt (2010) estimated the rate of increase for blue whales in the eastern North Pacific to be 3.2 percent annually (1.4 SE) between 1991 and 2005, while Calambokidis et al. (2010) estimated a growth rate of 3 percent annually.

To our knowledge, there have been no recent sightings of blue whales in the action area. The closest documented sighting occurred in 1995 near Cocos (Brent Tibbatts, pers. comm., June 25, 2013, as cited in Uyeyama (2014)). The Pacific Islands Fisheries Science Center has deployed high-frequency acoustic recording packages (HARPS) to monitor marine mammals in the U.S. Exclusive Economic Zone (EEZ) around the CNMI. These monitors have acoustically detected blue whales (Oleson et al. 2013), though given the long distance blue whale calls can travel it is not known if the animals were actually within the action area. With the exception of sightings by observers on fishing vessels (Carretta et al. 2011), there have been no sightings of blue whales during systematic surveys off Hawaii (Barlow 2006; Mobley Jr. et al. 2000), and no blue whales were detected during a 2007 winter survey of the action area nor during Navy-funded monitoring for the MIRC in 2009 through 2015 (Fulling et al. 2011; HDR 2011; HDR 2012a; Hill et al. 2013; Hill et al. 2011; Ligon et al. 2011; Oleson and Hill 2010) . The Navy's NMSDD estimates 0.00001 blue whales per km^2 in the MITT action area (DoN 2014).

4.2.1.9 Natural Threats

As the world's largest animals, blue whales are only occasionally known to be killed by killer whales (Sears et al. 1990; Tarpy 1979). Blue whales engage in a flight response to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Blue whales are known to become infected with the nematode *Carricauda boopis*, which are believed to have caused mortality in fin whale due to renal failure (Lambertsen 1986).

4.2.1.10 Anthropogenic Threats

Blue whales have faced threats from several historical and current sources. Blue whale populations are severely depleted originally due to historical whaling activity.

Increasing oceanic noise may impair blue whale behavior. Although available data do not presently support traumatic injury from sonar, the general trend in increasing ambient low-frequency noise in the deep oceans of the world, primarily from ship engines, could impair the ability of blue whales to communicate or navigate through these vast expanses (Aburto et al. 1997c; Clark 2006). Blue whales off California altered call levels and rates in association with changes in local vessel traffic (Mckenna 2011).

To date, there has not been a ship strike as a result of Navy training and testing activities in the Study Area; however, ship strike is a concern for balaenopterids in the North Pacific. In the California/Mexico stock of blue whales, annual incidental mortality due to ship strikes averaged one whale every 5 years, but we cannot determine if this reflects the actual number of blue whales struck and killed by ships (i.e., individuals not observed when struck and those who do not strand; Barlow et al. (1997a)). Ship strikes have recently averaged roughly one every other year (eight ship strike incidents are known (Jensen and Silber 2004)), but in September 2007, ships struck five blue whales within a few-day period off southern California (Calambokidis personal communication 2008)(Berman-Kowalewski et al. 2010). Dive data support a surface-oriented behavior during nighttime that would make blue whales particularly vulnerable to ship strikes during this time. Ship strikes were implicated in the deaths of five blue whales, from 2004 to 2008 (Carretta et al. 2012). Four of these deaths occurred in 2007, the highest number recorded for any year. During 2004 to 2008, there were an additional eight injuries of unidentified large whales attributed to ship strikes. Several blue whales have been photographed in California with large gashes in their dorsal surface that appear to be from ship strikes (J. Calambokidis, personal communication). Blue whale mortality and injuries attributed to ship strikes in California waters averaged 1.0 per year for 2004 to 2008. Ship strike is an issue for blue whales of Sri Lanka engaged in foraging in shipping lanes, with several individuals stranding or being found with evidence of being struck (De Vos et al. 2013; Ilangakoon 2012).

There is a paucity of contaminant data regarding blue whales. Available information indicates that organochlorines, including dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCB), benzene hexachloride (HCH), hexachlorobenzene (HCB), chlordane, dieldrin, methoxychlor, and mirex have been isolated from blue whale blubber and liver samples (Gauthier et al. 1997b; Metcalfe et al. 2004). Contaminant transfer between mother and calf occurs, meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional contaminant loads during life and passing higher loads to the next generation (Gauthier et al. 1997a; Metcalfe et al. 2004). This is supported by ear plug data showing maternal transfer of pesticides and flame retardants in the first year of life (Trumble et al. 2013). These data also support pulses of mercury in body tissues of the male studied (Trumble et al. 2013).

4.2.1.11 Critical Habitat

The NMFS has not designated critical habitat for blue whales.

4.2.2 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 can be found in social groups of 2 to 7 whales. Aguilar and Lockyer (1987) suggested annual natural mortality rates in northeast Atlantic fin whales may range from 0.04 to 0.06. This is supported by an estimated annual survival rate of 0.955 for Gulf of St. Lawrence fin whales (Ramp et al. 2014). Fin whales live 70 to 80 years (Kjeld et al. 2006b).

4.2.2.1 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 1997; Campbell et al. 2015; Dohl et al. 1983; Forney et al. 1995a). 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. 1999). 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. 1968; Shallenberger 1981;

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).

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 1995b; 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). Fin whales have been recorded in the eastern tropical Pacific (Ferguson 2005) and are frequently sighted there during offshore ship surveys.

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. 2006a; 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.

4.2.2.2 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 (Archer et al. 2013; Gray 1865; Van Waerebeek and Engblom 2007). 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. (1984a) 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 1974b; Sigurjonsson et al. 1989), which suggests that these management units are not geographically isolated populations.

Mizroch et al. (1984a) 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 1974b); (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 1974a); and (5) a group centered in the Sea of Cortez (Gulf of California).

4.2.2.3 Reproduction

Fin whales reach sexual maturity between 5–15 years of age (COSEWIC 2005; Gambell 1985a; Lockyer 1972). Mating and calving occurs primarily from October-January, gestation lasts approximately 11 months, and nursing occurs for 6 to 11 months (Boyd et al. 1999; Hain et al. 1992). The average calving interval in the North Atlantic is estimated at about 2 to 3 years (Agler et al. 1993; Christensen et al. 1992). The location of winter breeding grounds is uncertain but mating is assumed to occur in pelagic mid-latitude waters (Perry et al. 1999). This was recently contradicted by acoustic surveys in the Davis Strait and off Greenland, where singing by fin whales peaked in November through December; the authors suggested that mating may occur prior to southbound migration (Simon et al. 2010). Although seasonal migration occurs between presumed foraging and breeding locations, fin whales have been acoustically detected throughout the North Atlantic Ocean and Mediterranean Sea year-round, implying that not all individuals follow a set migratory pattern (Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010). (Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)(Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)(Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)(Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)(Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)(Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010)Reductions in pregnancy rates appear correlated with reduced blubber thickness and prey availability (Williams et al. 2013). Recent IWC scientific whaling data suggest that, compared to commercial whaling periods, preganancy rates have decreased, age at sexual maturity has increased, size growth is slowing, and males now compose a slightly higher proportion of the population than female (Gunnlaugsson et al. 2013).

4.2.2.4 Feeding

Most fin whales in the Southern Hemisphere migrate seasonally from Antarctic feeding areas in the summer to low-latitude breeding and calving grounds in winter. Fin whales in the North Atlantic eat pelagic crustaceans (mainly krill and schooling fish such as capelin, herring, and sand lance (Borobia and Béland 1995; Christensen et al. 1992; Hjort and Ruud 1929;

Ingebrigtsen 1929; Jonsgård 1966; Mitchell 1974a; Overholtz and Nicolas 1979; Sergeant 1977; Shirihai 2002; Watkins et al. 1984). In the North Pacific, fin whales also prefer euphausiids and large copepods, followed by schooling fish such as herring, walleye pollock, and capelin (Kawamura 1982a; Kawamura 1982b; Ladrón De Guevara et al. 2008; Nemoto 1970; Paloma et al. 2008). Fin whales frequently forage along cold eastern current boundaries (Perry et al. 1999). Antarctic fin whales feed on krill, *Euphausia superba*, which occurs in dense near-surface schools (Nemoto 1959). However, off the coast of Chile, fin whales are known to feed on the euphausiid *E. mucronata* (Antezana 1970; Perez et al. 2006). Feeding may occur in waters as shallow as 10 m when prey are at the surface, but most foraging is observed in high-productivity, upwelling, or thermal front marine waters (Gaskin 1972; Nature Conservancy Council 1979 as cited in ONR 2001; Panigada et al. 2008; Sergeant 1977). While foraging, fin whales in the Mediterranean Sea have been found to move through restricted territories in a convoluted manner (Lafortuna et al. 1999). Fin whales in the central Tyrrhenian Sea appear to ephemerally exploit the area for foraging during summer, particularly areas of high primary productivity (Arcangeli et al. 2013).

4.2.2.5 Diving and Social Behavior

The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5 to 20 shallow dives, each of 13 to 20 s duration, followed by a deep dive of 1.5 to 15 min (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale’s most common dives last 2 to 6 min (Hain et al. 1992; Watkins 1981a). 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).

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). Most fin whales in the Southern Hemisphere migrate seasonally from Antarctic feeding areas in the summer to low-latitude breeding and calving grounds in winter. Fin whales tend to avoid tropical and pack-ice waters, with the high-latitude limit of their range set by ice and the lower-latitude limit by warm water of approximately 15° C (Sergeant 1977). Fin whale concentrations generally form along frontal boundary, or mixing zones between coastal and oceanic waters, which corresponds roughly to the 200 m isobath (the continental shelf edge) (Cotte et al. 2009; Nasu 1974a).

4.2.2.6 Vocalization and Hearing

Fin whales produce a variety of low-frequency sounds in the 10 Hz to 200 Hz range (Edds 1988; Thompson et al. 1992; Watkins 1981b; Watkins et al. 1987). Typical vocalizations are long, patterned pulses of short duration (0.5 to 2 s) in the 18 Hz to 35 Hz range, but only males are known to produce these (Clark et al. 2002; Patterson and Hamilton 1964). Richardson et al. (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 (Au and Green 2000) reported moans of 14 Hz to 118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34 Hz 150 Hz, and songs of 17 Hz to 25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981b). Source levels for fin whale vocalizations are 140 to 200 dB re 1μPa-m (see also Clark and Gagnon 2004; as compiled by Erbe 2002b). The source depth of calling fin whales has been reported to be about 50 m (Watkins et al. 1987).

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

The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997b) hypothesized that large mysticetes have acute infrasonic hearing. In a study using computer tomography scans of a calf fin whale skull, Cranford and Krysl (2015) found 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 1997a; 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 1981b; Watkins et al. 1987). These pulses frequently occur in long sequenced patterns, are down swept (e.g., 23 to 18 Hz), and can be repeated over the course of many hours (Watkins et al. 1987). In temperate waters, intense bouts of these patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high latitude feeding areas (Clarke and Charif 1998b). The seasonality and stereotypic nature of these vocal sequences suggest that they are male reproductive displays (Watkins 1981b; Watkins et al. 1987); a notion further supported by recent data linking these vocalizations to male fin whales only (Croll et al. 2002). In Southern California, the 20 Hz pulses are the dominant fin whale call type associated both with call-counter-call between multiple animals and with singing (Navy 2010; Navy 2012). An additional fin whale sound, the 40 Hz call described by Watkins (1981b), was also frequently recorded, although these calls are not as common as the 20 Hz fin whale pulses. Seasonality of the 40 Hz calls differed from the 20 Hz calls, since 40 Hz calls were more prominent in the spring, as observed at other sites across the northeast Pacific (Sirovic et al. 2012). Source levels of Eastern Pacific fin whale 20-Hz calls has been reported as 189 +/- 5.8 dB re 1uPa at 1m (Weirathmueller et al. 2013). Although acoustic recordings of fin whales from many diverse regions show close adherence to the typical 20 Hz bandwidth and sequencing when performing these vocalizations, there have been slight differences in the pulse patterns, indicative of some geographic variation (Thompson et al. 1992; Watkins et al. 1987).

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

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

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, fin whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.2.14.

4.2.2.7 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 2011a).

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.

Fin whales are typically not expected south of 20°N during summer and are less likely to occur near Guam (Miyashita et al. 1996). Miyashita et al. (1996) presented a compilation of at-sea sighting results by species, from commercial fisheries vessels in the Pacific Ocean from 1964 to 1990. For fin whales in August, Miyashita et al. (1996) reported no sightings south of 20°N, and significantly more sightings north of 40°N. However, they also showed limited search effort south of 20°N. There were no fin whale sightings during the winter 2007 survey of the action area (Fulling et al. 2011)nor during Navy-funded monitoring for the MIRC in 2009 through 2013 (HDR 2011; HDR 2012a; Hill et al. 2013; Hill et al. 2011; Ligon et al. 2011; Oleson and Hill 2010). The Pacific Islands Fisheries Science Center has deployed several passive acoustic monitoring devices to monitor marine mammals and ambient noise levels in U.S. EEZ waters off the Mariana Islands. Recordings from these instruments are currently being analyzed, but it has been confirmed that fin whales have been acoustically detected (Oleson et al. 2013). The Navy’s NMSDD estimates 0.00001 fin whales per km^2 in the MITT action area (DoN 2014).

4.2.2.8 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 1992). Adult fin whales engage in a 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). Shark attacks may also result in serious injury or death in very young and sick individuals (Perry et al. 1999).

4.2.2.9 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 2003, two males and four females were landed and two others were struck and lost (IWC 2005). In 2004, five males and six females were killed, and two other fin whales were struck and lost. 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). In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each year for the 2005 to 2006 and 2006 to 2007 seasons under an Antarctic Special Permit NMFS (2006c). Japanese whalers plan 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. 2007b). 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). According to Waring et al. (2007b), 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. 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. 2005a; Nelson et al. 2007a). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. 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 fin whale ship strike mortality by 27 percent in the Bay of Fundy region. 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/Hawaii. 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). From 1994 to 1998, two fin whales were presumed killed by ship strikes. More recently, in 2002, three fin whales were struck and killed by vessels in the eastern North Pacific (Jensen and Silber 2003c). The vast majority of ship strike mortalities are never identified, and actual mortality is higher than currently documented; however, it is Navy policy to report all ship strikes.

Increased noise in the ocean stemming from shipping seems to alter the acoustic patterns of singing fin whales, possibly hampering reproductive parameters across wide regions (Castellote et al. 2012c).

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. 1997a; Gauthier et al. 1997b). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males (Aguilar and Borrell 1988).

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

4.2.2.10 Critical Habitat

The NMFS has not designated critical habitat for fin whales.

4.2.3 **Humpback Whale – Western North Pacific DPS**

The humpback whale is a widely distributed baleen whale found in all major oceans. Humpbacks are distinguishable from other whales by long pectoral fins and are typically dark grey with some areas of white. The humpback whale was originally listed as endangered on December 2, 1970 (35 FR 18319). Since then, NMFS has designated 14 distinct population segments (DPS's) with four identified as endangered (Cape Verde Islands/Northwest Africa, Western North Pacific, Central American, and Arabian Sea) and one as threatened (Mexico) (Figure 15) (81 FR 62259). Humpback whales from the Western North Pacific DPS occur in the action area.

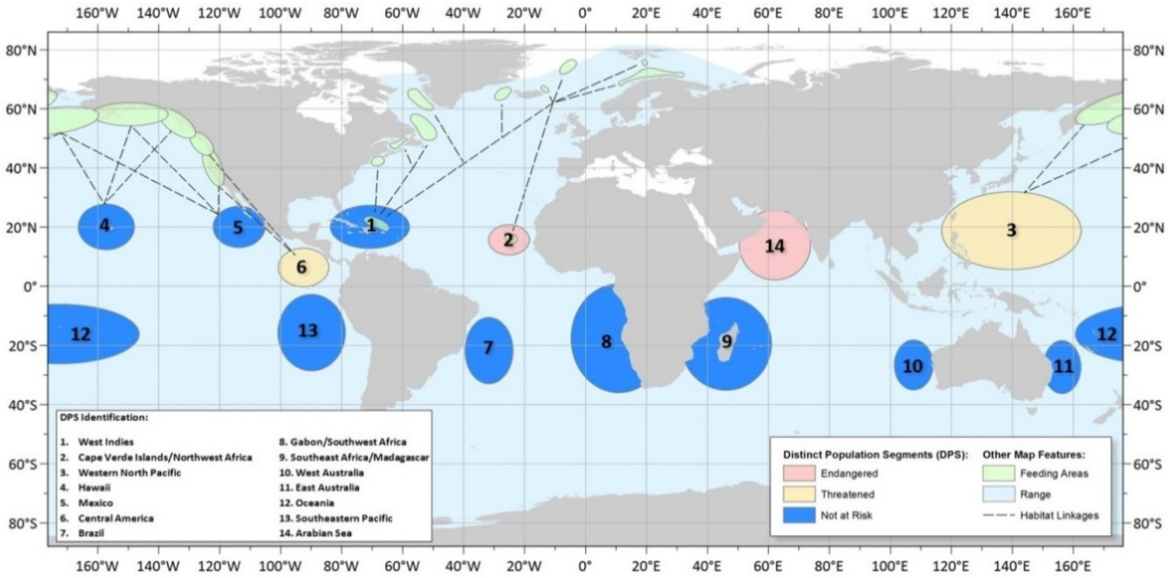

Figure 15. Map identifying 14 distinct population segments with 1 threatened and 4 endangered, based on primary breeding location of the humpback whale, their range, and feeding areas (Bettridge et al. 2015).

4.2.3.1 Distribution

Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In both regions, humpback whales tend to occupy coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985a).

The Western North Pacific DPS consists of humpback whales breeding/wintering in the area of Okinawa and the Philippines, another unidentified breeding area (inferred from sightings of whales in the Aleutian Islands area feeding grounds), and those transiting from the Ogasawara area. These whales migrate to feeding grounds in the northern Pacific, primarily off the Russian coast (Figure 15) (81 FR 62259)

4.2.3.2 Reproduction and Growth

Humpback whale calving and breeding generally occurs during winter at lower latitudes. Gestation takes about 11 months, followed by a nursing period of up to 1 year (Baraff and Weinrich 1993). Sexual maturity is reached at between 5 to 7 years of age in the western North Atlantic, but may take as long as 11 years in the North Pacific, and perhaps over 11 years (e.g., southeast Alaska, Gabriele et al. 2007). Females usually breed every 2 to 3 years, although consecutive calving is not unheard of (Clapham and Mayo 1987; 1990; Glockner-Ferrari and Ferrari 1985 as cited in NMFS 2005b; Weinrich et al. 1993). Males appear to return to breeding grounds more frequently than do females (Herman et al. 2011). Larger females tend to produce larger calves that may have a greater chance of survival (Pack et al. 2009). Females appear to preferentially select larger-sized males (Pack et al. 2012). In some Atlantic areas, females tend to prefer shallow nearshore waters for calving and rearing, even when these areas are extensively trafficked by humans (Picanco et al. 2009). Offspring appear to return to the same breeding areas at which they were born one they are independent (Baker et al. 2013).

In calving areas, males sing long complex songs directed towards females, other males, or both. The breeding season can best be described as a floating lek or male dominance polygamy (Clapham 1996). Calving occurs in the shallow coastal waters of continental shelves and oceanic islands worldwide (Perry et al. 1999). Males court females in escort groups and compete for proximity and presumably access to reproduce females (particularly larger females)(Pack et al. 2009). Although long-term relationships do not appear to exist between males and females, mature females do pair with other females; those individuals with the longest standing relationships also have the highest reproductive output, possibly as a result of improved feeding cooperation (Ramp et al. 2010). Site fidelity off Brazilian breeding grounds was extremely low, both within and between years (Baracho-Neto et al. 2012).

Generation time for humpback whales is estimated at 21.5 years, with individuals surviving from 80-100 years (COSEWIC 2011).

4.2.3.3 Feeding

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; Witteveen et al. 2011). 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). Additional evidence, such as songs sung in northern latitudes during winter, provide additional support to plastic seasonal distribution (Smith and G.Pike 2009). Relatively high rates of resighting in foraging sites in suggest whales return to the same areas year after year (Ashe et al. 2013; Kragh Boye et al. 2010). This trend appears to be maternally linked, with offspring returning to the same areas their mother brought them once calves are independent (Baker et al. 2013; Barendse et al. 2013). Humpback whales in foraging areas may forage largely or exclusively at night when prey are closer to the surface while in foraging areas (Friedlaender et al. 2013). Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992; Tynan et al. 2005b).

4.2.3.4 Diving and Social Behavior

In Hawaiian waters, humpback whales remain almost exclusively within the 1,800 m isobath and usually within water depths of less than 182 m. Maximum diving depths are approximately 170 m (but usually <60 m), with a very deep dive (240 m) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1 to 5.1 min in the North Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California, humpback whale dive durations averaged 3.5 min (Strong 1990). 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).

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; (Gendron and Urban 1993). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985b). Some individuals may not migrate, or species occurrence in foraging areas may extend beyond summer months (Van Opzeeland et al. 2013). Average group size near Kodiak Island is 2 to 4 individuals, although larger groups are seen near Shuyak and Sitkalidak islands and groups of 20 or more have been documented (Wynne et al. 2005).

4.2.3.5 Vocalization and Hearing

Humpback whale vocalization is much better understood than is hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop et al. 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144 to 174 dB (Au et al. 2006b; Au et al. 2000b; Frazer and Mercado III 2000; Richardson et al. 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. (2000a) noted that humpbacks off Hawaii tended to sing louder at night compared to the day. There is geographical variation in humpback whale song, with different populations singing a basic form of a song that is unique to their own group. However, the song evolves over the course of a breeding season, but remains nearly unchanged from the end of one season to the start of the next (Payne et al. 1983). The song is an elaborate series of patterned vocalizations that are hierarchical in nature, with a series of songs (‘song sessions’) sometimes lasting for hours (Payne and McVay 1971). Components of the song range from below 20 Hz up to 4 kHz, with source levels measured between 151 and 189 dB re 1 μPa-m and high-frequency harmonics extending beyond 24 kHz (Au et al. 2006b; Winn et al. 1970).

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

“Feeding” calls, unlike song and social sounds are a highly stereotyped series of narrow-band trumpeting calls. These calls are 20 Hz to 2 kHz, less than 1 second in duration, and have source levels of 162 to 192 dB re 1 μPa-m (D'Vincent et al. 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[6]) (Stimpert et al. 2007). Underwater lunge behavior was associated with nocturnal feeding at depth and with multiple bouts of broadband click trains that were acoustically different from toothed whale echolocation: Stimpert et al. (Stimpert et al. 2007) termed these sounds “mega-clicks” which showed relatively low received levels at the DTAGs (143 to 154 dB re 1 μPa), with the majority of acoustic energy below 2 kHz.

Humpback whale audiograms using a mathematical model based on the internal structure of the ear estimate sensitivity is from 700 Hz to 10 kHz, with maximum relative sensitivity between 2 kHz and 6 kHz (Ketten and Mountain 2014). Previously mentioned research by Au et al. (2001) and Au et al. (2006a) off Hawaii indicated the presence of high-frequency harmonics in

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

vocalizations up to and beyond 24 kHz. While recognizing this was the upper limit of the recording equipment, it does not demonstrate that humpbacks can actually hear those harmonics, which may simply be correlated harmonics of the frequency fundamental in the humpback whale song. The ability of humpbacks to hear frequencies around 3 kHz may have been demonstrated in a playback study. Maybaum (1990) reported that humpback whales showed a mild response to a handheld sonar marine mammal detection and location device with frequency of 3.3 kHz at 219 dB re 1µPa-m or frequency sweep of 3.1 kHz to 3.6 kHz (although it should be noted that this system is significantly different from the Navy's hull mounted sonar). In addition, the system had some low frequency components (below 1 kHz) which may have been an artifact of the acoustic equipment. This possible artifact may have affected the response of the whales to both the control and sonar playback conditions.

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

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, humpback whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.2.14.

4.2.3.6 Status and Trends

It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978a). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whales to roughly 1,000 (Perry et al. 1999). 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. 2011). 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. 2008b). Modeled abundance increase in southeastern Alaska was 5.1 percent annually from 1986 to 2008 (Hendrix et al. 2012); a more specific estimate from Glacier Bay, the site of a long-term monitoring study over roughly the same time frame found a rate of increase of 4.4 percent (Saracco et al. 2013). For Western North Pacific humpbacks, an annual rate of growth of 6.9 percent was estimated for the years between

1991-1993 and 2004-2006 (Calambokidis et al. 2008a). However, this growth rate could be biased upwards by the comparison of earlier estimates based on photo-identification records from Ogasawara and Okinawa with 2004-2006 estimates based on the more expansive records collected in Ogasawara, Okinawa, and the Philippines during the SPLASH program. From the years 2004-2006, the SPLASH program estimated humpback abundance in the Western North Pacific to be around 1,000 individuals. Most recently, Wade et al. (2016) estimated an abundance of 1,059 individuals for the Western North Pacific DPS (81 FR 62259). Because of the potential biases mentioned previously with the growth rate estimated by the SPLASH program, the final rule to revise the humpback whale listing determined the Western North Pacific DPS has an "unknown trend."

The Navy's NMSDD estimates 0.00089 humpback whales per km^2 in the MITT action area (DoN 2014). Humpback whales are infrequently sighted during the Navy's routine aerial surveys of Farallon de Medinilla (FDM); two sightings in 2006 (January and March), both close to the island, and another sighting in February 2007, 18 miles (mi.) (29 kilometers [km]) north of Saipan (Vogt 2008). During a ship survey in the action area (January through April 2007), humpback whales were observed and/or detected acoustically in both deep (2,625 to 3,940 feet [ft.] [800 to 1,200 m]) and shallow (1,234 ft. [374 m]) waters northeast of Saipan (Fulling et al. 2011). Acoustic detections of humpback song were also made during these sightings as well as on other occasions (Fulling et al. 2011). These observations suggest that there could be a small wintering population of humpback whales transiting during migration through the action area (Fulling et al. 2011; Ligon et al. 2011). During Navy-funded monitoring for the MIRC in 2009 through 2014, no humpback whales were sighted (Fulling et al. 2011; HDR 2011; HDR 2012a; Hill et al. 2013; Hill et al. 2011; Ligon et al. 2011; Oleson and Hill 2010). However, humpback whales were observed off Saipan in winter 2015 and 2016 (Hill et al. 2017; Hill et al. 2016).

4.2.3.7 Threats

Similar to some other large whale species, energy development, vessel strikes, and interactions with fisheries have been identified as threats to humpback whales from the Western North Pacific DPS. The Sea of Okhostsk currently has a high level of energy exploration and development and these activities are expected to expand with little regulation or oversight. The final rule to revise the humpback ESA listing identified energy exploration as a medium threat for the Okinawa/Phillipines portion of the Western North Pacific DPS. Some degree of illegal, unreported, or unregulated exploitation, including "commercial bycatch whaling" has been documented in Japan and South Korea (Baker et al. 2000; Baker et al. 2006a). In Japan and Korea, it is legal to kill and sell any entangled whale as long as the take is reported, though there is concern that this provides some level of incentive to entangle whales (Lukoschek et al. 2009). Entanglement in fisheries was identified a high threat to the Western North Pacific DPS in the final rule to revise the humpback ESA listing (81 FR 62259). Some degree of poaching is reported to occur in Korean waters and is suspected off Japan. Whaling was identified as a medium threat for the Western North Pacific DPS in the proposed rule to revise the humpback

ESA listing (80 FR 22303). Though specific information on prey abundance and competition with whales in the Western North Pacific is not available, the final rule identified competition with fisheries as a medium threat to humpbacks from the Western North Pacific DPS (81 FR 62259). The range of Western North Pacific DPS humpback whales includes some of the world's largest centers of human activity and shipping (81 FR 62259). Because of this level of shipping traffic and the co-occurrence of humpback whales in these areas, ship strikes are a concern. However, reporting of large whale ship strikes by Japan and Korea within the range of this DPS is suspected to be poor (Bettridge et al. 2015). The potential for ship strikes to occur is expected to increase with time along with increases in shipping traffic (81 FR 62259). The final rule to revise the humpback ESA listing identified vessel strikes as a medium threat for this DPS.

In summary, the final rule to revise the humpback ESA listing identified the following threats that may impact the survival and recovery of humpback whales from the Western North Pacific DPS: energy development, competition with fisheries, whaling, entanglement, and vessel collisions (81 FR 62259). All other potential threats identified in the proposed and final rules, including underwater noise from human activities, were considered to have no or minor impact on the population size and/or growth rate, or are unknown, for the Western North Pacific DPS.

4.2.3.8 Critical Habitat

The NMFS has not designated critical habitat for humback whales from the Western North Pacific DPS.

4.2.4 Sei Whale

Sei whales (pronounced "say" or "sigh"; *Balaenoptera borealis*) are members of the baleen whale family and are considered one of the "great whales" or rorquals. Two subspecies of sei whales are recognized, *B. b. borealis* in the Northern Hemisphere and *B. b. schlegellii* in the Southern Hemisphere. These large animals can reach lengths of 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 the fastest swimmer among the great whales, reaching bursts of speed in excess of 20 knots. When a sei whale begins a dive it usually submerges by sinking quietly below the surface, often remaining only a few meters deep, leaving a series of swirls or tracks as it move its flukes. When at the water's surface, sei whales can be sighted by a columnar or bushy blow that is about 10 to 13 feet (3 to 4 m) in height. The dorsal fin usually appears at the same time as the blowhole, when the animal surfaces to breathe. This species usually does not arch its back or raise its flukes when diving.

Sei whales 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.

4.2.4.1 Distribution

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

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 1985d; 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.

4.2.4.2 Population Structure

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

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 1974b). 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 1998a). 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 2001b; 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 1987b). In the north Pacific, sei whales are found feeding particularly along the cold eastern currents (Perry et al. 1999). 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.).

4.2.4.3 Reproduction

Very little is known regarding sei whale reproduction. Reproductive activities for sei whales occur primarily in winter. Gestation is about 12.7 months, calves are weaned at 6–9 months, and the calving interval is about 2–3 years (Gambell 1985c; Rice 1977). Sei whales become sexually mature at about age 10 (Rice 1977). Of 32 adult female sei whales harvested by Japanese whalers, 28 were found to be pregnant while one was pregnant and lactating during May–July 2009 cruises in the western North Pacific (Tamura et al. 2009).

4.2.4.4 Movement

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. 1999). Sei whales are often associated with deeper waters and areas along continental shelf edges (Hain et al. 1985b). This general offshore pattern is disrupted during occasional incursions into shallower inshore waters (Waring et al. 2004b). The species appears to lack a well-defined social structure and individuals are usually found alone or in small groups of up to six whales (Perry et al. 1999). When on feeding grounds, larger groupings have been observed (Gambell 1985c).

4.2.4.5 Feeding

Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2006). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Konishi et al. 2009; Mizroch et al. 1984b; Rice 1977). Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95 percent of their diets (Calkins 1986b). The dominant food for sei whales off California during June-August is northern anchovy, while in September-October whales feed primarily on krill (Rice 1977). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollack, capelin, and Atka mackerel (Nemoto and Kawamura 1977b). 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). Sei whales in the Southern Hemisphere may reduce direct interspecific competition with blue and fin whales by consuming a wider variety of prey and by arriving later to feeding grounds (Kirkwood 1992b). 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. In the North Pacific, sei whales appear to prefer feeding along the cold eastern currents (Perry et al. 1999). Sei whales have the flexibility to skim or engulf prey (Brodie and Vikingsson 2009).

4.2.4.6 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 1985d). 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 1985d).

Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2007a). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Mizroch et al. 1984a; Rice 1977). Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95 percent of their diets (Calkins 1986a). 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 1977a). 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 1992a). 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 two to five individuals are typically observed, but sometimes thousands may gather if food is abundant. However, these large aggregations may not be dependent on food supply alone, as they often occur during times of migration. Norwegian workers call the times of great sei whale abundance "invasion years." During mating season, males and females may form a social unit, but strong data on this issue are lacking.

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

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

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c), for the purposes of this analysis, sei whales were considered part of the low-frequency cetacean group, with a hearing range of 7 Hz to 22 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.2.14.

4.2.4.8 Status and Trends

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

Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000 to 38,000 whales by 1967, and reduced again to 20,600 to 23,700 whales by 1973. From 1910 to 1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Horwood 1987a; Perry et al. 1999). 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. 1984b). This estimate does not account for over-reporting by Soviet whalers, who took approximately 3,700 fewer individuals than were reported (Ivashchenko et al. 2013). 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 1977). The most current population estimate for sei whales in the entire north Pacific is 9,110 (Calambokidis et al. 2008b) and 25,000 individuals worldwide (Braham 1991).

NMFS has designated three stocks of sei whale for management purposes under the MMPA in the north Pacific: (1) the Hawaii stock, (2) the California/Oregon/Washington stock, and (3) the Alaska stock (Carretta et al. 2011). Little is known about the stock structure of sei whales in the action area. Various scientists have described the seasonal distribution of sei whales as occurring from 20° N to 23° N during the winter and from 35° N to 50° N during the summer (Horwood 2009; Masaki 1976b; Masaki 1977b; Smultea et al. 2010). However, sei whales were sighted during the 2007 survey of the action area, thus providing evidence that this species occurs south of 20°N in the winter (Fulling et al. 2011). Observations during the Navy-funded 2007 survey of the action area indicated this species most often occurs in deep water (10,381 to 30,583 ft. [3,164 to 9,322 m]). Most sei whale sightings were also associated with steep bathymetric relief (e.g., steeply sloping areas), including sightings adjacent to the Chamorro Seamounts east of the CNMI (Fulling et al. 2011). All confirmed sightings of sei whales were south of Saipan (approximately 15° N) with concentrations in the southeastern corner of the action area (Fulling et al. 2011). Sightings also often occurred in mixed groups with Bryde's whales. Sei whales were considered to be extralimital in the action area but during the 2007 systematic survey, sei whales were sighted on 16 occasions with a resulting abundance estimate of 166 individuals (CV = 0.49) (Fulling et al. 2011). No data on the current population trend are available; however, the population in the North Pacific is expected to have increased since sei whales began receiving protection in 1976 (Carretta et al. 2013a). A sei whale was also detected with sonobuoys on the January to February 2010 Oscar Elton Sette Cruise from Hawaii to Guam. However, the information we have did not allow us to determine if this detection occurred in the MITT action area. The Navy's NMSDD seasonally estimates between zero and 0.00029 sei whales per km^2 in the MITT action area (DoN 2014).

4.2.4.9 Natural Threats

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

4.2.4.10 Anthropogenic Threats

Human activities known to threaten sei whales include whaling, commercial fishing, and vessel strikes. 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. In 2009, 100 sei whales were killed during western North Pacific surveys (Bando et al. 2010).

Sei whales are occasionally killed in collisions with vessels; however, there are no known Navy ship strikes in the area. Of three sei whales that stranded along the U.S. Atlantic coast during 1975 to 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. 2005b; Nelson et al. 2007b). Two of these ship strikes were reported as having resulted in death. New rules for seasonal (June through December) slowing of vessel traffic in the Bay of Fundy to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to reduce sei whale ship strike mortality by 17 percent.

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

4.2.4.11 Critical Habitat

The NMFS has not designated critical habitat for sei whales.

4.2.5 Sperm Whale

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

The sperm whale is distinguished by its extremely large head, which takes up to 25 to 35 percent of its total body length. It is the only living cetacean that has a single blowhole asymmetrically situated on the left side of the head near the tip. Sperm whales have the largest brain of any animal (on average 17 pounds (7.8 kg) in mature males). However, compared to their large body size, the brain is not exceptional in size. Sperm whales are mostly dark gray, but oftentimes the interior of the mouth is bright white, and some whales have white patches on the belly. Their flippers are paddle-shaped and small compared to the size of the body, and their flukes are very triangular in shape. They have small dorsal fins that are low, thick, and usually rounded.

4.2.5.1 Distribution

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

4.2.5.2 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). Chemical analysis also suggest significant differences in diet for animals captured in different regions of the North Atlantic. However, vocal dialects indicate parent-offspring transmission that indicates differentiation in populations (Rendell et al. 2011). Vocal differences exist not only across ocean basins, but also over much smaller spatial scales (Amano et al. 2014). Therefore, population-level differences may be more extensive than are currently understood.

The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). The NMFS recognizes six stocks under the MMPA: three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; (Perry et al. 1999; Waring et al. 2004b). 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 et al. 2008). Matrilinear groups in the eastern Pacific share nuclear DNA within broader clans, but North Atlantic matrilinear groups do not share this genetic heritage (Whitehead et al. 2012).

Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 40^{o} N in winter (Gosho et al. 1984; Miyashita et al. 1995 as cited in Carretta et al. 2005; Rice 1974b). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995a; Dohl et al. 1983; Forney et al. 1995a; Lee 1993; Mobley Jr . et al. 2000; Rice 1960; Shallenberger 1981), but they reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974b). They are seen in every season except winter (December-February) off Washington and Oregon (Green et al. 1992). 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), although occurrence seems to be continuance from California through Hawaii (Barlow and Taylor 2005). The discreteness of the latter two areas remains uncertain (Mesnick et al. 2011).

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

4.2.5.3 Movement

Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring et al. 1993a) where adult males join them to breed. 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 et al. 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 kilometers 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.

4.2.5.4 Habitat

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

Sperm whales are frequently found in locations of high productivity due to upwelling or steep underwater topography, such as continental slopes, seamounts, or canyon features (Jaquet and Whitehead 1996a; Jaquet et al. 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. 2000a; Davis et al. 2000c; Davis et al. 2000d; Davis et al. 2000e; Davis et al. 2002a; Wormuth et al. 2000). 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 et al. 1996; Waring et al. 1993a). Sperm whales over George's Bank were associated with surface temperatures of 23.2 to 24.9° C (Waring et al. 2003).

Local information is inconsistent regarding some aspects of sperm whale habitat utilization. Gregr and Trites (2001a) reported that female sperm whales off British Columbia were relatively unaffected by the surrounding oceanography. However, Tynan et al. (2005a) reported increased sperm whales densities with strong turbulence-associated topographic features along the continental slope near Heceta Bank.

4.2.5.5 Reproduction

Female sperm whales become sexually mature at an average of 9 years or 8.25 to 8.8 m (Kasuya 1991). Males reach a length of 10 to 12 m at sexual maturity and take 9 to 20 years to become sexually mature, but require another 10 years to become large enough to successfully breed (Kasuya 1991; Würsig et al. 2000). Mean age at physical maturity is 45 years for males and 30 years for females (Waring et al. 2004b). Adult females give birth after roughly 15 months of gestation and nurse their calves for 2 to 3 years (Waring et al. 2004b). The calving interval is estimated to be every 4 to 6 years between the ages of 12 and 40 (Kasuya 1991; Whitehead et al. 2008). In the North Pacific, female sperm whales and their calves are usually found in tropical and temperate waters year round, while it is generally understood that males move north in the summer to feed in the Gulf of Alaska, Bering Sea, and waters off of the Aleutian Islands (Kasuya and Miyashita 1988). It has been suggested that some mature males may not migrate to breeding grounds annually during winter, and instead may remain in higher latitude feeding grounds for more than 1 year at a time (Whitehead and Arnbom 1987).

Sperm whale age distribution is unknown, but sperm whales are believed to live at least 60 years (Rice 1978b). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980). In addition to anthropogenic threats, there is evidence that sperm whale age classes are subject to predation by killer whales (Arnbom et al. 1987; Pitman et al. 2001).

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). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

4.2.5.6 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 et al. 1993; Watkins et al. 1985a). However, dives are generally shorter (25 to 45 min) and shallower (400 to 1,000 m). Dives are separated by 8 to 11 min rests at the surface (Gordon 1987; Jochens et al. 2006; Papastavrou et al. 1989; Watwood et al. 2006; Würsig et al. 2000). Sperm whales typically travel approximately 3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Unlike other cetaceans, there is a preponderance of dive information for this species, most likely because it is the deepest diver of all cetacean species 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 (Barlow et al. 1997b; Watkins and Schevill 1977), although Berzin (1971b) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956a; Rice 1989a). Sperm whales have been observed near Long Island, New York, in water between 40 and 55 m deep (Scott and Sadove 1997a).

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 1996b). 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. 2000b; Davis et al. 2000b; Davis et al. 2002b). 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 1996b; Waring et al. 1993b). Sperm whales over George's Bank were associated with surface temperatures of 23.2 to 24.9 °C (Waring et al. 2004a).

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

4.2.5.7 Feeding

Sperm whales appear to feed regularly throughout the year (NMFS 2006d). It is estimated they consume about 3 to 3.5 percent of their body weight daily (Lockyer 1981). They seem to forage mainly on or near the bottom, often ingesting stones, sand, sponges, and other non-food items (Rice 1989d). A large proportion of a sperm whale's diet consists of low-fat, ammoniacal, or luminescent squids (Clarke 1996; Clarke 1980b; Martin and Clarke 1986). While sperm whales feed primarily on large and medium-sized squids, the list of documented food items is fairly long and diverse. Prey items include other cephalopods, such as octopi, and medium- and large-sized demersal fishes, such as rays, sharks, and many teleosts (Angliss and Lodge 2004; Berzin 1972;

Clarke 1977; Clarke 1980a; Rice 1989d). The diet of large males in some areas, especially in high northern latitudes, is dominated by fish (Rice 1989d). In some areas of the North Atlantic, however, males prey heavily on the oil-rich squid *Gonatus fabricii*, a species also frequently eaten by northern bottlenose whales (Clarke 1997).

4.2.5.8 Vocalization and Hearing

Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broad-band clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200 to 236 dB re 1μPa), although lower source level energy has been suggested at around 171 dB re 1 μPa (Goold and Jones 1995; Møhl et al. 2003; Weilgart and Whitehead 1993a; Weilgart and Whitehead 1997a). Most of the energy in sperm whale clicks is concentrated at around 2 kHz to 4 kHz and 10 kHz to 16 kHz (Goold and Jones 1995; NMFS 2006d; Weilgart and Whitehead 1993a). The highly asymmetric head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Cranford 1992; Norris and Harvey 1972; Norris and Harvey. 1972). Long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993a; Weilgart and Whitehead 1997a). However, clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Weilgart and Whitehead 1993a). They may also aid in intra-specific communication. Another class of sound, “squeals”, are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

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

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

When sperm whales are socializing, they tend to repeat series of group-distinctive clicks (codas), which follow a precise rhythm and may last for hours (Watkins and Schevill 1977). Codas are shared between individuals in a social unit and are considered to be primarily for intragroup communication (Rendell and Whitehead 2004; Weilgart and Whitehead 1997b). Recent research in the South Pacific suggests that in breeding areas the majority of codas are produced by mature females (Marcoux et al. 2006). Coda repertoires have also been found to vary geographically and are categorized as dialects, similar to those of killer whales (Pavan et al. 2000; Weilgart and Whitehead 1997b). For example, significant differences in coda repertoire have been observed between sperm whales in the Caribbean and those in the Pacific (Weilgart and Whitehead 1997b). Three coda types used by male sperm whales have recently been described from data collected over multiple years: these include codas associated with dive cycles, socializing, and alarm (Frantzis and Alexiadou 2008).

Direct measures of sperm whale hearing have been conducted on a stranded neonate using the auditory brainstem response technique: the whale showed responses to pulses ranging from 2.5 kHz to 60 kHz and highest sensitivity to frequencies between 5 kHz to 20 kHz (Ridgway and Carder 2001). Other hearing information consists of indirect data. For example, the anatomy of the sperm whale's inner and middle ear indicates an ability to best hear high-frequency to ultrasonic hearing (Ketten 1992). The sperm whale may also possess better low-frequency hearing than other odontocetes, although not as low as many baleen whales (Ketten 1992). Reactions to anthropogenic sounds can provide indirect evidence of hearing capability, and several studies have made note of changes seen in sperm whale behavior in conjunction with these sounds. For example, sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echo-sounders and submarine sonar (Watkins et al. 1985b; Watkins and Schevill 1975a). In the Caribbean, Watkins et al. (1985b) observed that sperm whales exposed to 3.25 kHz to 8.4 kHz pulses (presumed to be from submarine sonar) interrupted their activities and left the area. Similar reactions were observed from artificial noise generated by banging on a boat hull (Watkins et al. 1985b). André et al. (1997)reported that foraging whales exposed to a 10 kHz pulsed signal did not ultimately exhibit any general avoidance reactions: when resting at the surface in a compact group, sperm whales initially reacted strongly, and then ignored the signal completely (André et al. 1997). Thode et al. (2007) observed that the acoustic signal from the cavitation of a fishing vessel's propeller (110 dB re 1 μPa^2 between 250 Hz and 1.0 kHz) interrupted sperm whale acoustic activity and resulted in the animals converging on the vessel. The full range of functional hearing for the sperm whale is estimated to occur between approximately 150 Hz and 160 kHz, placing them among the group of cetaceans that can hear mid-frequency sounds (Southall et al. 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 (Miller et al. 2012; Sivle et al. 2012; Southall et al. 2011b).

To facilitate the acoustic and effects analyses, marine mammals were divided into functional hearing groups (based on their hearing range), and the same criteria and thresholds were used for all species within a group. Consistent with Southall et al. (2007c) and for the purposes of this analysis, sperm whales were considered part of the mid-frequency cetacean group, with a nominal hearing range between approximately 150 Hz and up to 160 kHz (Finneran and Jenkins 2012). Additional detail on the acoustic effects analysis is presented in section 3.1 of this opinion and additional detail on the criteria used in the analysis are presented in section 6.2.14.

4.2.5.9 Status and Trends

Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 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 are sighted off Oregon in every season except winter (Green et al. 1992). However, sperm whales are found off California year-round (Barlow 1995a; Dohl et al. 1983; Forney et al. 1995a), with peak abundance from April to mid-June and from August to mid-November (Rice 1974b).

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947-1987. This estimate does not account for under-reporting by Soviet whalers, who took approximately 31,000 more individuals than were reported (Ivashchenko et al. 2013). 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, uncertainly 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.

NMFS has designated three stocks of sperm whale for management purposes under the MMPA in the north Pacific: (1) the Hawaii stock, (2) the California/Oregon/Washington stock, and (3) the Alaska stock (Carretta et al. 2013a). Little is known about the stock structure of sperm whales in the action area.

Preliminary results of a coda review (Norris et al. 2012) “provide new insights into the sperm whale social units occurring within this region of the western Pacific Ocean.” The presence of the 'short' and 'regular' vocal clans identified from the acoustic encounters in this dataset can be used as putative indicators of sperm whale stock structure for this region (Marcoux et al. 2006). The vocal repertoires identified in the review may provide information on the occurrence of vocal clans in the western Pacific Ocean. The vocal repertoires identified (Norris et al. 2012) suggest a potential cultural and acoustic link to vocal clans found in the eastern tropical Pacific Ocean and greatly extends the known range for the 'regular' and 'short' vocal clans. Whaling records demonstrate sightings year-round in the action area (Townsend 1935). There are also at least four stranding records for this area spanning the period from 1962 to 2013 (Eldredge 1991; Eldredge 2003; Kami and Lujan 1976; Kerrigan 2013). During the Navy-funded survey in 2007, there were multiple sightings that included young calves and large bulls (Fulling et al. 2011). These findings are consistent with an earlier sighting of a group of sperm whales that included a newborn calf off the west coast of Guam (Eldredge 2003). During the 2007 survey, sperm whales were observed in waters 2,670 to 32,584 ft. (809–9,874 m) deep (Fulling et al. 2011). During a small boat survey around Guam and Saipan in February and early March of 2010, there were two sperm whale sightings: (1) a group of nine animals off Orote Point, Guam, inshore from the 1,640 ft. (500 m) isobath; and (2) a group of six animals northwest of Saipan in waters greater than 3,281 ft. (1,000 m) deep (Ligon et al. 2011). A group of 10 sperm whales was also sighted during small boat surveys off western Guam in waters approximately 3,940 ft. deep (1,200 m) in March 2012 (HDR 2012a).

The sperm whale was the most frequently sighted cetacean (21 sightings) during the 2007 survey with acoustic detections almost three times higher (61) than visual detections in the field (Fulling et al. 2011). Post processing of the acoustic data resulted in 91 distinct localizations of individual sperm whales. Based on a preliminary analysis, the distribution of sperm whales appeared to be clustered in three main regions of MIRC, the northeast, central, and southwest portions, with a few others in the trench and offshore regions (Norris et al. 2012). Line-transect abundance estimates derived from these survey data yielded an estimate of 705 (CV = 0.60) sperm whales in the action area (Fulling et al. 2011). The Navy's NMSDD estimates 0.00291 sperm whales per km^2 in the MITT action area (DoN 2014).

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4.2.5.10 Natural Threats

Sperm whales are known to be occasionally predated upon by killer whales (Jefferson and Baird 1991; Pitman et al. 2001) and large sharks (Best et al. 1984) and harassed by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989c; Weller et al. 1996; Whitehead 1995). 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 of strandings remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

4.2.5.11 Anthropogenic Threats

Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959 to 1983). However, other estimates have included 436,000 individuals killed between 1800 and 1987 (Carretta et al. 2005). However, all of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947–1973. 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 and Zemsky 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber

2004). Japan maintains an active whaling fleet, killing up to 10 sperm whales annually (IWC 2008). In 2009, one sperm whale was killed during western North Pacific surveys (Bando et al. 2010).

Whale-watching vessels may influence sperm whale behavior (Richter et al. 2006).

In U.S. waters in the Pacific Ocean, sperm whales are known to have been incidentally captured only in drift gillnet operations, which killed or seriously injured an average of nine sperm whales per year from 1991 to 1995 (Barlow et al. 1997b). Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Hill and Demaster 1998b; Rice 1989b). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on fish caught in longline gear in the Gulf of Alaska. During 1997, the first entanglement of a sperm whale in Alaska’s longline fishery was recorded, although the animal was not seriously injured (Hill and Demaster 1998b). The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not yet clear. An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George’s Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine. In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine. Ten sperm whale entanglements have been observed in U.S. fisheries since 1990 in the Pacific (Carretta and Enriquez 2012). Two additional whales have been found to die from ingestion of fishing gear (Jacobsen et al. 2010b). Overall, it is estimated that 3.8 sperm whales die annually along the U.S. west coast due to fisheries interaction (Carretta et al. 2013b).

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 do not appear to accumulate chromium at higher levels.

Small changes in reproductive parameters, such as the loss of adult females, can significantly alter the population trajectory of sperm whale populations (Chiquet et al. 2013).

Ingestion of marine debris can have fatal consequences even for large whales. In 1989, a stranded sperm whale along the Mediterranean was found to have died from ingesting plastic that blocked its' digestive tract (Viale et al. 1992). A sperm whale examined in Iceland had a lethal disease thought to have been caused by the complete obstruction of the gut with plastic marine debris (Lambertsen 1990). The stomach contents of two sperm whales that stranded separately in California included extensive amounts of discarded fishing netting (NMFS 2009b). A fifth individual from the Pacific was found to contain nylon netting in its stomach when it washed ashore in 2004 (NMFS 2009b). In March 2012, a sperm whale stranded dead, apparently dying as a result of plastic ingestion (de Stephanis et al. 2013).

There have not been any recent documented ship strikes involving sperm whales in the eastern North Pacific, although there are a few records of ship strikes in the 1990s. Two whales described as "possibly sperm whales" are known to have died in U.S. Pacific waters in 1990 after being struck by vessels (Barlow et al. 1997a). There is an anecdotal record from 1997 of a fishing vessel that struck a sperm whale in southern Prince William Sound in Alaska, although the whale did not appear to be injured (Laist et al. 2001). More recently in the Pacific, two sperm whales were struck by a ship in 2005, but it is not known if these ship strikes resulted in injury or mortality (NMFS 2009a). The lack of recent evidence should not lead to the assumption that no mortality or injury from collisions with vessels occurs as carcasses that do not drift ashore may go unreported, and those that do strand may show no obvious signs of having been struck by a ship (NMFS 2009a). Worldwide, sperm whales are known to have been struck 17 times out of a total record of 292 strikes of all large whales, 13 of which resulted in mortality (Jensen and Silber 2003a; Laist et al. 2001). Given the current number of reported cases of injury and mortality, it does not appear that ship strikes are a significant threat to sperm whales (Whitehead 2003).

4.2.5.12 Critical Habitat

The NMFS has not designated critical habitat for sperm whales.

4.2.6 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.

4.2.6.1 Species 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).

4.2.6.2 Populations

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. 2002a).

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.

4.2.6.3 Growth and Reproduction

Most green sea turtles exhibit particularly slow growth rates, which have been attributed to their largely plant-eating diet (Bjorndal 1982). Growth rates of juveniles vary substantially among populations, ranging from <1 cm/year (Green 1993) to >5 cm/year (McDonald Dutton and Dutton 1998), likely due to differences in diet quality, duration of foraging season (Chaloupka et al. 2004b), and density of turtles in foraging areas (Balazs and Chaloupka 2004; Bjorndal et al. 2000a; Seminoff et al. 2002b). Hart et al. (2013a) found growth rates of green sea turtles in the U.S. Virgin Islands to range from 0 to 9.5 cm annually (mean of 4.1, SD 2.4). The largest growth rates were in the 30 to 39 cm class. If individuals do not feed sufficiently, growth is stunted and apparently does not compensate even when greater-than-needed resources are available (Roark et al. 2009). In general, there is a tendency for green sea turtles to exhibit monotonic growth (declining growth rate with size) in the Atlantic and non-monotonic growth (growth spurt in mid-size classes) in the Pacific, although this is not always the case (Balazs and Chaloupka 2004; Chaloupka and Musick 1997; Seminoff et al. 2002b). It is estimated that green sea turtles reach a maximum size just under 100 cm in carapace length (Tanaka 2009). A female-bias has been identified from studies of green sea turtles (Wibbels 2003).

Consistent with slow growth, age-to-maturity for green sea turtles appears to be the longest of any sea turtle species and ranges from ~20 to 40 years or more (Balazs 1982; Chaloupka et al. 2004b; Chaloupka and Musick 1997; Frazer and Ehrhart 1985b; Hirth 1997; Limpus and Chaloupka 1997; Seminoff et al. 2002b; Zug et al. 2002; Zug and Glor 1998). Estimates of reproductive longevity range from 17 to 23 years (Carr et al. 1978; Chaloupka et al. 2004b; Fitzsimmons et al. 1995). Considering that mean duration between females returning to nest ranges from 2 to 5 years (Hirth 1997), these reproductive longevity estimates suggest that a female may nest 3 to 11 seasons over the course of her life. Each female deposits 1 to 7 clutches (usually 2 to 3) during the breeding season at 12 to 14 day intervals. Mean clutch size is highly variable among populations, but averages 110 to 115 eggs/nest. Females usually have 2 to 4 or more years between breeding seasons, whereas males may mate every year (Balazs 1983). Based on reasonable means of three nests per season and 100 eggs per nest (Hirth 1997), a female may deposit 9 to 33 clutches, or about 900 to 3,300 eggs, during her lifetime. Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Once hatched, sea turtles emerge and orient towards a light source, such as light shining off the ocean. They enter the sea in a “frenzy” of swimming activity, which decreases rapidly in the first few hours and gradually over the first several weeks (Ischer et al. 2009; Okuyama et al. 2009). Factors in the ocean environment have a major influence on reproduction (Chaloupka 2001; Limpus and Nicholls 1988; Solow et al. 2002). It is also apparent that during years of heavy nesting activity, density dependent factors (beach crowding and digging up of eggs by nesting females) may impact hatchling production (Tiwari et al. 2005; Tiwari et al. 2006). Precipitation, proximity to the high tide line, and nest depth can also significantly affect nesting success (Cheng et al. 2009). Precipitation can also be significant in sex determination, with greater nest moisture resulting in a higher proportion of males (Leblanc and Wibbels 2009). Green sea turtles often return to the same foraging areas following nesting migrations (Broderick et al. 2006; Godley et al. 2002). Once there, they move within specific areas, or home ranges, where they routinely visit specific localities to forage and rest (Godley et al. 2003; Makowski et al. 2006; Seminoff and Jones 2006; Seminoff et al. 2002a; Taquet et al. 2006). It is also apparent that some green sea turtles remain in pelagic habitats for extended periods, perhaps never recruiting to coastal foraging sites (Pelletier et al. 2003).

In general, survivorship tends to be lower for juveniles and subadults than for adults. Adult survivorship has been calculated to range from 0.82 to 0.97 versus 0.58 to 0.89 for juveniles (Chaloupka and Limpus 2005; Seminoff et al. 2003; Troëng and Chaloupka 2007), with lower values coinciding with areas of human impact on green sea turtles and their habitats (Bjorndal et al. 2003; Campbell and Lagueux 2005).

4.2.6.4 Habitat

Green turtles appear to prefer waters that usually remain around 20º C in the coldest month, but may occur considerably north of these regions during warm-water events, such as El Niño. Stinson (1984b) found green turtles to appear most frequently in U.S. coastal waters with temperatures exceeding 18º C. Further, green sea turtles seem to occur preferentially in drift lines or surface current convergences, probably because of the prevalence of cover and higher prey densities that associate with flotsam. For example, in the western Atlantic Ocean, drift lines commonly containing floating *Sargassum* spp. are capable of providing juveniles with shelter (NMFS and USFWS 1998d). Underwater resting sites include coral recesses, the underside of ledges, and sand bottom areas that are relatively free of strong currents and disturbance. Available information indicates that green turtle resting areas are near feeding areas (Bjorndal and Bolten 2000). Strong site fidelity appears to be a characteristic of juveniles green sea turtles along the Pacific Baja coast (Senko et al. 2010b). Recent tagging data from off the northwestern coast of Saipan and the western coast of Tinian also indicate strong site fidelity (Jones and Van Houtan 2014).

4.2.6.5 Feeding

While offshore and sometimes in coastal habitats, green sea turtles are not obligate plant-eaters as widely believed, and instead consume invertebrates such as jellyfish, sponges, sea pens, and pelagic prey (Godley et al. 1998; Hart et al. 2013b; Hatase et al. 2006b; Heithaus et al. 2002; Parker and Balazs in press; Seminoff et al. 2002a). A shift to a more herbivorous diet occurs when individuals move into neritic habitats, as vegetable mater replaces an omnivorous diet at around 59 cm in carapace length off Mauritania (Cardona et al. 2009). This transition may occur rapidly starting at 30 cm carapace length, but animal prey continue to constitute an important nutritional component until individuals reach about 62 cm (Cardona et al. 2010). Foraging within seagrass ecosystems by green sea turtles can be significant enough to alter habitat and ecological parameters, such as species composition (Lal et al. 2010). Although populations can consume a variety of prey and be considered generalists as a whole, individuals maintain a highly-selective diet over long time frames (Vander Zanden et al. 2013).

4.2.6.6 Migration and Movement

Green sea turtles are highly mobile and undertake complex movements through geographically disparate habitats during their lifetimes (Musick and Limpus 1997b; Plotkin 2003). The periodic migration between nesting sites and foraging areas by adults is a prominent feature of their life history. After departing as hatchlings and residing in a variety of marine habitats for 40 or more years (Limpus and Chaloupka 1997), green sea turtles make their way back to the same beach from which they hatched (Carr et al. 1978; Meylan et al. 1990). At approximately 20 to 25 cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997a). Green sea turtles spend the majority of their lives in coastal foraging grounds (MacDonald et al. 2012). These areas include both open coastline and protected bays and lagoons. While in these areas, green sea turtles rely on marine algae and seagrass as their primary dietary constituents, although some populations also forage heavily on invertebrates. Although green sea turtles in tropical areas seem to undergo a sudden, permanent switch in habitat from oceanic to neritic habitats, individuals in more temperate areas seem to utilize a wider array of habitats dependent upon oceanographic conditions (González Carman et al. 2012). There is some evidence that individuals move from shallow seagrass beds during the day to deeper areas at night (Hazel 2009). However, avoidance of areas of greater than 10 m when moderate depths of 5 to 10 m with sea grass beds has been found, with speed and displacement from capture locations being similar at night as during the daytime (Senko et al. 2010a).

4.2.6.7 Occurrence in the MITT Study Area

Green turtles have been documented nesting on many beaches in Guam and the surrounding islands (e.g., (Brindock 2013)), though long-term information regarding nesting population trends in in the area are not available. There is, however, indication that the Marianas may provide more important foraging nearshore habitat than nesting (Kolinski et al. 2001; Pultz et al. 1999). Aerial surveys conducted by the Guam Division of Aquatic and Wildlife Resources indicate the year-round presence of green sea turtles in Guam’s nearshore waters (Kolinski et al. 2001; NMFS and USFWS 1998d; Pultz et al. 1999). Aggregations of foraging and resting green turtles are often seen in close proximity to Guam’s well-developed seagrass beds and reef flats, which are found in Cocos Lagoon, Apra Harbor, along Tarague Beach and Hila’an; in deeper waters south of Falcona Beach; and at several other locations throughout the island’s shelf (DON 2003b). Guam Division of Aquatic and Wildlife Resources aerial surveys have identified turtles within Agat Bay, and stranded sea turtles have been recovered from the bay (including one with spear gun injuries). In August 2013, PIFSC researchers conducted snorkeling and boat surveys in Guam's Cocos Lagoon and observed 9 green turtles over the two days of surveys. All individuals observed were juveniles (Jones and Houtan 2014).

On Tinian, green turtle abundance and densities are highest along the island's relatively uninhabited east coast. The most recent estimate of the number of green turtles inhabiting the nearshore waters around Tinian was 832 turtles in 2001 (Kolinski et al. 2006) and densities of approximately 11.8 animals per km^2. Between November 2013 and March 2014 the CNMI Department of Lands and Natural Resources captured 54 unique green sea turtles (and 3 recaptures), 44 around Saipan and 10 around Tinian. Catch per unit effort for all sea turtles was 3.75 turtle catches per dive-hour (Palacios et al. 2014). In August 2013, a PIFSC researcher conducted snorkeling and boat surveys off the northwestern coast of Saipan and the western coast of Tinian. The team captured four sub-adult green turtles over a four day period (Jones and Houtan 2014).

Green turtles are not as abundant at FDM as they are at some of the larger islands of the Marianas chain. At FDM, at least 9 green turtles were observed during underwater surveys in both 1999 and 2000, at least 12 green turtles were observed during surveys in 2001, and 4 were observed at the northern end of the island in 2003 (DoN 2005a). Annual diver surveys between 2005 and 2012 observed between three (2005) and nine (2009) green sea turtles at FDM (DoN 2013a). Most green turtles at FDM were found either swimming over the reef platform or resting in holes or caves (DoN 2005a) . Due to strong current and tidal conditions, the beaches at FDM are very susceptible to inundation and are highly unsuitable for nesting (DoN 2003a). Also, seagrasses and benthic algae are relatively sparse around the island and can probably support no more than a few green turtles at a time (NMFS and USFWS 1998d). Seven sea turtles were documented in 2006 and 19 in 2007 during monthly monitoring (helicopter surveys) of FDM (DoN 2010c). Monthly observations are usually low (between one and three turtle sightings); however, 12 turtles were observed in waters off FDM on 13 November 2007 (DoN 2010c). Identifying sea turtles to the species level is not possible due to safe flying heights of the helicopter, although due to the higher abundance of green sea turtles relative to hawksbill turtles, the majority of sea turtle observations are assumed to be green sea turtles (DoN 2010c).

Based on the above information, green turtles are expected to occur year round in all shelf waters of the action area from FDM to Guam. Around the larger islands, green turtle occurrence is concentrated in waters less than 328 ft. (100.01 m) deep, approximately 11.8 animals per km^2 (4.6 mi.^{2}). It is at these water depths where green turtle foraging and resting habitats (e.g., fringing reefs, reef flats, and seagrass beds) are usually found. Although there may not be long-term data available for Guam or CNMI, data from other Pacific regions show that green sea turtles exhibit strong site fidelity to nearshore foraging habitats for extended periods of time (Balazs 1995; Balazs and Chaloupka 2004). The Navy's NMSDD estimates 0.2968, 11.8, and 0.000391 green sea turtles per km^2 in the nearshore Guam, nearshore Tinian and other islands, and pelagic portions of the MITT action area, respectively (DoN 2014). Nesting females and early juveniles are known to move through oceanic waters of the Marianas chain during their reproductive and developmental migrations (Kolinski et al. 2006), but likely do not do so in large numbers. Additionally, sea turtles from more distant areas may migrate to the MITT study area to forage. For example, genetic analysis has indicated that approximately 3 percent of green sea turtles found foraging arounc CNMI were coming from French Frigate Shoals (Peter Dutton, NMFS, personal communication to Eric MacMillan June 1, 2016).

4.2.6.8 Diving

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume 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 (Hazel et al. 2009; NMFS and USFWS 1998d). Recent data from Australia indicate green sea turtles rarely dive deep, staying in upper 8 m of the water column (Hazel et al. 2009). Daytime dives were shorter and shallower than were nighttime dives (Ballorain et al. 2013; Hazel et al. 2009). Green turtles migrating between the northwestern and main Hawaiian Islands reached a maximum depth greater than 445 ft. (135.6 m) at night (the deepest dives ever recorded for a green turtle) with a mean maximum night dive depth of 115 to 164 ft. (35–50 m) but only 14.1 ft. (4.3 m) during the day (Rice and Balazs 2008b). In their coastal habitat, green turtles typically make dives shallower than 100 ft. (30.5 m) (Hatase et al. 2006a; Hays et al. 2000; Hochscheid et al. 2005; Houghton et al. 2002) and often do not exceed 55 ft. (16.8 m) (Hays et al. 2000; Rice and Balazs 2008a), although they are known to feed and rest at depths of 65 to 165 ft. (19.8–50.3 m) (Balazs 1980a; Brill et al. 1995a). Also, time spent resting and dive duration increased significantly with decreases in seasonal water temperatures. Subadults routinely dive to 20 m for 9 to 23 min, with a maximum recorded dive of over 1 h (Brill et al. 1995b; I-Jiunn 2009). Green sea turtles along Taiwan may rest during long, shallow dives (I-Jiunn 2009). Dives by females may be shorter in the period leading up to nesting (I-Jiunn 2009).

4.2.6.9 Hearing

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). Piniak et al. (2012) found green sea turtle juveniles capable of hearing underwater sounds at frequencies of 50 Hz to 1,600 Hz (maximum sensitivity at 200 to 400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a). Based upon auditory brainstem responses green sea turtles have been measured to hear in the 50 Hz to 1,600 Hz range (Dow et al. 2008), with greatest response at 300 Hz (Yudhana et al. 2010); a value verified by Moein Bartol and Ketten (2006). Other studies have found greatest sensitivities are 200 to 400 Hz for the green turtle with a range of 100 Hz to 500 Hz (Moein Bartol and Ketten 2006; Ridgway et al. 1969) and around 250 Hz or below for juveniles (Bartol et al. 1999c). However, Dow et al. (2008) found best sensitivity between 50 Hz and 400 Hz.

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles respond best to sounds between 200 Hz and 700 Hz, with slow declines below 100 Hz and rapid declines above 700 Hz, and almost no sensitivity above 3,000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1,000 Hz and almost no responses beyond 3,000 or 4,000 Hz (Patterson 1966).

4.2.6.10 Natural Threats

Herons, gulls, dogfish, and sharks prey upon hatchlings. Adults face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can be lethal. Green sea turtles are also threatened by a disease called fibropapillomatosis (Work et al. 2015). Additionally, a to-date unidentified virus may aid in the development of fibropapillomatosis (Work et al. 2009). Predators (primarily of eggs and hatchlings) also include dogs, pigs, rats, crabs, sea birds, reef fishes, and groupers (Bell et al. 1994; Witzell 1981). Green sea turtles with an abundance of barnacles have been found to have a much greater probability of having health issues (Flint et al. 2009). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in excess of 90 percent of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

4.2.6.11 Anthropogenic Threats

Major anthropogenic impacts to the nesting and marine environment affect green sea turtle survival and recovery (Patino-Martinez 2013). At nesting beaches, green sea turtles rely on intact dune structures, native vegetation, and normal beach temperatures for nesting (Ackerman 1997). Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). On the Pacific coast of Mexico in the mid-1970s, >70,000 green turtle eggs were harvested every night. Hundreds of mostly immature green sea turtles were killed between 2006 and 2008 due to bycatch and direct harvest along Baja California Sur (Senko et al. 2014). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991b). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999; Lee Long et al. 2000; Waycott et al. 2005). Ingestion of plastic and other marine debris is another source of morbidity and mortality (Stamper et al. 2009). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n = 34), although mortality appears to have resulted in three cases (Tourinho et al. 2009). Low-level bycatch has also been documented in longline fisheries (Petersen et al. 2009). Further, the introduction of alien algae species threatens the stability of some coastal ecosystems and may lead to the elimination of preferred dietary species of green sea turtles (De Weede 1996). Very few green sea turtles are bycaught in U.S. fisheries (Finkbeiner et al. 2011). However, a legal fishery operates in Madagascar that harvested about 10,000 green turtles annually in the mid-1990s. Green sea turtles are killed because they are seen as competitors for fishery resources in parts of India (Arthur et al. 2013). Between 1991 and 2011, an average of 8,169 green sea turtles were harvested annually along the Caribbean coast of Nicaragua (over 171,000 over this period); a rate that has been in decline potentially due to population depletion (Lagueux et al. 2014).

Sea level rise may have significant impacts upon green turtle nesting on Pacific atolls. These low-lying, isolated locations could be inundated by rising water levels associated with global warming, eliminating nesting habitat (Baker et al. 2006b; Fuentes et al. 2010). Fuentes et al. (2010) predicted that rising temperatures would be a much greater threat in the long term to the hatching success of sea turtle turtles in general and green sea turtles along northeastern Australia particularly. Green sea turtles emerging from nests at cooler temperatures likely absorb more yolk that is converted to body tissue than do hatchlings from warmer nests (Ischer et al. 2009). Predicted temperature rises may approach or exceed the upper thermal tolerance limit of sea turtle incubation, causing widespread failure of nests (Fuentes et al. 2010). Although the timing of loggerhead nesting depends upon sea-surface temperature, green sea turtles do not appear to be affected (Pike 2009).

Green sea turtles have been found to contain the organochlorines chlordane, lindane, endrin, endosulfan, dieldrin, DDT and PCB (Gardner et al. 2003; Miao et al. 2001). Levels of PCBs found in eggs are considered far higher than what is fit for human consumption (Van de Merwe et al. 2009). The heavy metals copper, lead, manganese, cadmium, and nickel have also been found in various tissues and life stages (Barbieri 2009). Arsenic also occurs in very high levels in green sea turtle eggs (Van de Merwe et al. 2009). These contaminants have the potential to cause deficiencies in endocrine, developmental, and reproductive health, and depress immune function in loggerhead sea turtles (Keller et al. 2006a; Storelli et al. 2007b). Exposure to sewage effluent may also result in green sea turtle eggs harboring antibiotic-resistant strains of bacteria (Al-Bahry et al. 2009). DDE has not been found to influence sex determination at levels below cytotoxicity (Keller and McClellan-Green 2004; Podreka et al. 1998). To date, no tie has been found between pesticide concentration and susceptibility to fibropapillomatosis, although degraded habitat and pollution have been tied to the incidence of the disease (Aguirre et al. 1994; Foley et al. 2005). Flame retardants have been measured from healthy individuals (Hermanussen et al. 2008). It has been theorized that exposure to tumor-promoting compounds produced by the cyanobacteria *Lyngbya majuscule* could promote the development of fibropapillomatosis (Arthur et al. 2008). It has also been theorized that dinoflagellates of the genus *Prorocentrum* that produce the tumorogenic compound okadoic acid may influence the development of fibropapillomatosis (Landsberg et al. 1999).

The primary, human-related threats to green turtles in Guam and the CNMI include direct harvesting of sea turtles and eggs as well as habitat loss due to rapidly expanding tourism, including increased coastal development on nesting beaches (NMFS and USFWS 1998d). The impacts of vessel strikes in the West Pacific are unknown, but not thought to be of great consequence, except possibly in Palau where high speed skiffs constantly travel throughout the lagoon south of the main islands (NMFS and USFWS 1998d). However, green turtles have been documented as occasionally being hit by boats in Guam and CNMI (e.g., unpublished data from Tammy Mae Summers, personal communication to NMFS, April 7, 2015).

4.2.6.12 Green Sea Turtle DPSs

On April 6, 2016 NMFS published a final rule to list 11 DPSs of green sea turtles as threatened or endangered under the ESA (Figure 16; 81 FR 20057). The green sea turtles that are expected to occur in the action area are from the Central West Pacific, East Indian-West Pacific, and Central North Pacific DPSs (see Section 4.2.6.16 for additional detail). The majority of the action area overlaps with the DPS delineation of the Central West Pacific DPS, but a portion of the action area overlaps with the DPS delineation of the East Indian-West Pacific DPS.

It’s worth noting that green sea turtle DPSs were delineated according to nesting beaches. The DPS delineation map below (Figure 16) does not represent the oceanic range of turtles from each DPS. For example, the Hawaii-based pelagic, deep-set longline fishery operates inside and outside the EEZ primarily around the main Hawaiian Islands and Northwestern Hawaiian Islands, with some trips to the EEZs around the remote U.S. Pacific Islands of Johnson Atoll, Kingman Reef, Palmyra, Jarvis, Howland, Baker, Midway, and Wake Islands. The NMFS Southwest Fisheries Science Center conducted a genetic analysis on 13 green sea turtles caught in the fishery and found that turtles caught in the fishery could be attributed to nesting populations from multiple DPSs including the East Pacific, Central North Pacific, East Indian-West Pacific and Southwest Pacific DPSs (NMFS 2016). This indicates the oceanic range of green sea turtle DPSs extends well beyond the DPS delineations presented in Figure 16.

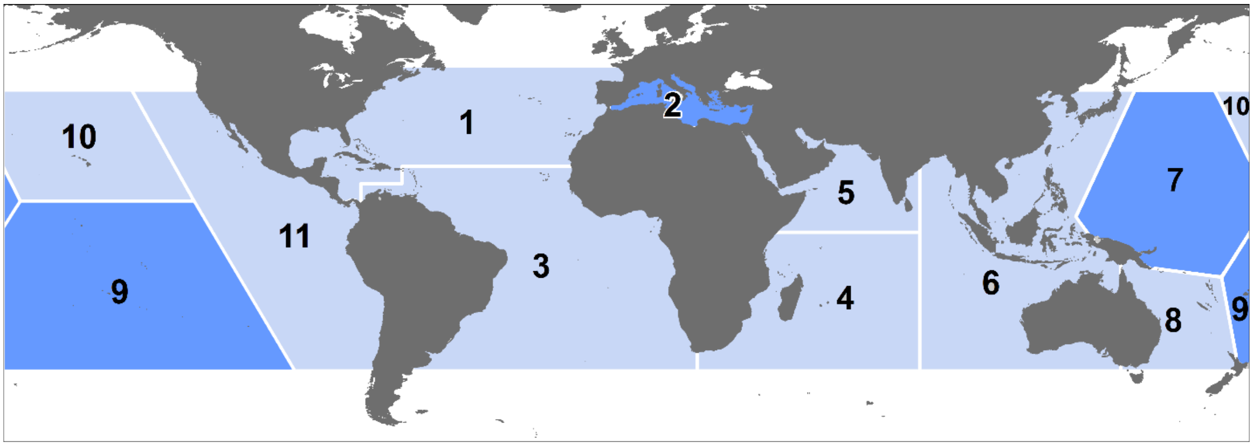

Figure 16. Threatened (light blue) and endangered (dark blue) green turtle DPSs: 1) North Atlantic, 2) Mediterranean, 3) South Atlantic, 4) Southwest Indian, 5) North Indian, 6) East Indian-West Pacific, 7) Central West Pacific, 8) Southwest Pacific, 9) Central South Pacific, 10) Central North Pacific, and 11) East Pacific.

4.2.6.13 Central West Pacific DPS

The nesting range of the Central West Pacific DPS encompasses the Republic of Palau (Palau), Federated States of Micronesia (FSM), New Guinea, Solomon Islands, Marshall Islands, Guam, the Commonwealth of the Northeren Mariana Islands (CNMI), and a portion of Japan. Green turtle nesting occurs at least at low levels throughout the geographic distribution of the population, with isolated locations having high nesting activity. Currently, there are approximately 51 nesting sites and 6,518 nesting females in the Central West Pacific (NMFS 2015c). There are a number of unquantified nesting sites, possibly with small numbers; however, specifics regarding these sites are unknown. The largest nesting site is in the FSM, and that particular site hosts approximately 22 percent of the total annual nesting females for this DPS. The highest numbers of females nesting in this DPS are located in Gielop and Iar Island, Ulithi Atoll, Yap, FSM (1,412); Chichijima (1,301) and Hahajima (394), Ogasawara, Japan; Bikar Atoll, Marshall Islands (300); and Merir Island, Palau (441) (Barr 2006; Maison et al. 2010; NMFS and USFWS 1998c; Resources 2005; Resources 2008). There are numerous other populations in the FSM, Solomon Islands, and Palau, Guam, and the CNMI. Historical baseline nesting information in general is not widely available in this region, but exploitation and trade of green sea turtles throughout the region is well-known.

Green sea turtles departing nesting grounds in this DPS travel throughout the western Pacific Ocean. Results of three post-nesting green sea turtles from Palau in 2006 showed they remained nearby or traveled to the Aru Islands in Indonesia – roughly 1,100 km away (NMFS 2015c). Five postnesting green sea turtles leaving Erikub Atoll in the Marshall Islands in 2007 traveled to the Philippines, Kiribati, FSM, or remained in the Marshallese EEZ (NMFS 2015c). Turtles tagged in Yap (FSM) were recaptured in the Philippines, Marshall Islands, Papua New Guinea, Palau, and Yap (Cruce 2009; Resources 2008). A turtle tagged on Gielop Island, Yap in 1991 was recaptured in Muroto Kochi prefecture, Japan in 1999 (Miyawaki et al. 2000). A nesting female tagged on Merir Island, Palau was captured near the village of Yomitan Okinawa, Japan (Resources 2008). Hundreds of nesting females tagged in Ogasawara Island were recaptured in the main islands of Japan, the Ryukyu Archipelago (Okinawa), Taiwan, China, and Philippines (NMFS 2015c). A turtle tagged in Japan was recorded nesting in Yap, FSM (NMFS 2015c).

In addition to nesting beaches, green sea turtles occupy coastal waters in low to moderate densities at foraging areas throughout the DPS. Aerial sea turtle surveys show that an in-water population exists around Guam (NMFS 2015c). In-water green turtle density in the Marianas Archipelago is low and mostly restricted to juveniles (Kolinski et al. 2006; Palacios 2012; Pultz et al. 1999). In-water information in this DPS overall is particularly limited.

There is insufficient long-term and standardized monitoring information to describe abundance and population trends adequately for many areas of the Central West Pacific DPS. The limited available information suggests a nesting population decrease in some portions of the DPS like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the FSM (Maison et al. 2010). Martin et al. (2016) analyzed data from five decades of marine megafauna surveys around Guam and found that since the 1960s, sea turtle abundance increased by 7%, though this increase was largely restricted to one geographic area where optimal habitat coincides with low human density. The authors suggested that protections in the region may be working to recover turtle populations (Martin et al. 2016).

In addition to the natural and anthropogenic threats to all DPSs of green sea turtles, as described in sections 4.2.6.10 and 4.2.6.11, threats specific to the Central West Pacific DPS are discussed in the proposed rule to list 11 DPSs of green sea turtles under the ESA (80 FR 15271). Threats include, but are not limited to, direct harvest, incidental bycatch in fisheries, destruction and modification of nesting habitat, debris, activities associated with national defense, disease, predation, toxic compounds, and climate change.

4.2.6.14 East Indian-West Pacific DPS

Green turtle nesting is widely dispersed throughout the East Indian-West Pacific DPS, with important nesting sites occurring in Northern Australia, Indonesia, Malaysia (Sabah and Sarawak Turtle Islands), Peninsular Malaysia, and the Philippine Turtle Islands. The largest nesting site lies within Northern Australia, which supports approximately 25,000 nesting females, calculated from the 5,000 nesting female's order of magnitude (Limpus 2009). Currently, the East Indian-West Pacific DPS hosts 58 reported nesting sites (in some cases nesting sites are made up of multiple beaches based on nesting survey information) with six of these sites supporting more than 5,000 nesting females each (including the 25,000 nesters in Northern Australia). Seven sites have between 1,001 and 5,000 nesting females and eight sites have between 500 and 1,000 nesting females (80 FR 15271). This results in a total of at least 61,000 nesting females in this DPS. Green turtle populations within this DPS have experienced increases at some nesting sites and decreases at others. Nonetheless, populations are substantially depleted from historical levels.

The in-water range of the East Indian-West Pacific DPS is similarly widespread with shared foraging sites throughout the DPS. Tagged green sea turtles that nest in Western Australia have been resighted in Arnhem Land and as far north as the Java Sea near Indonesia (Baldwin et al. 2003; Limpus et al. 2007). The extensive coastline and islands of Indonesia support a large range of nesting and foraging habitat for green sea turtles (Halim and Dermawan 1999). Waayers and Fitzpatrick (2013) found that in the Kimberly region of Australia, the green turtle appears to have a broad migration distribution and numerous potential foraging areas. A satellite-tagged female green turtle at Redang, Malaysia, travelled near Koh Samui, Thailand (Liew 2002). Green turtle foraging grounds occur around the Andaman and Nicobar Islands (Andrews et al. 2006). Additionally, a green sea turtle from this DPS was incidentally caught in the Hawaii deep-set pelagic longline fishery which operates inside and outside the EEZ primarily around the main Hawaiian Islands and Northwestern Hawaiian Islands, with some trips to the EEZs around the remote U.S. Pacific Islands of Johnson Atoll, Kingman Reef, Palmyra, Jarvis, Howland, Baker, Midway, and Wake Islands (NMFS 2015a). The estimated total nester abundance for this DPS is approximately 77,009 (Seminoff et al. 2015).

In addition to the natural and anthropogenic threats to all DPSs of green sea turtles, as described in sections 4.2.6.10 and 4.2.6.11, threats specific to the East Indian-West Pacific DPS are discussed in the proposed rule (80 FR 15271). Threats include, but are not limited to, directed harvest, incidental bycatch in fisheries, destruction and modification of nesting habitat, debris, activities associated with national defense, disease, predation, toxic compounds, and climate change.

4.2.6.15 Central North Pacific DPS

The nesting range of the Central North Pacific DPS covers the Hawaiian Archipelago and Johnson Atoll. The principal nesting site for green turtles in the Central North Pacific DPS is FFS, where 96 percent of the population (3,710 of 3,846 nesting females) currently nests (Balazs 1980b). Current nesting by green turtles occurs in low numbers (3-36 nesting females at any one site) throughout the Northwest Hawaiian Islands (NWHI) at Laysan, Lisianski, Pearl and Hermes Reef, and very uncommonly at Midway. Since 2000, green turtle nesting on the MHI has been identified in low numbers (1-24) on seven islands (80 FR 15271). Green turtles in the Central North Pacific DPS bask on beaches throughout the NWHI and in the MHI.

Since nesting surveys were initiated in 1973, there has been a marked increase in annual green turtle nesting at East Island, FFS, where approximately 50 percent of the nesting on FFS occurs (Balazs and Chaloupka 2004; Balazs and Chaloupka 2006). During the first 5 years of monitoring (1973-1977), the mean annual nesting abundance was 83 females, and during the most recent 5 years of monitoring (2009-2012), the mean annual nesting abundance was 464 females (Balazs and Chaloupka 2006; G. Balazs unpublished data). This increase over the last 40 years corresponds to an annual increase of 4.8 percent. Information on in-water abundance trends is consistent with the increase in nesting (80 FR 15271). The number of immature green turtles residing in foraging areas of the eight MHI has increased (Balazs et al. 1996). In addition, although the causes are not totally clear, there has been a dramatic increase in the number of basking turtles in the Hawaiian Islands over the last 2 decades, both in the southern foraging areas of the main islands (Balazs et al. 1996) as well as at northern foraging areas at Midway Atoll (Balazs et al. 2005).

The majority of tagged green sea turtles from nesting beaches in this DPS have been recovered within the Hawaiian Archipelago. The three outliers involved a recovery in Japan, one in the Marshall Islands and one in the Philippines. Additionally, genetic analysis has indicated that approximately 3 percent of green sea turtles found foraging arounc CNMI are from French Frigate Shoals (Peter Dutton, NMFS, personal communication to Eric MacMillan June 1, 2016). This indicates at least some turtles from this DPS make more distant migrations outside of the Hawaiian Archipelago.

In addition to the natural and anthropogenic threats to all DPSs of green sea turtles, as described in sections 4.2.6.10 and 4.2.6.11, threats specific to the Central North Pacific DPS are discussed in the proposed rule (80 FR 15271). Threats include, but are not limited to, direct harvest, incidental bycatch in fisheries, destruction and modification of nesting habitat, disease, predation, and climate change.

4.2.6.16 Green sea turtle DPS occurrence in the MITT Study Area

We expect green sea turtles from all three of the DPSs discussed above to occur in the MITT study area. The majority of green sea turtles in the action area are expected to be from the Central West Pacific DPS (T. Todd Jones personal communication to Eric MacMillan on May 24, 2016; Peter Dutton, NMFS, personal communication to Eric MacMillan June 1, 2016). Most of the action area overlaps with the nesting range of this DPS, though according to the narrative and figure (Figure 16) in the proposed and final rule (80 FR 15271; 81 FR 20057), the western portion of the action area overlaps with the nesting range of the East Indian-West Pacific DPS. Additionally, the oceanic range of this DPS may extend further east into other portions of the action area where Navy training and testing activities will occur. The limited available genetic data has not indicated green sea turtles from this DPS forage in nearshore areas around CNMI. Available genetic data indicates that most green sea turtles foraging in nearshore areas around CNMI are from nesting beaches in the Republic of Marshall Islands and Yap (included in the Central West Pacific DPS) and a small percentage (~3%) are from nesting beaches on French Frigate Shoals (included in the Central North Pacific DPS) (Peter Dutton, personal communication to Eric MacMillan, June 1, 2016).

It’s worth noting that to date, genetic sampling of green sea turtles in the action area has only occurred in nearshore areas around CNMI. Genetic sampling in other portions of the action area (including nearshore areas around other islands and in non-neritic areas) has not yet occurred (T. Todd Jones personal communication to Eric MacMillan on May 24, 2016).

4.2.7 Hawksbill Sea 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.

4.2.7.1 Populations

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.

4.2.7.2 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 (NMFS 2013). 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 1997a). 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). 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.

4.2.7.3 Migration and Movement

Upon first entering the sea, neonatal hawksbills in the Caribbean are believed to enter an oceanic phase that may involve long distance travel and eventual recruitment to nearshore foraging habitat (Boulon Jr. 1994). In the marine environment, the oceanic phase of juveniles (i.e., the "lost years") remains one of the most poorly understood aspects of hawksbill life history, both in terms of where turtles occur and how long they remain oceanic. Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon et al. 2010). Adults along the Pacific coast of Central America exhibit highly restrictive, inshore ranges between foraging and nesting locations, most of which was mangrove estuaries (Gaos et al. 2012a). Subadult hawksbill sea turtles captured satellite tracked in the Dry Tortugas National Park showed high-degrees of site fidelity for extended periods, although all three eventually moved to other areas outside the park (Hart et al. 2012). The same trend was found for adults tracked after nesting in the Dominican Republic, with some remaining for extended periods in the nesting area and other migrating to Honduras and Nicaragua (Hawkes et al. 2012). Satellite tracking for these individuals showed repeated returns to the same Dominican and Central American areas (Hawkes et al. 2012). Hawksbills dispersing from nesting areas along Brazil moved along coastal areas until they reached foraging areas (Marcovaldi et al. 2012). Here, genetically-identified hawksbill-loggerhead hybrids dispersed more broadly than pure-bred hawksbills (Marcovaldi et al. 2012). Home ranges tend to be small (a few square kilometers)(Berube et al. 2012). Recent tagging data from off the coast of Tinian, indicated that one subadult hawksbill remained in the area in which it was tagged for 51 days following capture, and then traveled 286 km, eventually taking up residency in the deeper waters outside of Cocos Lagoon, Guam (Jones and Houtan 2014).

4.2.7.4 Habitat

Hawksbill sea turtles are highly migratory and use a wide range of broadly separated localities and habitats during their lifetimes (Musick and Limpus 1997b; Plotkin 2003). Small juvenile hawksbills (5 to 21 cm straight carapace length) have been found in association with *Sargassum* spp. in both the Atlantic and Pacific oceans (Musick and Limpus 1997b) and observations of newly hatched hawksbills attracted to floating weeds have been made (Hornell 1927; Mellgren and Mann 1996; Mellgren et al. 1994). Post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997b), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007c). Eastern Pacific adult females have recently been tracked in saltwater mangrove forests along El Salvador and Honduras, a habitat that this species was not previously known to occupy (Gaos et al. 2011). Individuals of multiple breeding locations can occupy the same foraging habitat (Bass 1999; Bowen et al. 1996; Bowen et al. 2007; Diaz-Fernandez et al. 1999; Velez-Zuazo et al. 2008). As larger juveniles, some individuals may associate with the same feeding locality for more than a decade, while others apparently migrate from one site to another (Blumenthal et al. 2009a; Mortimer et al. 2003;

Musick and Limpus 1997b). Larger individuals may prefer deeper habitats than their smaller counterparts (Blumenthal et al. 2009a). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

4.2.7.5 Growth and Reproduction

The best estimate of age at sexual maturity for hawksbill sea turtles is 20 to 40 years (Chaloupka and Limpus 1997; Crouse 1999). Reproductive females undertake periodic (usually non-annual) migrations to their natal beaches to nest. Movements of reproductive males are less well known, but are presumed to involve migrations to their nesting beach or to courtship stations along the migratory corridor (Meylan 1999). Females nest an average of 3 to 5 times per season (Meylan and Donnelly 1999; Richardson et al. 1999). Clutch size can be up to 250 eggs; larger than that of other sea turtles (Hirth 1980). Reproductive females may exhibit a high degree of fidelity to their nest sites.

The life history of hawksbills consists of a pelagic stage that lasts from hatching until they are approximately 22 to 25 cm in straight carapace length (Meylan 1988; Meylan and Donnelly 1999), followed by residency in coastal developmental habitats. Growth accelerates early on until turtles reach 65 to 70 cm in curved carapace length, after which it slows to negligible amounts after 80 cm (Bell and Pike 2012). As with other sea turtles, growth is variable and likely depends upon nutrition available (Bell and Pike 2012). Juvenile hawksbills along the British Virgin Islands grow at a relatively rapid rate of roughly 9.3 cm per year and gain 3.9 kg annually (Hawkes et al. 2014).

4.2.7.6 Feeding

Dietary data from oceanic stage hawksbills are limited, but indicate a combination of plant and animal material (Bjorndal 1997b). Sponges and octocorals are common prey off Honduras (Berube et al. 2012; Hart et al. 2013b).

4.2.7.7 Diving

Hawksbill diving ability varies with age and body size. As individuals increase with age, diving ability in terms of duration and depth increases (Blumenthal et al. 2009b). Studies of hawksbills in the Caribbean have found diurnal diving behavior, with dive duration nearly twice as long during nighttime (35 to 47 min) compared to daytime (19-26 min Blumenthal et al. 2009b; Van Dam and Diez 1997). Daytime dives averaged 5 m, while nighttime dives averaged 43 m (Blumenthal et al. 2009b). However, nocturnal differences were not observed in the eastern Pacific (Gaos et al. 2012b).

Hawksbills have long dive durations, although dive depths are not particularly deep. Adult females along St. Croix reportedly have average dive times of 56 min, with a maximum time of 73.5 min (Starbird et al. 1999). Average day and night dive times were 34 to 65 and 42 to 74 min, respectively. Immature individuals have much shorter dives of 8.6 to 14 min to a mean depth of 4.7 m while foraging (Van Dam and Diez 1997).

4.2.7.8 Vocalization and Hearing

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). Piniak et al. (2012) found hawksbill hatchlings capable of hearing underwater sounds at frequencies of between 50 and 1,600 Hz (maximum sensitivity at 200 to 400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a).

4.2.7.9 Status and Trends

Hawksbill turtles were listed as endangered throughout their range under the ESA in 1970. 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. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS 2007c). Worldwide, an estimated 21,212 to 28,138 hawksbills nest each year among 83 sites. Among the 58 sites with historic trends, all show a decline during the past 20 to 100 years.

American Samoa and Western Samoa host fewer than 30 females annually (Grant et al. 1997; Tuato'o-Bartley et al. 1993). In Guam, only 5 to 10 females are estimated to nest annually (NMFS and USFWS 2013) and the same is true for Hawaii, but there are indications that this population is increasing (G. Balazs, pers. comm. in NMFS and USFWS 2007c). Additional populations are known from the eastern Pacific (potentially extending from Mexico through Panama), northeastern Australia, and Malaysia (Hutchinson and Dutton 2007). El Salvador is now known to host the majority of hawksbill turtle nesting activity in the eastern Pacific, with 79.6 percent (n = 5430) of all nesting observation records, and Mexico hosting the majority of records of hawksbill turtles at sea, with 60.3 percent (n = 544) of all in-water observation records (Gaos et al. 2010). The most recent estimate of the total annual number of nesting females for the Central Pacific hawksbill population was 950 to 1,185 females annually, with an overall downward trend (NMFS and USFWS 2013).

In August 2013, a PIFSC researcher and his crew captured two sub-adult hawksbills over a four days survey period off the northwestern coast of Saipan and the western coast of Tinian (Jones and Houtan 2014). Between November 2013 and March 2014 the CNMI Department of Lands and Natural Resources captured three hawksbill sea turtles, two around Tinian and one around Saipan (Palacios et al. 2014). The Navy's NMSDD estimates 1.0734, 0.1342, and 0.000024 hawksbill sea turtles per km^2 in the nearshore FDM, nearshore other islands, and pelagic portions of the MITT action area, respectively (DoN 2014).

4.2.7.10 Natural Threats

Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can be lethal. The only other significant natural threat to hawksbill sea turtles is from hybridization of hawksbills with other species of sea turtles. This is especially problematic at certain sites where hawksbill numbers are particularly low (Mortimer and Donnelly in review). Predators (primarily of eggs and hatchlings) include dogs, pigs, rats, crabs, sea birds, reef fishes, groupers, feral cats, and foxes (Bell et al. 1994; Ficetola 2008). In some areas, nesting beaches can be almost completely destroyed and all nests can sustain some level of depredation (Ficetola 2008). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in excess of 90 percent of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

4.2.7.11 Anthropogenic Threats

Threats to hawksbill sea turtles are largely anthropogenic, both historically and currently. Impacts to nesting beaches include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). Because hawksbills prefer to nest under vegetation (Horrocks and Scott 1991; Mortimer 1982), they are particularly impacted by beachfront development and clearing of dune vegetation (Mortimer and Donnelly in review). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991b).

One of the most detrimental human threats to hawksbill sea turtles is the intensive harvest of eggs from nesting beaches. Between 1950 and 1992, approximately 1.3 million hawksbill shells were collected to supply tortoiseshell to the Japanese market, the world's largest. 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. Japan stopped importing tortoiseshell in 1993 in order to comply with CITES (Limpus and Miller 2008). Until recently, tens of thousands of hawksbills were captured and killed each year to meet demand for jewelry, ornamentation, and whole stuffed turtles (Eckert 1993c). 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 (Eckert 1993c). Large numbers of nesting and foraging hawksbill sea turtles are captured and killed for trade in Micronesia, the Mexican Pacific coast, southeast Asia and Indonesia (NMFS and USFWS 1998e).

In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999; Lee Long et al. 2000; Waycott et al. 2005). Hawksbills are typically associated with coral reefs, which are among the world's most endangered marine ecosystems (Wilkinson 2000). Although primarily spongivorous, bycatch of hawksbill sea turtles in the swordfish fishery off South Africa occurs (Petersen et al. 2009). Finkbeiner et al. (2011) estimated that annual bycatch interactions total at least 20 individuals annually for U.S. Atlantic fisheries (resulting in less than ten mortalities) and no or very few interactions in U.S. Pacific fisheries.

Future impacts from climate change and global warming may result in significant changes in hatchling sex ratios. The fact that hawksbill turtles exhibit temperature-dependent sex determination (Wibbels 2003) suggests that there may be a skewing of future hawksbill cohorts toward strong female bias (since warmer temperatures produce more female embryos).

4.2.7.12 Critical Habitat

On September 2, 1998, the 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. No critical habitat is designated within the MITT action area for this species.

4.2.8 Leatherback Sea 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 1971b) 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.

4.2.8.1 Populations

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.

Leatherbacks are found from tropical waters north to Alaska within the North Pacific and is the most common sea turtle in the eastern Pacific north of Mexico (Eckert 1993a; Stinson 1984c; Wing and Hodge 2002). The west coast of Central America and Mexico hosts nesting from September-March, although Costa Rican nesting peaks during April-May (Chacón-Chaverri and Eckert 2007; LGL Ltd. 2007). Leatherback sea turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila et al. 2000). In Fiji, Thailand, and Australia, leatherback sea turtles have only been known to nest in low densities and scattered sites. Leatherback nesting aggregations occur widely in the Pacific, including China, Malaysia, Papua New Guinea, Indonesia, Thailand, Australia, Fiji, the Solomon Islands, and Central America (Dutton et al. 2007; Limpus 2002). Significant nesting also occurs along the Central American coast (Márquez 1990). Although not generally known to nest on Japanese shores, two nests were identified in the central Ryukyu Islands in 2002 (Kamezaki et al. 2002a).

Nesting beaches also occur in Mexico and Costa Rica (nesting occurs October through March) are a separate population from the western Pacific beaches (Benson et al. 2007a; summary in NMFS and USFWS 2007d; Spotila 2004a). In Costa Rica, leatherbacks nest at Playa Naranjo in Santa Rosa National Park, the second-most important nesting beach on the Pacific coast (Yañez et al. 2010), Rio Oro on the Osa Peninsula, and at various beaches in Las Baulas National Park, which includes Playa Langosta and Playa Grande and contains the largest colony of leatherbacks in the Pacific (Spotila 2004a). Females typically lay six clutches per season (average nine days between nests), which incubate for 58–65 days (Lux et al. 2003). Limited nesting also occurs along Nicaragua, Panama, El Salvador, Vanuatu, and Guatemala.

In the Pacific Ocean, genetic studies have identified three distinct populations (referred to also as genetic stocks or Management Units; see (Wallace et al. 2010a)) of leatherback turtles: (1) Mexico and Costa Rica, which are genetically homogenous but distinct from the western populations; (2) Papua Barat in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu, which comprise a metapopulation representing a single genetic stock; and (3) Malaysia (Barragan and Dutton 2000; Dutton et al. 1999; Dutton 2005-2006; Dutton et al. 2000a; Dutton et al. 2006; Dutton 2007). The genetically distinct Malaysia nesting population likely is extirpated (Chan and Liew 1996b; Dutton et al. 1999; Dutton 2005-2006).

4.2.8.2 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 1999b; Morreale et al. 1994b). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert 1998b). 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. 2007b). Leatherbacks from both eastern and western Pacific Ocean nesting populations migrate to northern Pacific Ocean foraging grounds, where longline fisheries operate (Dutton et al. 1998b). 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. 2007b). 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.

4.2.8.3 Migration and Movement

Leatherback sea turtles migrate throughout open ocean convergence zones and upwelling areas, along continental margins, and in archipelagic waters (Eckert 1998a; Eckert 1999a; Morreale et al. 1994a). In a single year, a leatherback may swim more than 9,600 km to nesting and foraging areas throughout ocean basins (Benson et al. 2007a; Benson et al. 2007c; Eckert 1998a; Eckert 2006; Eckert et al. 2006; Ferraroli et al. 2004; Hays et al. 2004; Sale et al. 2006). Much of this travel may be due to movements within current and eddy features, moving individuals along (Sale and Luschi 2009). Return to nesting beaches may be accomplished by a form of geomagnetic navigation and use of local cues (Sale and Luschi 2009). Leatherback females will either remain in nearshore waters between nesting events, or range widely, presumably to feed on available prey (Byrne et al. 2009; Fossette et al. 2009a).

Fossette et al. (2009b) identified three main migratory strategies in leatherbacks in the North Atlantic (almost all of studied individuals were female). One involved 12 individuals traveling to northern latitudes during summer/fall and returning to waters during winter and spring. Another strategy used by six individuals was similar to this, but instead of a southward movement in fall, individuals overwintered in northern latitudes (30–40º N, 25–30º W) and moved into the Irish Sea or Bay of Biscay during spring before moving south to between 5 and 10º in winter, where they remained or returned to the northwest Atlantic. A third strategy, which was followed by three females remaining in tropical waters for the first year subsequent to nesting and moving to northern latitudes during summer/fall and spending winter and spring in latitudes of 40–50º N. Individuals nesting in Caribbean Islands migrate to foraging areas off Canada (Richardson et al. 2012).

Females tracked from nesting beaches in Brazil stayed in waters off Brazil, Uruguay, and Argentina (Almeida et al. 2011). Adult and subadult leatherbacks caught in fisheries operating in southern waters off Uruguay (Fossette et al. 2010a; Lopez-Mendilaharsu et al. 2009) and Brazil (Almeida et al. 2011) remained in the southwestern Atlantic Ocean.

Genetic studies support the satellite telemetry data indicating a strong difference in migration and foraging fidelity between the breeding populations in the northern and southern hemispheres of the Atlantic Ocean (Dutton et al. 2013; Stewart et al. 2013). Genetic analysis of rookeries in Gabon and Ghana confirm that leatherbacks from West African rookeries migrate to foraging areas off South America (Dutton et al. 2013). Foraging adults off Nova Scotia, Canada, mainly originate from Trinidad and none are from Brazil, Gabon, Ghana, or South Africa (Stewart et al. 2013).

Leatherbacks occur along the southeastern U.S. year-round, with peak abundance in summer (TEWG 2007). In spring, leatherback sea turtles appear to be concentrated near the coast, while other times of the year they are spread out at least to the Gulf Stream. From August 2009 through August 2010 off Jacksonville, Florida, surveys sighted 48 leatherback sea turtles, while simultaneous vessel surveys sighted four leatherback sea turtles (U.S. Department of the Navy 2010).

Leatherback sea turtles feed, rest, and migrate regularly in the northern Gulf of Mexico, inhabiting deep offshore waters in the vicinity of DeSoto Canyon (Davis et al. 2000a; Landry and Costa 1999). Leatherback sea turtles feed in shallow waters on the continental shelf waters along the Florida Panhandle, the Mississippi River Delta, and the Texas coast on dense aggregations of (Collard 1990).

Satellite tracking data reveal that leatherback females leaving Mexican and Central American nesting beaches migrate towards the equator and into Southern Hemisphere waters, some passing the Galápagos Islands, and disperse south of 10º S (Dutton et al. 2006; Shillinger et al. 2010a). However, observations of leatherbacks in the Galápagos Islands are rare (Zárate et al. 2010).

Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon et al. 2010). Individuals nesting in Malaysia undergo migrations to tropical feeding areas, taking 5–7 months to arrive there from nesting locations (Benson et al. 2011b). Additional foraging occurs in temperate locations, including across the Pacific basin along the U.S. west coast; individuals take 10–12 months to migrate here (Benson et al. 2011b). Individuals nesting during the boreal summer move to feeding areas in the North China Sea, while boreal winter nesters moved across the Equator to forage in the Southern Hemisphere (Benson et al. 2011b).

4.2.8.4 Habitat

Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Grant and Ferrell 1993; Schroeder and Thompson 1987; Shoop and Kenney 1992; Starbird et al. 1993). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011b; Collard 1990; Davenport and Balazs 1991; Frazier 2001; HDLNR (Hawai'i Department of Land and Natural Resources) 2002). Cool, shallow, productive waters are areas where leatherbacks concentrate during late fall, winter, and early spring, where their dives become shallower and shorter, presumeably associated with foraging opportunities (Dodge et al. 2014). Aerial surveys off the western U.S. support continental slope waters as having greater leatherback occurrence than shelf waters (Bowlby et al. 1994; Carretta and Forney 1993; Green et al. 1992; Green et al. 1993). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Areas above 30º N in the Atlantic appear to be popular foraging locations (Fossette et al. 2009b). Northern foraging areas were proposed for waters between 35º and 50º N along North American, Nova Scotia, the Gulf of Saint-Laurent, in the western and northern Gulf Stream, the Northeast Atlantic, the Azores front and northeast of the Azores Islands, north of the Canary Islands. Southern foraging was proposed to occur between 5º and 15º N in the Mauritania upwelling, south of the Cape Verde islands, over the Guinea Dome area, and off Venezuela, Guyana and Suriname.

For the western Pacific population, seven ecoregions (South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension, equatorial Eastern Pacific, and California Current Extension) were identified as important seasonal foraging areas (Benson et al. 2011a). Off the U.S. west coast, two areas were identified as essential (“critical”) habitat for leatherbacks in 2012. One includes the nearshore waters between Cape Flattery, Washington, and Cape Blanco, Oregon extending offshore to the 2000 meter isobaths. This area was identified as the principal Oregon/Washington foraging area and included important habitat associated with the Columbia River Plume, and Heceta Bank, Oregon. Here, great densities of primary prey species, brown sea nettle, occur seasonally north of Cape Blanco (Brodeur et al. 2005; Reese 2005; Shenker 1984). The second area identified as “critical habitat” includes offshore waters between the 200 and 3000 meter isobaths from Point Arena to Point Sur, California and waters between the coastline and the 3000 meter isobath from Point Sur to Point Arguello, California.

In the eastern Pacific Ocean, post-nesting females from Playa Grande, Costa Rica, commonly forage offshore in the South Pacific Gyre in upwelling areas of cooler, deeper water and high productivity (Shillinger et al. 2011). During the nesting season, they stay within the shallow, highly productive, continental shelf waters (Shillinger et al. 2010b).

4.2.8.5 Growth and Reproduction

It has been thought that leatherbacks reach sexual maturity somewhat faster than other sea turtles (except Kemp’s ridley), with an estimated range of 3–6 years (Rhodin 1985) to 13–14 years (Zug and Parham 1996). However, recent research suggests otherwise, with western North Atlantic leatherbacks possibly not maturing until as late as 29 years of age (Avens and Goshe 2007; Avens and Goshe 2008; Avens et al. 2009). Female leatherbacks nest frequently (up to 13, average of 5–7 nests per year and about every 2-3 years)(Eckert et al. 2012). The average number of eggs per clutch varies by region: Atlantic Ocean (85 eggs), western Pacific Ocean (85 eggs), eastern Pacific Ocean (65 eggs) and Indian Ocean (>100 eggs (Eckert et al. 2012). However, up to ~30 percent of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55–75 days before hatching.

4.2.8.6 Sex Ratio

A significant female bias exists in all leatherback populations thus far studied. An examination of strandings and in-water sighting data from the U.S. Atlantic and Gulf of Mexico coasts indicates that 60 percent of individuals were female. Studies of Suriname nesting beach temperatures suggest a female bias in hatchlings, with estimated percentages of females hatched over the course of each season at 75.4, 65.8, and 92.2 percent in 1985, 1986, and 1987, respectively (Plotkin 1995). Binckley et al. (1998) found a heavy female bias upon examining hatchling gonad histology on the Pacific coast of Costa Rica, and estimated male to female ratios over three seasons of 0:100, 6.5:93.5, and 25.7:74.3. James et al. (2007) also found a heavy female bias (1.86:1) as well as a primarily large sub-adult and adult size distribution. Leatherback sex determination is affected by nest temperature, with higher temperatures producing a greater proportion of females (Mrosovsky 1994; Witzell et al. 2005).

4.2.8.7 Feeding

Leatherbacks may forage in high-invertebrate prey density areas formed by favorable features (Eckert 2006; Ferraroli et al. 2004). Although leatherbacks forage in coastal waters, they appear to remain primarily pelagic through all life stages (Heppell et al. 2003). The location and abundance of prey, including medusae, siphonophores, and salpae, in temperate and boreal latitudes likely has a strong influence on leatherback distribution in these areas (Plotkin 1995). Leatherback prey are frequently found in the deep-scattering layer in the Gulf of Alaska (Hodge and Wing 2000). North Pacific foraging grounds contain individuals from both eastern and western Pacific rookeries, although leatherbacks from the eastern Pacific generally forage in the Southern Hemisphere along Peru and Chile (Dutton 2005-2006; Dutton et al. 2000b; Dutton et al. 1998a). Mean primary productivity in all foraging areas of western Atlantic females is 150 percent greater than in eastern Pacific waters, likely resulting in twice the reproductive output of eastern Pacific females (Saba et al. 2007). Leatherbacks have been observed feeding on jellyfish in waters off Washington State and Oregon (Eisenberg and Frazier 1983; Stinson 1984c).

4.2.8.8 Diving

Leatherbacks are champion deep divers among sea turtles with a maximum recorded dive of over 4,000 m (Eckert et al. 1989; López-Mendilaharsu et al. 2009). Dives are typically 50 to 84 m and 75 to 90 percent of time duration is above 80 m (Standora et al. 1984). Leatherbacks off South Africa were found to spend <1 percent of their dive time at depths greater than 200 m (Hays et al. 2009). Dive durations are impressive, topping 86 min, but routinely 1 to 14 min (Eckert et al. 1989; Eckert et al. 1996; Harvey et al. 2006; López-Mendilaharsu et al. 2009). Most of this time is spent traveling to and from maximum depths (Eckert et al. 1989). Dives are continual, with only short stays at the surface (Eckert et al. 1989; Eckert et al. 1986; Southwood et al. 1999). Off Playa Grande, Costa Rica, adult females spent 57–68 percent of their time underwater, diving to a mean depth of 19 m for 7.4 min (Southwood et al. 1999). Off St. Croix, adult females dove to a mean depth of 61.6 m for an average of 9.9 min, and spent an average of 4.9 min at the surface

(Eckert et al. 1989). During shallow dives in the South China Sea, dives averaged 6.9 to 14.5 min, with a maximum of 42 min (Eckert et al. 1996). Off central California, leatherbacks dove to 20 to 30 m with a maximum of 92 m (Harvey et al. 2006). This corresponded to the vertical distribution if their prey (Harvey et al. 2006). Leatherback prey in the Gulf of Alaska are frequently concentrated in the deep-scattering layer (Hodge and Wing 2000). Mean dive and surface durations were 2.9 and 2.2 min, respectively (Harvey et al. 2006). In a study comparing diving patterns during foraging versus travelling, leatherbacks dove shallower (mean of 53.6 m) and moved more slowly (17.2 km/day) while in foraging areas while travelling to or from these areas (81.8 m and 51.0 km/day) (Fossette et al. 2009b).

4.2.8.9 Vocalization and Hearing

Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999c; Lenhardt 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Piniak et al. (2012) found leatherback hatchlings capable of hearing underwater sounds at frequencies of 50 to 1,200 Hz (maximum sensitivity at 100 to 400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a).

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles respond best to sounds between 200 and 700 Hz, with slow declines below 100 Hz and rapid declines above 700 Hz, and almost no sensitivity above 3 kHz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1 kHz and almost no responses beyond 3 or 4 kHz (Patterson 1966).

4.2.8.10 Status and Trends

Leatherback sea turtles received protection on June 2, 1970 (35 FR 8491) under the Endangered Species Conservation Act and, since 1973, have been listed as endangered under the ESA, but declines in nesting have continued worldwide. Consideration of the status of populations outside of the action area is important under the present analysis to determine the risk to the affected population(s) bears on the status of the species as a whole. Breeding females were initially estimated at 29,000 to 40,000, but were later refined to approximately 115,000 (Pritchard 1971a; Pritchard 1982). Spotila et al. (1996) estimated 34,500 females, but later issued an update of 35,860 (Spotila 2004b). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001b; NMFS 2001a).

Reliable estimates of survival or mortality at different life history stages are not easily obtained. The annual survival rate for leatherbacks that nested at Playa Grande, Costa Rica, was estimated to be 0.654 for 1993 and 1994 and 0.65 for those that nested in 1994 and 1995 (Spotila et al. 2000). Rivalan et al. (2005) estimated the mean annual survival rate of adult leatherbacks in French Guiana to be 0.91. Pilcher and Chaloupka (2013) used capture-mark-recapture data for 178 nesting leatherbacks tagged at Lababia beach, Kamiali, on the Huon Coast of Papua New Guinea over a 10-year austral summer nesting period (2000-2009). Annual survival probability (ca. 0.85) was constant over the 10-year period. Annual survival was lower than those estimated for Atlantic rookeries (Dutton et al. 2005; Rivalan et al. 2005). For the St. Croix, U.S. Virgin Islands population, the annual survival rate was approximately 0.893 (confidence interval = 0.87 to 0.92) for adult female leatherbacks at St. Croix (Dutton et al. 2005). Annual juvenile survival rate for St. Croix was estimated to be approximately 0.63, and the total survival rate from hatchling to first year of reproduction for a female hatchling was estimated to be between 0.004 and 0.02, given assumed age at first reproduction between 9 and 13 (Eguchi *et al*. 2006). In Florida, annual survival for nesting females was estimated to be 0.956 (Stewart 2007). Spotila et al. (1996) estimated the first year (from hatching) of survival for the global population to be 0.0625.

Heavy declines have occurred at all major Pacific basin rookeries, as well as Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. This includes a nesting decline of 23 percent between 1984 to 1996 at Mexiquillo, Michoacán, Mexico (Sarti et al. 1996). According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. 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 USFWS 2013), but is now estimated to number 3,172 total nests (NMFS USFWS 2013). Leatherback turtles have experienced major declines at all major Pacific basin rookeries. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998 to 1999 and 1999 to 2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila et al. (2000) estimated that the colony could fall to less than 50 females by 2003 to 2004. Fewer than 1,000 females nested on the Pacific coast of Mexico from 1995 to 1996 and fewer than 700 females are estimated for Central America (Spotila et al. 2000). The number of leatherback turtles nesting in Las Baulas National Park declined rapidly during the 1990s, from about 1,500 females during the 1988 to 1989 nesting season, to about 800 in 1990 to 1991 and 1991 to 1992 to 193 in 1993 to 1994 (Williams et al. 1996) and 117 in 1998 to 1999 (Spotila et al. 2000). Spotila (2004a) reported that between 59 and 435 leatherbacks nest at Las Baulas each year depending on the El Niño–La Niña cycle.

There are 28 known nesting sites for the western Pacific Ocean stock, with 5,000 to 9,100 leatherback nests laid annually across the western tropical Pacific Ocean, from Australia and Melanesia (Papua New Guinea, Solomon Islands, Fiji, and Vanuatu) to Indonesia, Thailand, and China (Chaloupka et al. 2004a; Dutton 2006; Hirth et al. 1993; Hitipeuw et al. 2007; Suarez et al. 2000). The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the northern Vogelkop coast of Irian Jaya (West Papua), Indonesia, with roughly 3,000 nests recorded annually (Dutton et al. 2007; Putrawidjaja 2000; Suárez et al. 2000). The Western Pacific leatherback metapopulation harbors the last remaining nesting aggregation of significant size in the Pacific with approximately 2700 to 4500 breeding females (Dutton et al. 2007; Hitipeuw et al. 2007). The total number of nests per year for the Jamursba-Medi leatherback nesting population ranged between a high of 6,373 nests in 1996 and a low of 1,537 nests in 2010 (Hitipeuw et al. 2007) and 1,596 in 2011 (Tapilatu et al. 2013). Nesting at Terengganu, Malaysia is 1 percent of that in 1950s (Chan and Liew 1996a). The South China Sea and East Pacific nesting colonies have undergone catastrophic collapse. Overall, Pacific populations have declined from an estimated 81,000 individuals to < 3,000 total adults and subadults (Spotila et al. 2000). The number of nesting leatherbacks has declined by an estimated 95 percent over the past 20 years in the Pacific (Gilman 2009). Drastic overharvesting of eggs and mortality from fishing activities is likely responsible for this tremendous decline (Eckert 1997; Sarti et al. 1996). The most recent overall estimate for Papua Barat, Indonesia, Papua New Guinea, and Solomon Islands is 5,000 to 10,000 nests per year (Nel 2012).

Of the three sea turtle species that have been sighted around Guam and the CNMI during marine surveys, the leatherback turtle is the least common (DON 2003b). This species is occasionally encountered in the deep, pelagic waters of the Marianas archipelago, although only a few occurrence records exist (Eckert et al. 1999). Recent National Oceanic and Atmospheric satellite tracking of leatherback turtles indicates sea turtles departing from regional nesting habitats and transiting through MITT waters (Benson et al. 2007a; Benson et al. 2011a). As for nearshore waters, Eldredge (2003)noted a rescue in 1978 of a 249 lb (112.9 kg) leatherback from waters southeast of Cocos Island, Guam. From 1987 to 1989, divers reported seeing leatherbacks in the waters off Harmon Point, Rota; however, none have been seen in the area in recent times (DoN 2010c). Leatherbacks do not nest at any of the islands in Micronesia. As a result, the occurrence of leatherback turtles would be considered rare throughout the year in nearshore waters of the action area. Since leatherback occurrences in the waters off Guam and the CNMI would most likely involve individuals in transit, occurrence is not expected in coastal (i.e., shelf) waters around any of the islands in the action area. The Navy's NMSDD estimates 0.00022 leatherback sea turtles per km^2 in the MITT action area (DoN 2014).

4.2.8.11 Natural Threats

Sea turtles face predation primarily by sharks and to a lesser extent by killer whales (Pitman and Dutton 2004). Hatchlings are preyed upon by herons, gulls, dogfish, and sharks. Leatherback hatching success is particularly sensitive to nesting site selection, as nests that are overwashed have significantly lower hatching success and leatherbacks nest closer to the high-tide line than other sea turtle species (Caut et al. 2009a). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in excess of 90 percent of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014). Lost timber on beaches and beachcast logs acan trap adult females nesting on Gabon beaches, resulting in mortality (Ikaran 2013).

4.2.8.12 Anthropogenic Threats

Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann et al. 2006; Hernandez et al. 2007; Maison 2006; Santidrián Tomillo et al. 2007). Structural impacts to beaches include building and piling construction, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Bourgeois et al. 2009; Chacón Chaverri 1999; Formia et al. 2003; Laurance et al. 2008). Lights on or adjacent to nesting beaches alter nesting adult behavior and is often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Bourgeois et al. 2009; Cowan et al. 2002; Deem et al. 2007; Witherington 1992; Witherington and Bjorndal 1991b). Leatherbacks are much more likely to emerge and not nest on developed beaches and much more likely to emerge and nest on undeveloped stretches (Roe et al. 2013). Plastic ingestion is very common in leatherbacks and can block gastrointestinal tracts leading to death (Mrosovsky et al. 2009). Along the coast of Peru, 13 percent of 140 leatherback carcasses were found to contain plastic bags and film (Fritts 1982). A leatherback found stranded along the northern Adriatic had been weakened by plastic ingestion, likely leading to an infection that ultimately killed the individual (Poppi et al. 2012). Although global warming may expand foraging habitats into higher latitude waters, increasing temperatures may increase feminization of nests (Hawkes et al. 2007b; James et al. 2006; McMahon and Hays 2006a; Mrosovsky et al. 1984). Rising sea levels may also inundate nests on some beaches. Egg collection is widespread and attributed to catastrophic declines, such as in Malaysia. Harvest of females along nesting beaches is of concern worldwide.

Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale et al. 2008; Fossette et al. 2009a; Gless et al. 2008; Petersen et al. 2009). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010c); many of these turtles are expected to be leatherbacks.

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. He estimates that this represented about a 23 percent mortality rate (or 33 percent if most mortality was focused on the East Pacific population). In the Pacific Ocean, between 1,000 and 1,300 leatherback sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004). Shallow-set longline fisheries based out of Hawaii are estimated to have captured and killed several hundred leatherback sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about 1 or 2 leatherback sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii are estimated to have captured about 19 leatherback sea turtles, killing about 5 of these sea turtles.

Donoso and Dutton (2010) found that 284 leatherbacks were bycaught between 2001 and 2005 as part of the Chilean longline fishery, with two individuals observed dead; leatherbacks were the most frequently bycaught sea turtle species. Between 8 and 17 leatherback turtles likely died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and, before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year. Currently, the U.S. tuna and swordfish longline fisheries managed under the Highly Migratory Species Fishery Management Plan (HMS FMP) are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011). All leatherbacks were released alive, with all gear removed for the majority of captures. While 2010 total estimates are not yet available, in 2009, 285.8 (95 percent CI: 209.6 to 389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). Lewison et al. (2004) estimated that 30,000 to 60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries, as well as others). Use of circle versus traditional J hooks can severely curtail bycatch (Santos et al. 2012) and new regulations are being developed and implemented in several countries around the world for their use. Finkbeiner et al. (2011) estimated that annual bycatch interactions total 1,400 individuals annually for U.S. Atlantic fisheries (resulting in roughly 40 mortalities) and one hundred interactions in U.S. Pacific fisheries (resulting in about ten mortalities). Garrison and Stokes estimated 597 interactions between leatherbacks and the U.S. Atlantic longline fishery in 2012, mostly with hooks embedded (Garrison and Stokes 2013). Mortality of leatherbacks in the U.S. shrimp fishery is now estimated at 54 turtles per year. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54 to 92 percent. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though 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 (NMFS 2001b).

Egg collection occurs in many countries around the world (e.g., (Billes and Fretey 2004; Brautigam and Eckert 2006; Fretey et al. 2007; Hilterman and Goverse 2007; Kinan 2002; Maison et al. 2010; Santidrián Tomillo et al. 2007; Tomillo et al. 2008). For example, during the 2012 nesting season, 55% (283 of 514) of leatherback nests were poached on Pacuare Playa, Costa Rica (Fonseca and Chacon 2012). Egg harvest has been attributed to catastrophic declines such as in Malaysia. Despite conservation efforts, egg harvest continues at certain levels in Indonesia, Papua New Guinea, Solomon Islands and Vanuatu (Committee 2008). We know little about the effects of contaminants on leatherback sea turtles. The metals arsenic, cadmium, copper, mercury, selenium, and zinc bioaccumulate, with cadmium in highest concentration in leatherbacks versus any other marine vertebrate (Caurant et al. 1999; Gordon et al. 1998). A diet of primarily jellyfish, which have high cadmium concentrations, is likely the cause (Caurant et al. 1999). Organochlorine pesticides have also been found (McKenzie et al. 1999). PCB concentrations are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (PCB 209: 500-530 ng/g wet weight Davenport et al. 1990; Oros et al. 2009).

4.2.8.13 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. However, studies do not currently support significant critical habitat deterioration.

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 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. No critical habitat is designated within the MITT action area for this species.

4.2.9 Loggerhead Sea 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).

4.2.9.1 Populations

Five groupings represent loggerhead sea turtles by major sea or ocean basin: Atlantic, Pacific, and Indian oceans, as well as Caribbean and Mediterranean seas. As with other sea turtles, populations are frequently divided by nesting aggregation (Hutchinson and Dutton 2007). On September 22, 2011, the NMFS designated nine distinct population segments (DPSs) of loggerhead sea turtles: South Atlantic Ocean and southwest Indian Ocean as threatened as well as Mediterranean Sea, North Indian Ocean, North Pacific Ocean, northeast Atlantic Ocean, northwest Atlantic Ocean, South Pacific Ocean, and southeast Indo-Pacific Ocean as endangered (75 FR 12598). Recent ocean-basin scale genetic analysis supports this conclusion, with additional differentiation apparent based upon nesting beaches (Shamblin et al. 2014).

Pacific Ocean rookeries are limited to the western portion of the basin. These sites include Australia, New Caledonia, New Zealand, Indonesia, Japan, and the Solomon Islands.

Population structure in the Pacific is comprised of a northwestern Pacific nesting aggregation in Japan and a smaller southwestern nesting aggregation in Australia and New Caledonia (NMFS 2006e). Genetics of Japanese nesters suggest that this subpopulation is comprised of genetically distinct nesting colonies (Hatase et al. 2002a). Almost all loggerheads in the North Pacific seem to stem from Japanese nesting beaches (Bowen et al. 1995a; Resendiz et al. 1998a). The fidelity of nesting females to their nesting beach allowed differentiation of these subpopulations and the loss of nesting at a beach means a significant loss of diversity and the beach is unlikely to be recolonized (NMFS 2006e).

4.2.9.2 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 1998f). 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. 2004b). 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 1991; NRC 1990b; 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.

4.2.9.3 Migration and Movement

Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (Carr 1986). After 14 to 32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (Bowen et al. 2004a; NMFS 2001b). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998a). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer to the nesting location (Girard et al. 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. As water temperatures drop from October to December, most loggerheads emigrate from their summer developmental habitats to warmer waters south of Cape Hatteras, where they winter (Morreale and Standora 1998). For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10 to 15˚ C isotherm (Mansfield et al. 2009). Wallace et al. (2009) suggested differences in growth rate based upon these foraging strategies. Long Island Sound, Core Sound, Pamlico Sound, Cape Cod Bay, and Chesapeake Bay are the most frequently used juvenile developmental habitats along the Northeast United States Continental Shelf Large Marine Ecosystem (Burke et al. 1991; Delannoy et al. 2013; Epperly et al. 1995a; Epperly et al. 1995b; Epperly et al. 1995c; Hoffman et al. 2013; Mansfield 2006). There is conflicting evidence that immature loggerheads roam the oceans in currents and eddies and mix from different natal origins or distribute on a latitudinal basis that corresponds with their natal beaches (Monzon-Arguello et al. 2009; Wallace et al. 2009). McCarthy et al. (2010) found that movement patterns of loggerhead sea turtles were more convoluted when sea surface temperatures were higher, ocean depths shallower, ocean currents stronger, and chlorophyll a levels lower. Satellite tracking of loggerheads from southeastern U.S. nesting beaches supports three dispersal modes to foraging areas: one northward along the continental shelf to the northeastern U.S., broad movement through the southeastern and mid-Atlantic U.S., and residency in areas near breeding areas (Reina et al. 2012).

Aerial surveys sponsored by the U.S. Navy January to August 2009 sighted 193 loggerhead turtles off the coast of Jacksonville, Florida, while line-transect surveys off North Carolina during the same period sighted 41 loggerhead sea turtles (Arbelo et al. 2012). Aerial observations in Onslow Bay from August 2009 through August 2010 sighted 495 loggerhead sea turtles, while vessel surveys during the same period sighted 47 loggerhead sea turtles (Ramsey 2013). Aerial surveys conducted between August 2009 and August 2010 off Jacksonville, Florida, sighted 716 loggerhead sea turtles, while vessel surveys during the same period sighted 47 loggerhead sea turtles (Ramsey 2013).

Individuals in the western Pacific also show wide-ranging movements. Loggerheads hatched on beaches in the southwest Pacific have been found to range widely in the southern portion of the basin, with individuals from populations nesting in Australia found as far east as Peruvian coast foraging areas still in the juvenile stage (Boyle et al. 2009). Individuals hatched along Japanese coasts have been found to migrate to waters off Baja California via the North Pacific Subtropical Gyre (and the Kuroshio Extension) to feed for several years before migrating back to western Pacific waters to breed (Bowen et al. 1995b; Nichols 2005; Polovina et al. 2006; Polovina et al. 2000; Resendiz et al. 1998b). Adult loggerheads also reside in oceanic waters off Japan (Hatase et al. 2002b). Habitat use off Japan may further be partitioned by sex and size (Hatase et al. 2002b; Hatase and Sakamoto 2004; Hatase et al. 2002c). Loggerheads returning to Japanese waters seem to migrate along nutrient-rich oceanic fronts (Kobayashi et al. 2008; Nichols et al. 2000; Polovina et al. 2000). Individuals bycaught and satellite tracked in Hawaii longline fisheries show individual movement north and south within a thermal range of 15-25º C, or 28-40º N, with juveniles following the 17 to 20º C isotherm (Kobayashi et al. 2008; Nichols et al. 2000; Polovina et al. 2004). The Transition Zone Chlorophyll Front and Kuroshio Extension Current are likely important foraging areas for juvenile loggerheads (Polovina et al. 2004). The Kuroshio Current off Japan may be significant for juvenile and adult loggerheads as a wintering areas for those individuals not migrating south (Hatase et al. 2002c).

Sighting and stranding records support loggerhead sea turtles to be common, year-round residents of the Gulf of Mexico, although their abundance is much greater in the northeastern region versus the northwestern (Davis et al. 2000b; Fritts et al. 1983; Landry and Costa 1999). An estimated 12 percent of all western North Atlantic Ocean loggerhead sea turtles reside in the eastern Gulf of Mexico, with the vast majority in western Florida waters (Davis et al. 2000a; TEWG 1998b) . Loggerheads may occur in both offshore habitats (particularly around oil platforms and reefs, where prey and shelter are available; (Davis et al. 2000b; Fritts et al. 1983; Gitschlag and Herczeg 1994; Lohoefener et al. 1990; Rosman et al. 1987), as well as shallow bays and sounds (which may be important developmental habitat for late juveniles in the eastern Gulf of Mexico (Davis et al. 2000b; Lohoefener et al. 1990; USAF 1996). Offshore abundance in continental slope waters increases during the winter in the eastern Gulf of Mexico, as cooler inshore waters force individuals into warmer offshore areas (Davis et al. 2000b).

Hatchlings dispersing from Libyan shores preferentially move into the eastern Mediterranean and eventually transition into a neritic phase along southern Tunisia (Saied et al. 2012). Positive North Atlantic Oscillation phases tend to promote loggerhead presence in the western Mediterranean (Baez et al. 2014).

4.2.9.4 Growth and Reproduction

Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but absent from tropical areas (NMFS and USFWS 1991b; NRC 1990a; Witherington et al. 2006b). The life cycle of loggerhead sea turtles can be divided into seven stages: eggs and hatchlings, small juveniles, large juveniles, subadults, novice breeders, first year emigrants, and mature breeders (Crouse et al. 1987). Hatchling loggerheads migrate to the ocean (to which they are drawn by near ultraviolet light Kawamura et al. 2009), where they are generally believed to lead a pelagic existence for as long as 7 to 12 years (NMFS 2005b). Based on growth rate estimates, the duration of the open-ocean juvenile stage for North Atlantic loggerhead sea is roughly 8.2 years (Bjorndal et al. 2000b). Loggerheads in the Mediterranean, similar to those in the Atlantic, grow at roughly 11.8 cm/yr for the first six months and slow to roughly 3.6 cm/yr at age 2.5 to 3.5. As adults, individuals may experience a secondary growth pulse associated with shifting into neritic habitats, although growth is generally monotypic (declines with age Casale et al. 2009a; Casale et al. 2009b). Individually-based variables likely have a high impact on individual-to-individual growth rates (Casale et al. 2009b). At 15 to 38 years, loggerhead sea turtles become sexually mature, although the age at which they reach maturity varies widely among populations (Casale et al. 2009b; Frazer and Ehrhart 1985a; Frazer et al. 1994; NMFS 2001b; Witherington et al. 2006b). However, based on new data from tag returns, strandings, and nesting surveys, NMFS (2001b) estimated ages of maturity ranging from 20 to 38 years and a benthic immature stage lasting from 14 to 32 years. Notably, data from several studies showed decreased growth rates of loggerheads in U.S. Atlantic waters from 1997 to 2007, corresponding to a period of 43 percent decline in Florida nest counts (Bjorndal et al. 2013).

Loggerhead mating likely occurs along migration routes to nesting beaches, as well as offshore from nesting beaches several weeks prior to the onset of nesting (Dodd 1988a; NMFS and USFWS 1998d). Females usually breed every 2 to 3 years, but can vary from 1 to 7 years (Dodd 1988a; Richardson et al. 1978). Females lay an average of 4.1 nests per season (Murphy and Hopkins 1984) , although recent satellite telemetry from nesting females along southwest Florida support 5.4 nests per female per season, with increasing numbers of eggs per nest during the course of the season (Tucker 2009). The authors suggest that this finding warrants revision of the number of females nesting in the region. The western Atlantic breeding season is March-August. Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

The Japanese rookeries are the most significant nesting sites for loggerheads in the North Pacific, with nesting occurring on the Japanese mainland, except for Hokkaido, as well as the Ryukyu Islands to the south (Kamezaki 1989; Kamezaki et al. 2003; Sea Turtle Association of Japan 2010; Uchida and Nishiwaki 1995). Nesting generally occurs through summer and fall (April-August, peaking in July), with females returning every two to three years (Iwamoto et al. 1985). Nesting females lay at least three nests of 60 to 115 eggs per nest each season, with roughly two weeks between nests (Eckert 1993b; Iwamoto et al. 1985; Nishimura 1994). Between nests, females appear to swim offshore into the Kuroshio Current, possibly to speed egg development (NMFS and USFWS 1998g; Sato et al. 1998).

Nesting in the Gulf of Mexico does occur, although primarily in Florida, with rare nesting along North and South Padre Island in Texas from April through September, with a peak in June and July (Dodd 1988b; Dodd Jr. 1988a; Hildebrand 1983; Weishampel et al. 2006; Williams-Walls et al. 1983).

4.2.9.5 Gender, Age, and Survivorship

Although information on males is limited, several studies identified a female bias, although a single study has found a strong male bias to be possible (Dodd 1988a; NMFS 2001b; Rees and Margaritoulis 2004). Nest temperature seems to drive sex determination. Along Florida, males primarily derive from earlier-season (LeBlanc et al. 2012). Here, nests ranged from an average sex ratio of 55 percent female to 85 percent between 2000 and 2004 (LeBlanc et al. 2012). Juvenile and adult age classes have a slight female bias in the central Mediterranean Sea of 51.5 percent (Casale et al. 2014).

Additionally, little is known about longevity, although Dodd (1988a) estimated the maximum female life span at 47 to 62 years. Heppell et al. (2003a) estimated annual survivorship to be 0.81 (southeast U.S. adult females), 0.78 to 0.91 (Australia adult females), 0.68–0.89 (southeast U.S. benthic juveniles, and 0.92 (Australia benthic juveniles). Another recent estimate suggested a survival rate of 0.41 or 0.60 (CIs 0.20 to 0.65 and 0.40 to 0.78, respectively), depending upon assumptions within the study (Sasso et al. 2011). Survival rates for hatchlings during their first year are likely very low (Heppell et al. 2003a; Heppell et al. 2003). Higher fecundity is associated with warmer February and lower May temerperatures for loggerheads on the northern Gulf of Mexico (Lamont and Fujisaki 2014).

4.2.9.6 Feeding

Loggerhead sea turtles are omnivorous and opportunistic feeders through their lifetimes (Parker et al. 2005). Hatchling loggerheads feed on macroplankton associated with *Sargassum* spp. communities (NMFS and USFWS 1991b). Pelagic and benthic juveniles forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988a; Wallace et al. 2009). Loggerheads in the deep, offshore waters of the western North Pacific feed on jellyfish, salps, and other gelatinous animals (Dodd Jr. 1988b; Hatase et al. 2002b). Sub-adult and adult loggerheads prey on benthic invertebrates such as gastropods, mollusks, and decapod crustaceans in hard-bottom habitats, although fish and plants are also occasionally eaten (NMFS and USFWS 1998d). Stable isotope analysis and study of organisms on turtle shells has recently shown that although a loggerhead population may feed on a variety of prey, individuals composing the population have specialized diets (Reich et al. 2010; Vander Zanden et al. 2010).

4.2.9.7 Diving

Loggerhead diving behavior varies based upon habitat, with longer surface stays in deeper habitats than in coastal ones. Off Japan, dives were shallower than 30 m (Sakamoto et al. 1993). Routine dives can last 4–172 min (Byles 1988; Renaud and Carpenter 1994; Sakamoto et al. 1990). The maximum-recorded dive depth for a post-nesting female was over 230 m, although most dives are far shallower (9–21 m (Sakamoto et al. 1990). Loggerheads tagged in the Pacific over the course of 5 months showed that about 70 percent of dives are very shallow (<5 m) and 40 percent of their time was spent within 1 m of the surface (Polovina et al. 2003; Spotila 2004a). During these dives, there were also several strong surface temperature fronts that individuals were associated with, one of 20° C at 28° N latitude and another of 17° C at 32° N latitude. In the Mediterranean, dives of over 300 min have been recorded in association with depressed water temperatures and are proposed as an overwintering strategy (Luschi et al. 2013).

4.2.9.8 Vocalization and Hearing

Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999c; Lenhardt 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a). Bartol et al. (1999c) reported effective hearing range for juvenile loggerhead turtles is from at least 250 to 750 Hz. Both yearling and two-year old loggerheads had the lowest hearing threshold at 500 Hz (yearling: about 81 dB re 1 μPa and two-year-olds: about 86 dB re 1 μPa), with thresholds increasing rapidly above and below that frequency (Moein Bartol and Ketten 2006). 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).

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles respond best to sounds between 200 and 700 Hz, with slow declines below 100 Hz and rapid declines above 700 Hz, and almost no sensitivity above 3000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Patterson 1966).

4.2.9.9 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 2004b). 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. (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. (2002b) 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.

The Navy's NMSDD estimates 0.000022 loggerhead sea turtles per km^2 in the MITT action area (DoN 2014).

4.2.9.10 Natural Threats

Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can pose lethal effects. In January 2010, an unusually large cold-stunning event occurred throughout the southeast U.S., with well over 3,000 sea turtles (mostly greens but also hundreds of loggerheads) found cold-stunned. Most survived, but several hundred were found dead or died after being discovered in a cold-stunned state. Eggs are commonly eaten by raccoons and ghost crabs along the eastern U.S. (Barton and Roth 2008). In the water, hatchlings are hunted by herons, gulls, dogfish, and sharks. Heavy loads of barnacles are associated with unhealthy or dead stranded loggerheads (Deem et al. 2009). Brevetoxin-producing algal blooms can result in loggerhead sea turtle death and pathology, with nearly all stranded loggerheads in affected areas showing signs of illness or death resulting from exposure (Fauquier et al. 2013). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in excess of 90 percent of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

4.2.9.11 Anthropogenic Threats

Anthropogenic threats impacting loggerhead nesting habitat are numerous: coastal development and construction, placement of erosion control structures, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach nourishment, beach pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992; Margaritoulis et al. 2003; Mazaris et al. 2009b; USFWS 1998). Surprisingly, beach nourishment also hampers nesting success, but only in the first year post-nourishment before hatching success increases (Brock et al. 2009). Loggerhead sea turtles face numerous threats in the marine environment as well, including oil and gas exploration, marine pollution, trawl, purse seine, hook and line, gill net, pound net, longline, and trap fisheries, underwater explosions, dredging, offshore artificial lighting, power plant entrapment, entanglement in debris, ingestion of marine debris, marina and dock construction and operation, boat collisions, and poaching. At least in the Mediterannean Sea, Anthorpogenic threats appear to disproportionally impact larger (more fecund) loggerheads (Bellido et al. 2010).

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Along the Atlantic coast of the U.S., the NMFS estimated that shrimp trawls capture almost 163,000 loggerhead sea turtles each year in the Gulf of Mexico, of which 3,948 die. However, more recent estimates from suggest interactions and mortality has decreased from pre-regulatory periods, with a conservative estimate of 26,500 loggerheads captured annually in U.S. Atlantic fisheries causing mortality up to 1,400 individuals per year (Finkbeiner et al. 2011). Commercial gillnet fisheries are estimated to have killed 52 loggerheads annually along the U.S. mid-Atlantic (Murray 2013). Pacific bycatch is much less, with about 400 individuals bycaught annually in U.S. fisheries resulting in at least 20 mortalities (Finkbeiner et al. 2011). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die. As a result of the 2006 and 2007 tri-national fishermen’s exchanges in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (Peckham et al. 2008). Prior to this closure, the longline fleet interacted with an estimated 1,160 to 2,174 loggerheads annually, with nearly all (89 percent) of the takes resulting in mortalities (Peckham et al. 2008).

Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (Aguilar et al. 1995; Bolten et al. 1994; Carruthers et al. 2009; Howell et al. 2008; Marshall et al. 2009; Petersen et al. 2009; Tomás et al. 2008). In the Pacific Ocean, between 2,600 and 6,000 loggerhead sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004). Shallow-set Hawaii based longline fisheries are estimated to have captured and killed several hundred loggerhead sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about fewer than 5 loggerhead sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii are estimated to have captured about 45 loggerhead sea turtles, killing about 10 of these sea turtles. Longline hooking along Hawaii and California suggests a 28 percent mortality rate for hooked and released loggerheads, with no significant difference between shallow- versus deep-hooked individuals (Swimmer et al. 2013). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino et al. 2010). (Lino et al. 2010). Roughly 10,000 loggerheads are bycaught by longline fisheries in the southwestern Mediterranean annually (Àlvarez de Quevedo et al. 2013). Of these, from 30 to 40 percent are expected to die, resulting in 3,400-4,000 deaths per year, or about 10 percent of the loggerheads present in the region (Àlvarez de Quevedo et al. 2013). In the Pacific, loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including Japanese longline fisheries in the western Pacific Ocean and South China Seas direct harvest and commercial fisheries off Baja California, Mexico commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru purse seine fisheries for tuna in the eastern tropical Pacific Ocean California/Oregon drift gillnet fisheries (NMFS 2006e). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010c); many of these are expected to be loggerhead sea turtles.

Marine debris ingestion can be a widespread issue for loggerhead sea turtles. More than one-third of loggerheads found stranded or bycaught had ingested marine debris in a Mediterranean study, with possible mortality resulting in some cases (Lazar and Gračan 2010). Another study in the Tyrrhenian Sea found 71 percent of stranded and bycaught sea turtles had plastic debris in their guts (Campani et al. 2013). Another threat marine debris poses is to hatchlings on beaches escaping to the sea. Two thirds of loggerheads contacted marine debris on their way to the ocean and many became severely entangled or entrapped by it (Triessnig et al. 2012).

Climate change may also have significant implications on loggerhead populations worldwide. In addition to potential loss of nesting habitat due to sea level rise, loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1 to 2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (Hawkes et al. 2007a). Over time, this can reduce genetic diversity, or even population viability, if males become a small proportion of populations (Hulin et al. 2009). Sea surface temperatures on loggerhead foraging grounds correlate to the timing of nesting, with higher temperatures leading to earlier nesting (Mazaris et al. 2009a; Schofield et al. 2009). Increasing ocean temperatures may also lead to reduced primary productivity and eventual food availability. This has been proposed as partial support for reduced nesting abundance for loggerhead sea turtles in Japan; a finding that could have broader implications for other populations in the future if individuals do not shift feeding habitat (Chaloupka et al. 2008b). Warmer temperatures may also decrease the energy needs of a developing embryo (Reid et al. 2009). Pike (2014) estimated that loggerhead populations in tropical areas produce about 30 percent fewer hatchlings than do populations in temperate areas. Historical climactic patterns have been attributed to the decline in loggerhead nesting in Florida, but evidence for this is tenuous (Reina et al. 2013).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Alava et al. 2006; Corsolini et al. 2000; Gardner et al. 2003; Keller et al. 2005; Keller et al. 2004a; Keller et al. 2004b; McKenzie et al. 1999; Monagas et al. 2008; Oros et al. 2009; Perugini et al. 2006; Rybitski et al. 1995; Storelli et al. 2007a). It appears that levels of organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2004c; Keller et al. 2006b; Oros et al. 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007a). It is likely that the omnivorous nature of loggerheads makes them more prone to bioaccumulating toxins than other sea turtle species (Godley et al. 1999; McKenzie et al. 1999). PAH pollution from petroleum origins has been found in Cape Verde loggerheads, where oil and gas extraction is not undertaken in the marine environment (Camacho et al. 2012).

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Anan et al. 2001; Fujihara et al. 2003; Garcia-Fernandez et al. 2009; Gardner et al. 2006; Godley et al. 1999; Saeki et al. 2000; Storelli et al. 2008). These metals likely originate from plants and seem to have high transfer coefficients (Anan et al. 2001; Celik et al. 2006; Talavera-Saenz et al. 2007). Elevated mercury levels are associated with deformities in hatchlings versus healthy individuals (Trocini 2013). Loggerhead sea turtles have higher mercury levels than any other sea turtle studied, but concentrations are an order of magnitude less than many toothed whales (Godley et al. 1999; Pugh and Becker 2001). Arsenic occurs at levels several fold more concentrated in loggerhead sea turtles than marine mammals or seabirds.

Also of concern is the spread of antimicrobial agents from human society into the marine environment. Loggerhead sea turtles may harbor antibiotic-resistant bacteria, which may have developed and thrived as a result of high use and discharge of antimicrobial agents into freshwater and marine ecosystems (Foti et al. 2009).

4.2.9.12 Critical Habitat

On July 10, 2014, NMFS designated critical habitat for loggerhead sea turtles along the U.S. Atlantic and Gulf of Mexico coasts from North Carolina to Mississippi (79 FR 39856). No critical habitat is designated within the MITT action area for this species.

4.2.10 Scalloped Hammerhead Shark – Indo-West Pacific DPS

Scalloped hammerhead sharks are moderately large sharks with a flat, laterally extended head with a scalloped anterior margin. 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).

4.2.10.1 Populations

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 (*Sphyrna lewini*) as threatened species under the Endangered Species Act (ESA) (79 FR 38213). 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 (79 FR 38213). Populations are generally delineated by ocean basins based on discrete differences in genetic structure and limited transoceanic migrations of this species.

4.2.10.2 Distribution

The scalloped hammerhead shark can be found in coastal warm temperate and tropical seas worldwide. In the western Atlantic Ocean the scalloped hammerhead range extends from the northeast coast of the United States to Brazil, including the Gulf of Mexico and the Caribbean Sea. In the eastern Atlantic, it can be found from the Mediterranean to Namibia. Populations in the Indian Ocean are found from South Africa to Pakistan, India and Myanmar. In the western Pacific the scalloped hammerhead can be found from Japan and China to New Caledonia, including throughout the Philippines, Indonesia, Australia, and the U.S. territorial islands. In the eastern Pacific the scalloped hammerhead can be found southern California to Peru, including the Gulf of California. In the central pacific the scalloped hammerhead can be found in Hawaii and Tahiti.

4.2.10.3 Migration and movement

Scalloped hammerhead sharks are highly mobile and partly migratory. Along continental margins and between oceanic islands in tropical waters migration is common. Adult migratory movements are generally less than 200 km but this species is also capable of moving much greater distances up to approximately 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.

4.2.10.4 Habitat

Scalloped hammerhead sharks primarily occur over continental and insular shelves and rarely in waters cooler than 22° C. It ranges from surface waters to depths of 512 m with occasional dives to deeper water up to 1000 m. It is also known to occur in bays and estuaries. Neonates and juveniles aggregations are more common in nearshore nursery habitats that may provide valuable refuge from predation. Anecdotal information suggests that Guam's inner Apra Harbor could provide nursery habitat for this species (Miller et al. 2014). Scalloped hammerhead sharks appear to prefer areas with stronger currents, greater turbidity, and higher sedimentation and nutrient flow.

4.2.10.5 Growth and reproduction

The scalloped hammerhead shark is viviparous (i.e., give birth to live young), with a gestation period of 9 to 12 months. Females attain sexual maturity at 200 to 250 cm total length (TL) while males reach maturity at 128 to 200 cm TL. Maturity times vary regionally but can range from 4 to 15 years for females and 3.8 to 10 years for males. Parturition likely occurs inshore with peak neonate abundance occurring during the spring and summer. Females give birth to litter sizes up to 41 live pups measuring 31 to 59 cm TL. Maximum size for females and males is over 3 m TL and a maximum age up to 30 years.

4.2.10.6 Feeding

Scalloped hammerhead sharks are high trophic level predators and opportunistic feeders with a diet including teleosts, cephalopods, crustaceans, and rays. Juvenile sharks in Kāneohe Bay, Hawaii were observed to feed primarily at night. Feeding occurs both at reef sites and in pelagic waters.

4.2.10.7 Vocalization and hearing

Scalloped hammerhead sharks, like all fish, have an inner ear capable of detecting high-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 Hz to 1000 Hz with the highest sensitivity to sounds at the lower ranges (Casper et al. 2003; Casper and Mann 2006; Myrberg Jr. 2001).

4.2.10.8 Status and trends

The Final Rule to list the Central SW Atlantic DPS, Eastern Atlantic DPS, Indo-West Pacific DPS, and the Eastern Pacific DPS (79 FR 38213) and the Status Review Report (Miller et al. 2014) provide detailed discussion of the status of each DPS. Logistic and Fox modeling efforts using the best available data suggest a decrease in global abundance from 142,000 and 169,000 individuals (respectively by model) in 1981 to 24,850 and 27,900 individuals (respectively) in 2005. This represents an 83 percent decrease in global abundance based on both the logistic and Fox models over a 15 year period.

The Indo-West Pacific DPS of scalloped hammerhead sharks is listed as threatened and no take prohibitions have been implemented (79 FR 38213). For the Indo-West Pacific DPS, commercial and artisanal fisheries were identified as a high risk to the extinction due to targeted catch and bycatch. The inadequacy of current regulatory mechanisms was also identified as a moderate risk, with illegal fishing significantly contributing to the DPS' risk of extinction. Multiple Regional Fishery Management Organizations (RFMO) cover the Indo-West Pacific DPS area with requirements of full utilization of any retained catches of sharks and regulations that onboard fins cannot weigh more than 5 percent of the weight of the sharks. These regulations are aimed at curbing the practice of shark finning, but do not prohibit the fishing of sharks. In addition, these regulations may not even be effective in stopping finning of scalloped hammerhead sharks, as a recent study found the scalloped hammerhead shark to have an average wet-fin-to-round-mass ratio of only 2.13 percent (n = 81; (Biery and Pauly 2012). This ratio suggests that fishing vessels operating in these RFMO convention areas would be able to land more scalloped hammerhead shark fins than bodies and still pass inspection. There are no scalloped hammerhead-specific RFMO management measures in place for this region, even though this DPS is heavily fished. Consequently, this species has seen population declines off the coasts of South Africa and Australia, so much so that in 2012, New South Wales, Australia, listed it as an endangered species.

Few countries within this DPS' range have regulations aimed at controlling the exploitation of shark species. Oman, Seychelles, Australia, South Africa, Taiwan, and most recently India all have measures to prevent the waste of shark parts and discourage finning. The Maldives have designated their waters as a shark sanctuary. A number of Pacific Island countries (including U.S. territories) have also created shark sanctuaries, prohibited shark fishing, or have strong management measures to control the exploitation of sharks in their respective waters, including Tokelau, Palau, Marshall Islands, American Samoa, CNMI, Cook Islands, and French Polynesia, although effective enforcement of these regulations is an issue for some of the countries. Additionally, many of the top shark fishing nations and world's exporters of fins are also located within the range of this DPS, and have little to no regulation (or enforcement) of their expansive shark fisheries. For example, off northern Madagascar, where there is an active artisanal fin fishery, sharks are an open access resource, with no restrictions on gear, established quotas, or fishing area closures (Robinson and Sauer 2011). Indonesia, which is the top shark fishing nation in the world, does not currently have restrictions pertaining to shark fishing or finning. Indonesian small-scale fisheries, which account for around 90 percent of the total fisheries production, are not required to have fishing permits (Varkey et al. 2010), nor are their vessels likely to have insulated fish holds or refrigeration units (Tull 2009), increasing the incentive for shark finning by this sector (Lack and Sant 2012). Ultimately, their fishing activities remain largely unreported (Varkey et al. 2010), which suggests that the estimates of Indonesian shark catches are greatly underestimated. In fact, in Raja Ampat, an archipelago in Eastern Indonesia, Varkey et al. (2010)estimated that 44 percent of the total shark catch in 2006 was unreported (including small-scale and commercial fisheries' unreported catch and illegal, unregulated, and unreported (IUU) fishing). Although Indonesia adopted an FAO recommended shark conservation plan (National Plan of Action—Shark) in 2010, due to budget constraints, it can only focus its implementation of key conservation actions in one area, East Lombok (Satria et al. 2011). Due to this historical and current absence of shark management measures, especially in the small-scale fisheries sector, many of the larger shark species in Indonesian waters have already been severely overfished (Field et al. 2009).

In addition to the largely unregulated fishing of this DPS, illegal fishing, especially for shark fins, has been identified as a significant contributor to the extinction risk of this DPS. Scalloped hammerhead sharks are valued for their large fins, which fetch a high commercial value in the Asian shark fin trade (Abercrombie et al. 2005)and comprise the second most traded fin category in the Hong Kong market (Clarke et al. 2006). Due to this profit incentive, there have been many reports of finning and seizures of illegally gained shark fins throughout the range of this DPS, including inwaters of Australia (Field et al. 2009), Mozambique, South Africa, Bay of Bengal, Arabian Gulf, Palau, the Federated States of Micronesia (FSM) (Paul 2009), and Somalia (Force 2006). Agnew et al. (2009)provided regional estimates of illegal fishing (using FAO fishing areas as regions) and found the Western Central Pacific (Area 71) and Eastern Indian Ocean (Area 57) regions to have relatively high levels of illegal fishing (compared to the rest of the regions), with illegal and unreported catch constituting 34 and 32 percent of the region's catch, respectively.

Off the coast of Oman, scalloped hammerhead sharks experienced a notable decline in abundance in 2003 and is apparently being replaced by smaller elasmobranch species and smaller individuals of the same species, a trend that may be occurring in other areas as well (Henderson et al. 2007). Declines in abundance in Indonesia may also be occurring as catch in longline fisheries has decreased from 15 to 2 percent from 2001 to 2011 (FAO 2013). In contrast, catch off of the coast of India suggest a potential increase in abundance, although scalloped hammerhead shark size appears to be decreasing (CITES 2012). Data from Australia indicates a decline of 58 to 76 percent from 1996 to 2005 along Northern Australia (Heupel and McAuley 2007) and a decline of more than 90 percent from 1973 to 2008 along New South Wales (Reid and Krogh 1992; Williamson 2011). In the coastal waters of Papau New Guinea scalloped hammerhead catch decreased by 43 percent from 2011 to 2012 (De Young 2006).

Although the number of shark management and conservation measures for this DPS is on the rise, the NMFS Extinction Risk Analysis team noted that the current protections that they afford the Indo-West Pacific DPS may be minimal if illegal fishing is not controlled. We agree and conclude that the inadequacy of current regulatory mechanisms, in the form of ineffective enforcement of current regulations or lack of existing regulatory measures, in combination with illegal fishing, is contributing significantly to the risk of extinction of this DPS.

4.2.10.9 Natural threats

While not actually threats, natural factors such as the lengthy age to sexual maturity, relatively small maximum size, and obligate ram ventilation systems of this species makes them particularly vulnerable to depletion and slow recovery from anthropogenic threats.

4.2.10.10 Anthropogenic threats

The Final Rule to list the Central SW Atlantic DPS, Eastern Atlantic DPS, Indo-West Pacific DPS, and the Eastern Pacific DPS (79 FR 38213) and the Status Review Report (Miller et al. 2014) provide detailed discussion of the threats to each DPS. As described in the Rule, the primary factors responsible for the decline of these four DPSs are overutilization, due to both catch and bycatch of these sharks in fisheries, and inadequate regulatory mechanisms for protecting these sharks, with illegal fishing identified as a significant problem.

4.2.10.11 Critical habitat

The NMFS has not designated critical habitat for scalloped hammerhead sharks.

4.2.11 Acropora globiceps

Colonies of *Acropora globiceps* have finger-like branches. The size and appearance of branches depend on degree of exposure to wave action but are always closely compacted. Colonies exposed to strong wave action have pyramid-shaped branchlets. Colonies can be uniform blue, cream, brown, or fluorescent green in color. *Acropora globiceps* can easily be confused with other *Acropora* species due to both colony and branch morphology, as colonies often appear similar to *Acropora gemmifera*, and in strong wave action colonies appear similar to *Acropora monticulosa*. In addition, *Acropora globiceps*' branch thickness and colony shape are similar to that of *Acropora humilis*, and its branch shape and radial corallite morphology is similar to that of *Acropora samoensis*. It appears that this species has often been mistaken for *A. humilis* (Fenner 2014). However, Veron (2014) states that *Acropora globiceps* is distinctive, and the final rule concluded that it can be identified by experts (79 FR 53852).

4.2.11.1 Distribution and Abundance

Acropora globiceps is distributed from the oceanic west Pacific to the central Pacific as far east as the Pitcairn Islands. Veron (2014) reported that *Acropora globiceps* is confirmed in 22 of his 133 Indo-Pacific ecoregions, and strongly predicted to be found in an additional 16. Wallace (1999a) reports its occurrence in seven of her 29 Indo-Pacific areas, many of which are significantly larger than Veron's ecoregions. The map presented in Wallace (1999a) shows it from a smaller area than Veron (Veron, 2000; Veron, 2014). Based on the Wallace (1999a) range, *Acropora globiceps* has a relatively small range, estimated at 5 million km^2 (Richards

2009). Within U.S. waters, this species is confirmed in American Samoa, Guam, CNMI, and PRIAs (http://www.fpir.noaa.gov/PRD/prd_coral.html).

Veron (2014) reports that *Acropora globiceps* occupied 3.2 percent of 2,984 dive sites sampled in 30 ecoregions of the Indo-Pacific, and had a mean abundance rating of 1.95 on a 1 to 5 rating scale at those sites in which it was found. Based on this semi-quantitative system, the species' abundance was characterized as ''uncommon.'' Overall abundance was described as ''sometimes common.'' Veron did not infer trends in abundance from these data. According to the Final Rule (79 FR 53852) *Acropora globiceps*' absolute abundance is at least tens of millions of colonies at the time of listing.

Within the Mariana Islands, *Acropora globiceps* is confirmed in both Guam and CNMI (NMFS 2015 "Listed Corals in the Indo-Pacific: *Acropora globiceps*, 4 p."). On Guam, a recent review of available coral survey data from numerous sites around the island showed *Acropora globiceps* at dozens of locations around the island. Several surveys have been conducted within Apra Harbor (e.g., (Smith et al. 2009); (Starmer 2008)) and in only three instances (Lybolt 2015; Schils et al. 2011), was *Acropora globiceps* observed. In CNMI, coral survey data shows *Acropora globiceps* on reef slopes around Rota, Tinian, Saipan, Pagan, Anatahan, and Maug (Doug Fenner, personal communication, 2015; Stephen H. Smith, personal communication to Julie Rivers, August 2017). It has also been observed in low abundances around FDM (DoN 2005a). *Acropora globiceps* is by far the most common ESA-listed coral species in the Mariana Islands.

4.2.11.2 Habitat

Acropora globiceps inhabits intertidal, upper reef slopes and reef flats (Veron 2000). Although it most commonly occurs at depths of 0 to 8 m (Veron (2000), it has been recorded as deep as 20 m in the Mariana Islands (Fenner and Burdick 2016) and 15 m in American Samoa (Doug Fenner, pers. comm. to Lance Smith, NMFS).

4.2.11.3 Reproduction and Growth

Acropora are sessile colonies that spawn their gametes into the water column, and the azooxanthellate larvae can survive in the planktonic stage from 4 to 209 days. All species of the genus *Acropora* studied to date are simultaneous hermaphrodites, with a gametogenic cycle in which eggs develop over a period of about 9 months and testes over about 10 weeks. Fecundity in *Acropora* colonies is generally described as ranging from 3.6 to 15.8 eggs per polyp. Mature eggs of species of *Acropora* are large when compared with those of other corals, ranging from 0.53 to 0.90 mm in mean diameter. For five *Acropora* species, the minimum reproductive size ranged from 4 to 7 cm, and the estimated ages ranged from 3 to 5 years (Brainard et al. 2011b).

Acropora globiceps is a hermaphroditic spawner with lecithotrophic (yolk-sac) larvae (79 FR 53852). *Acropora* spp. release gametes as egg-sperm bundles that float to the sea surface, each polyp releasing all its eggs and sperm in one bundle. Fertilization takes place after the bundles break open at the sea surface. Sperm concentrations of 10^6 ml^{-1} have been found to be optimal for fertilization in the laboratory, and concentrations of this order have been recorded in the field during mass spawning events. Self-fertilization, although possible, is infrequent. Gametes remain viable and achieve high fertilization rates for up to 8 hours after spawning. Embryogenesis takes place over several hours, and further development leads to a planula that is competent to settle in 4 to 5 days after fertilization. *Acropora* spp. can show a high degree of hybridization, which can complicate taxonomic classification but allow persistence of the genus if the hybrids are reproductively viable (Brainard et al. 2011).

Larvae presumably experience considerable mortality (up to 90 percent or more) from predation or other factors prior to settlement and metamorphosis (NMFS 2014). Such mortality cannot be directly observed, but is inferred from the large amount of eggs and sperm spawned versus the much smaller number of recruits observed later. Coral larvae are relatively poor swimmers; therefore, their dispersal distances largely depend on the duration of the pelagic phase and the speed and direction of water currents transporting the larvae. The potential for long-term dispersal of coral larvae, at least for some species, may be substantially greater than previously understood and may partially explain the large geographic ranges of many species (NMFS 2014).

The tissue thickness of *Acropora* species is 1 to 2 mm thick, considerably thinner than many coral species, which allows them to grow quicker (Loya et al. 2001). The smaller reserve of nutrients in the thin layer of tissue also makes *Acropora* species more susceptible to environmental distrurbance (Grottoli et al. 2004; Rodrigues and Grottoli 2007; Rodrigues et al. 2008). *Acropora* species generally reach sexual maturity at 4 to 7 cm or roughly 3 to 5 years of age (Wallace 1985).

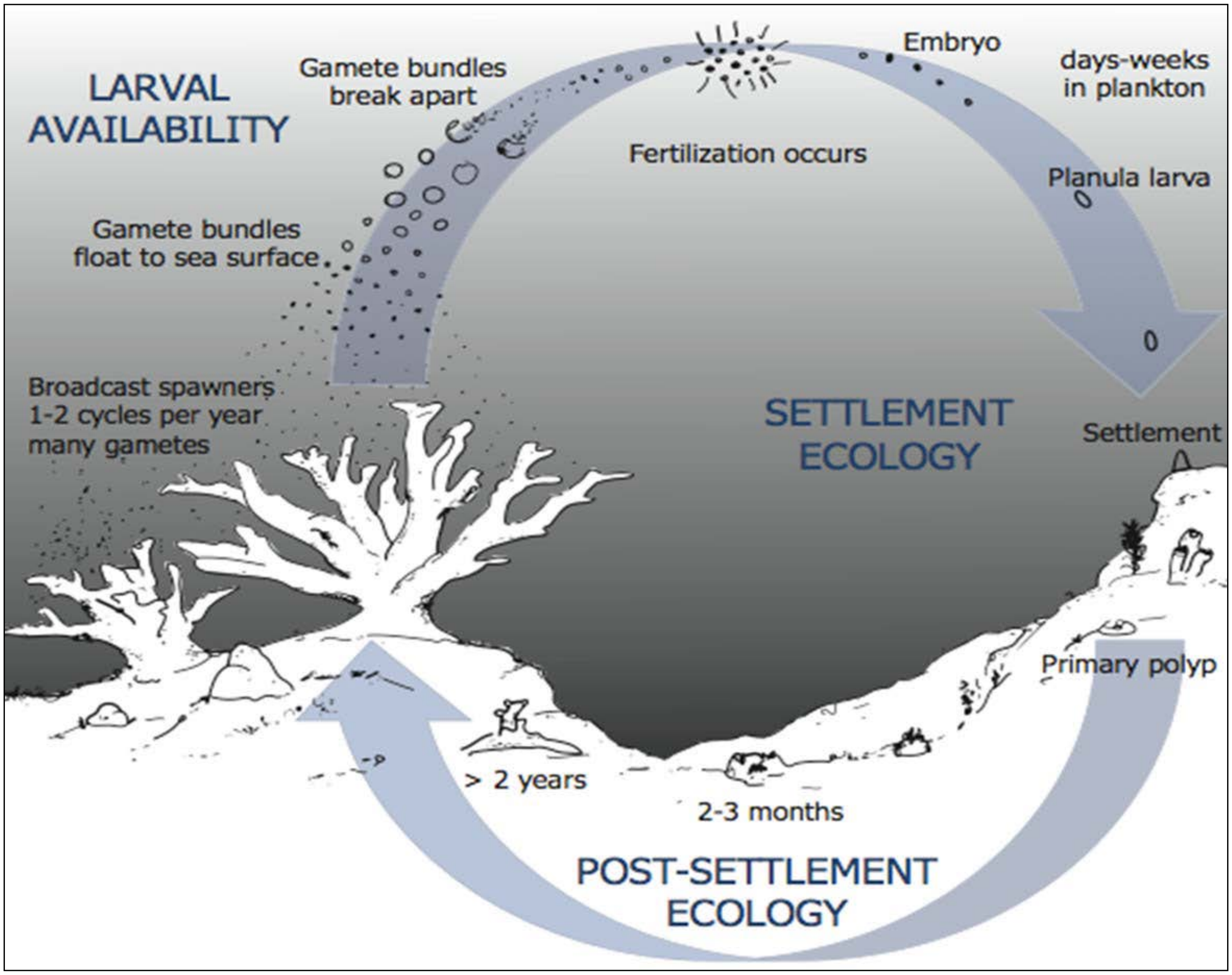

Figure 17. Typical Life Cycle of Broadcast Spawning Corals

4.2.11.4 Status and trends

Acropora globiceps was listed as threatened on September 10, 2014 (79 FR 53851). The abundance of *Acropora globiceps* has likely declined over the past 50 to 100 years although a precise quantification is not possible based on the limited species specific information. The species is highly susceptible to ocean warming, and is susceptible to coral disease, ocean acidification, trophic effects of fishing, nutrients over-enrichment, and predation, all of which have increased in the past 50 to 100 years, and continue to increase throughout much of its range. *Acropora globiceps*' geographic distribution includes the Coral Triangle, but also includes many coral reef ecoregions in the western and central Pacific Ocean, as far east as the Pitcairn Islands. Its depth range is at least zero to 8 meters in upper reef slopes, reef flats, and adjacent habitats (79 FR 53852), though at FDM, the species is typically observed between 15 and 25 meter depths (Stephen H. Smith, personnel communication to NMFS; March 3, 2017). Its absolute abundance is at least tens of millions of colonies across its range. While spatial variability of threats such as ocean warming and acidification across the species' range moderates vulnerability because many colonies are either not exposed to threats or do not negatively respond to a threat at any given point in time, the threats are increasing and will continue to increase in the foreseeable future, thus the species is expected to continue to decline. However, there is inadequate data to quantify current and future population status and trends on any spatial scale, let alone across the range of the speces (79 FR 53852).

4.2.11.5 Natural threats

Of the nine primary threats to reef-building corals (ocean warming, coral disease, ocean acidification, sedimentation, nutrients, trophic effects of fishing, sea-level rise, predation, and collection and trade), only predation is considered a natural threat. Due to the lack of species-specific information on the susceptibility of *Acropora globiceps* to predation, we summarize the available information on this threat to *Acropora* species in general: The studies cited in the Status Review Report (Brainard et al. 2011b) and Final Rule (79 FR 53852) on predation in *Acropora* report that predators such as crown-of-thorns starfish and *Drupella* snails prefer to eat *Acropora* over other genera. Thus, the available information suggests that *Acropora globiceps* is likely to have some susceptibility to predation (79 FR 53852).

4.2.11.6 Anthropogenic threats

Of the nine primary threats to reef-building corals (ocean warming, coral disease, ocean acidification, sedimentation, nutrients, trophic effects of fishing, sea-level rise, predation, and collection and trade), all but predation are considered anthropogenic threats. Due to the lack of species-specific information on the susceptibilities of *Acropora globiceps* to these threats, we summarize the available information for *Acropora* species in general.

Based on the information from the Status Review Report (Brainard et al. 2011b) and Final Rule (79 FR 53852), we make the following inferences regarding the susceptibilities of an unstudied *Acropora* species to these eight anthropogenic threats. For ocean warming, nearly all the studies cited on thermal stress in *Acropora* report high levels of bleaching in response to warming events. Thus, it is possible to predict that an unstudied *Acropora* species is likely to be highly susceptible to warming-induced bleaching, as long as some considerations are kept in mind: (1) Despite high overall susceptibility within the genus to warming-induced bleaching, there can be high variability between species and habitats (Done et al. 2003); (2) colonies that bleach do not necessarily die (in general, *Acropora* species have higher post-bleaching mortality than corals as a whole, but there is high variability in response throughout the genus); (3) recovery from bleaching provides the mechanism for acclimatization; and (4) while most *Acropora* species readily bleach in response to warming events, most also have the capacity to reestablish local populations relatively quickly through their rapid growth and asexual reproduction capacity.

Diseases are generally more common in *Acropora* than other coral genera, although there are numerous documented exceptions, depending on location. These studies also demonstrate high variability in disease susceptibility across *Acropora* species, depending on growth form, with wide divergence of disease susceptibilities among colony morphological groups under the same conditions. Thus, it is possible to predict that an unstudied *Acropora* species is likely to have some susceptibility to disease. The studies cited above on ocean acidification in *Acropora* report impacts on skeletal growth rates. Thus, it is possible to predict that an unstudied *Acropora* species is likely to have some susceptibility to ocean acidification in terms of impacts on skeletal growth. Most studies on the effects of land-based sources of pollution suggest that an unstudied *Acropora* species is likely to have some susceptibility to sedimentation and nutrient enrichment. The trophic effects of fishing alters trophic interactions by reducing herbivory on algae by removal of herbivorous fish from coral reef systems by fishing, thereby providing a competitive advantage for space to algae over coral. An unstudied *Acropora* species is likely to have some susceptibility to the trophic effects of fishing. Because *Acropora* are not generally sediment-tolerant and are faster growing species, an unstudied *Acropora* species is likely to have some susceptibility to sea-level rise. Because *Acropora* species are some of the most popular coral species to collect and trade, an unstudied *Acropora* species is likely to have some susceptibility to collection and trade.

According to the Status Review Report (Brainard et al. 2011b) and Final Rule (79 FR 53852), there is little if any species-specific information on the susceptibilities of *Acropora globiceps* to threats. However, based on genus-level information, we infer that *Acropora globiceps* is likely to be highly susceptible to ocean warming and to have some susceptibility to disease, acidification, sedimentation, nutrients, trophic effects of fishing, sea-level rise, and collection and trade (79 FR 53852).

4.2.11.7 Critical habitat

The NMFS has not designated critical habitat for *Acropora globiceps*.

5 ENVIRONMENTAL BASELINE

The “environmental baseline” includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The following information summarizes the principal natural and human-caused phenomena in the MITT 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 (2006b) 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. 1999; 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). For example, rising sea levels are projected to inundate some sea turtle nesting beaches (Caut et al. 2009b; 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. 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 (Doney et al. 2012; Sydeman et al. 2009; Veit et al. 1996). In the past, warming events have resulted in reduced food supplies for marine mammals along the U.S. west coast (Feldkamp et al. 1991; Hayward 2000; Le Boeuf and Crocker 2005). Some marine mammal distributions may have shifted northward in response to persistent prey occurrence in more northerly waters during El Niño events (Benson et al. 2002; Danil and Chivers 2005; Lusseau et al. 2004; Norman et al. 2004b; Shane 1994; Shane 1995). Low reproductive success and body condition in humpback whales may have resulted from the 1997/1998 El Niño (Cerchio et al. 2005).

In the NMFS's final rule to list 20 coral species as threatened (79 FR 53851), ocean warming and acidification, associated wtih climate change, were identified as two of the most important threats to the current or expected future extinction risk of reef building corals. Reef building organisms, such as the four coral species considered in this opinion, are predicted to decrease the rate at which they deposit $CaCO_3$ in response to increased ocean acidity and warmer water temperatures (Raymundo et al. 2008). Further, 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). Corals can typically withstand mild to moderate bleaching, but severe or prolonged bleaching events can lead to coral colony death (79 FR 53851). While the susceptibility to ocean warming and acidication associated with climate change is expected to vary by species and specific coral colony (based on latitude, depth, bathymetry, etc), climate change is expected to have major impacts on the coral species considered in this opinion. Specifically, the final listing rule identified *Acropora globiceps* as having high vulnerability to the effects of ocean warming and moderate vulnerability to the effects of ocean acidification (79 FR 53851).

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.

5.2 Coastal development and pollution

Coastal development intensifies use of coastal resources, resulting in potential impacts on water quality, marine habitat, and air quality. Development impacts coastal resources through point and nonpoint source pollution, increased sedimentation, concentrated recreational use, and intensive ship traffic using major port facilities. The action area coastline also includes coastal tourism development (e.g., hotels, resorts, restaurants, food industry, vacation homes) and the infrastructure supporting coastal development (e.g., retail businesses, marinas, fishing tackle stores, dive shops, fishing piers, recreational boating harbors, beaches, recreational fishing facilities). Coastal development is regulated by states and territories through the Coastal Zone Management Act and associated state and local programs.

Habitat degradation issues associated with development, such as poor water quality, invasive species, and disease, can alter ecosystems, limiting food availability and altering survival rates. For example, on Saipan, golf course, hotel and tourism-related development has severely impacted most of the historical sea turtle nesting areas on the western portion of the island and residential development threatens the eastern portion of the island. On Rota, turtle nesting beaches appear limited to undeveloped private land due to heavy recreational use and shoreside tourist developments. On Tinian, the majority of the nesting beaches are on military-leased land where no construction is presently expected.

Pollution of the marine environment is a pervasive problem throughout Guam and the inhabited Mariana Islands. Portions of the nearshore marine environment around Guam were severely degraded by impacts from intense combat during World War II (WWII); sunken ships still rest on the sea floor at several locations throughout Apra Harbor on Guam, as does metallic wreckage and other debris. Since WWII, the health of Guam’s marine environment has been affected by the recreational, industrial and commercial operations associated with an increasing population. More recently, sedimentation (from illegal wildfires, improper development, and upland erosion), stormwater runoff and associated pollutants such as fertilizers and oil (from inadequate protections during coastal development and insufficient stormwater management practices and infrastructure) have been identified as the most serious threats to the health of Guam’s nearshore marine environment. Increases in soil erosion can lead to sediment loading in coastal waters which directly impact reef building corals and indirectly impact sea turtles by altering habitat including coral reefs and sea grass beds.

The most common direct effect of sedimentation is deposition of sediment on coral surfaces as it settles out from the water column. Corals with certain morphologies (e.g., mounding) can passively reject settling sediments or corals can actively displace sediment by ciliary action or mucous production, both of which require energetic expenditures (Bak and Elgershuizen 1976; Dallmeyer et al. 1982; Lasker 1980; Stafford-Smith 1993; Stafford-Smith and Ormond 1992). Corals that are unsuccessful in removing sediment will be smothered and die (Golbuu et al. 2003; Riegl and Branch 1995; Rogers 1983). Sediment can also induce sublethal effects, such as reductions in tissue thickness (Flynn et al. 2006) and excess mucus production (Marszalek 1981). In addition, suspended sediment can reduce the amount of light in the water column, making lessenergy available for coral photosynthesis and growth (Anthony and Hoegh Guldberg 2003; Bak 1978; Rogers 1979). While some corals may be more tolerant of short-term elevated levels of sedimentation, sediment stress and turbidity can induce bleaching (Philipp and Fabricius 2003; Rogers 1979). Finally, sediment impedes fertilization of spawned gametes (Gilmour 2002; Humphrey et al. 2008) and reduces larval settlement, as well as the survival of recruits and juveniles (Birrell et al. 2005; Fabricius et al. 2003).

The effects of chemical pollution on marine mammals are just starting to be understood (Aguilar Soto et al. 2008). Recently, the 5.5-year expedition of the *Odyssey* collected 955 biopsy samples from sperm whales around the world to provide a consistent baseline database of ocean contamination and to measure future effects (Ocean Alliance 2010). Chemical pollutants found in pesticides and other substances flow into the marine environment from human use on land and are absorbed into the bodies of marine mammals, accumulating in their blubber, internal organs, or are transferred to the young from mother’s milk (Fair et al. 2010). Important factors that determine the levels of pesticides, heavy metals, and industrial pollutants that accumulate in marine mammals are gender (i.e., adult males have no way to transfer pesticides whereas females may pass pollutants to their calves through milk), habitat, and diet. Living closer to the source of pollutants and feeding on higher-level organisms increase the potential to accumulate toxins (Moon et al. 2010). The buildup of human-made persistent compounds in marine mammals not only increases their likelihood of contracting diseases or developing tumors but also compromises the function of their reproductive systems (Fair et al. 2010).

In addition to the effects of sediment-laden and polluted runoff, Guam’s nearshore waters have been impacted by years of poorly treated wastewater effluent discharges around the island. In 1986, the U.S. Environmental Protection Agency issued permits under the National Pollutant Discharge Elimination System (NPDES) that allowed Guam Waterworks Authority to discharge wastewater effluent into the nearshore marine environment following primary treatment of wastewater. Primary treatment follows coarse screening and grit settlement, and removes only about 60 percent of the suspended solids in the wastewater by allowing the water to rest in settlement tanks that are used to remove material that floats or settles out (Mancl 1996).

Following that, chlorination is normally employed to reduce pathogens. However, the NPDES permits expired in 1991 and were not reissued due to GWA's inability to meet the required standards (Guam 2003).

The lack of adequate wastewater treatment on Guam has contributed to nutrient inputs to nearshore waters. A 2010 assessment by GEPA determined that while most of the 24 assessed bays met water quality guidelines for recreational activities and harvesting, 11 of the bays were impaired. Over 700 swimming advisories due to bacterial counts in marine waters were issued in 2009, likely stemming from faulty septic tanks and non-compliance by treatment facilities with NPDES regulations for various parameters.

The overall health of Guam's coral reefs has declined over time; while it is difficult to assign the causes of this decline to local versus global causes, increased sedimentation and pollutant runoff are known stressors to reef-building corals. The average live coral cover in Guam's nearshore waters was approximately 50 percent in the 1960s, but dwindled to less than 25 percent by the 1990s, with only a few areas having over 50 percent live cover.

5.2.1 Light Pollution

The east end of Apra Harbor is highly developed and brightly illuminated at night. The existing lights from the commercial port are particularly bright and may be clearly visible to nesting turtles and hatchlings in the water, and on the beach, at Spanish Steps. Sea turtle hatchlings are strongly attracted to light (Witherington and Bjorndal 1991a), and use light wavelengths and shape patterns to find the ocean after emerging from the nest (Lohmann et al. 1997; Witherington 1992). Lighting from the commercial port in Apra Harbor, is clearly visible from Spanish Steps (Kevin Brindock, personal communication to Julie Rivers: 29 Oct. 2014); this light pollution may have the potential to impact green turtle reproductive success at the Spanish Steps.

5.3 Marine Debris

Debris can be introduced into the marine environment by its improper disposal, accidental loss, or natural disasters (Watters et al. 2010), and can include plastics, glass, derelict fishing gear, derelict vessels, or military expendable materials. Despite debris removal and outreach to heighten public awareness, marine debris in the environment has not been reduced (Academies 2008) and continues to accumulate in the ocean and along shorelines within the action area. Marine debris affects marine habitats and marine life worldwide, primarily by entangling or choking individuals that encounter it. Entanglement in marine debris can lead to injury, infection, reduced mobility, increased susceptibility to predation, decreased feeding ability, fitness consequences, and mortality for all listed species in the action area. Entanglement can also result in drowning for air breathing marine species including sea turtles, cetaceans, and pinnipeds. Marine debris ingestion can lead to intestinal blockage which can impact feeding ability and lead to death. Data on marine debris in the action area is largely lacking; therefore it is difficult to draw conclusions as the extent of the problem and its impacts on populations of listed species.

Sea turtles can mistake plastic bags for jellyfish, which are eaten by many turtle species in early life phases, and exclusively by leatherback turtles throughout their lives. One study found plastic in 37 percent of dead leatherbacks and determined that 9 percent of those deaths were a direct result of plastic ingestion (Mrosovsky et al. 2009). Other marine debris, including derelict fishing gear and cargo nets, can entangle and drown turtles of all life stages. In studying ingestion in 115 green and hawksbill sea turtles stranded in Queensland, Schuyler et al. (2012) found that the probability of debris ingestion was inversely correlated with size (curved carapace length), and when broken down into size classes, smaller pelagic turtles were significantly more likely to ingest debris than larger benthic feeding turtles. 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.

Plastic debris is a major concern because it degrades slowly and many plastics float. The floating debris is transported by currents throughout the oceans and has been discovered accumulating in oceanic gyres (Law et al. 2010). Additionally, plastic waste in the ocean chemically attracts hydrocarbon pollutants such as PCB and DDT. Fish, marine mammals and sea turtles can mistakenly consume these wastes containing elevated levels of toxins instead of their prey. In the North Pacific Subtropical Gyre it is estimated that the fishes in this area are ingesting 12,000 to 24,000 U.S. tons (10,886,216 to 21,772,433 kilograms [kg]) of plastic debris a year (Davison and Asch 2011).

In December 2013 a distressed juvenile hawksbill turtle was found entangled in marine debris in Garapan Lagoon, Saipan; a nylon rope tied in a loop had caught around the turtle’s carapace and the turtle’s body had apparently distorted around the restricting rope as it grew (Figure 16). The rope was removed and the turtle was released alive.

Figure 18: Juvenile hawksbill turtle entangled in marine debris recovered in Garapan Lagoon, Saipan, December 2013. Image: CNMI Department of Lands & Natural Resources.

Between October 2004 and September 2008, the American Samoa Department of Marine and Wildlife Resources (DMWR) necropsied four green turtles that stranded on Tutuila. Of those four, 2 had plastic and aluminum in their guts (Tagarino et al. 2008). However, because only a small percent of dead or dying sea turtles strand, little information is available to adequately quantify the impacts on sea turtles that may result from marine debris in the action area. Accumulated marine debris on sea turtle nesting beaches can also impede nesting success by altering nest excavation and through potential entrapment of hatchlings under debris that is inadvertently buried over them when the nesting female covers the clutch. The nesting beaches in the Spanish Steps area on Guam heavily impacted by accumulated marine debris and non-native plants.

5.4 Fisheries

Fisheries constitute an important and widespread use of the ocean resources throughout the action area. Fisheries can adversely affect fish populations, other species, and habitats. Direct effects of fisheries interactions include entanglement and entrapment which can lead to fitness consequences or mortality as a result of injury or drowning. Indirect effects include reduced prey availability and destruction of habitat. Potential impacts of fisheries include overfishing of targeted species and bycatch, both of which negatively affect fish stocks and other marine resources. Bycatch is the capture of fish, marine mammals, sea turtles, marine birds, and other nontargeted species that occurs incidental to normal fishing operations. Use of mobile fishing gear, such as bottom trawls, disturbs the seafloor and reduces structural complexity. Indirect impacts of trawls include increased turbidity, alteration of surface sediment, removal of prey (leading to declines in predator abundance), removal of predators, ghost fishing (i.e., lost fishing gear continuing to ensnare fish and other marine animals), and generation of marine debris. Lost gill nets, purse seines, and long-lines may foul and disrupt bottom habitats and have the potential to entangle or be ingested by marine mammals.

Fisheries can have a profound influence on fish populations. In a study of retrospective data, Jackson et al. (2001) analyzed paleoecological records of marine sediments from 125,000 years ago to present, archaeological records from 10,000 years before the present, historical documents, and ecological records from scientific literature sources over the past century. Examining this longer-term data and information, Jackson et al. (2001) concluded that ecological extinction caused by overfishing precedes all other pervasive human disturbance of coastal ecosystems, including pollution and anthropogenic climatic change. Fisheries bycatch has been identified as a primary driver of population declines in several groups of marine species, including sharks, mammals, marine birds, and sea turtles (Wallace et al. 2010b).

Fisheries in the action area range from relatively small-scale, nearshore fisheries around Guam and the Commonwealth of the Northern Mariana Islands, to large-scale longline and purse seine fisheries prosecuted further offshore. Nearshore fisheries in the action area are based out of Guam or CNMI and operate from shore or out of small boats with little distinction among commercial, subsistence, or recreation trips (Council 2011), whereas the offshore fishery in the action area is primarily a commercial fishery, and includes high seas fishing activity from foreign vessels. Domestic fisheries based in Guam and the CNMI likely represent only a small percentage of the total fishing effort in the action area. International fleets, mainly from Asian nations, operate offshore and target pelagic species such as tunas, sharks and mahimahi.

Domestic fisheries based in Guam are divided into two basic categories: offshore and inshore fishing. Offshore fishing typically involves small boats (12–48 feet in length) that engage in one to two day trolling and bottom fishing trips to nearby banks, isles and pelagic areas. Inshore fishing is usually conducted without the use of a boat and consists mainly of casting (rod & reel fishing), throw-netting, and spearfishing. Data from the NOAA Pacific Islands Fishery Science Center indicates that the top fisheries in Guam in 2012, by weight, included skipjack tuna, mahimahi, wahoo, and marlins, as well as reef fish such as parrotfish and unicornfish.

The domestic commercial fishery of the CNMI is mainly a small boat troll fishery. Most fishing vessels are outboard-powered vessels less than 24 feet in length that make trolling trips of generally a day or less in duration. There are a few larger boats that have been used in recent years for bottom fishing around the islands north of Saipan in addition to trolling. Reef fishes make up a significant portion of the total commercial catch and are an important component of the local diet. While the vast majority of the domestic catch is consumed locally, there have been some intermittent exports to Guam, Hawaii, and Japan. Data from the NOAA Pacific Islands Fishery Science Center indicates that the top fisheries in Guam in 2012, by weight, included skipjack tuna, yellowfin tuna, mahimahi, wahoo, and reef fish.

Harvest from small-scale fisheries in the tropical Pacific in general is usually underestimated in official statistics due to the difficulty and costs of quantifying these very spatially diverse fisheries (Zeller et al. 2007). Zeller et al. (2005) re-estimated harvest of reef and bottom fishery catches in Guam and CNMI. The authors used estimated human consumption rates of the local population, and estimated values of exported/imported catches to estimate more representative catch numbers. The authors indicated a 2.5-fold underreporting of catches in Guam from 1965-2002, and a 2.2-fold underreporting of catches in CNMI from 1983 to 2002.

5.4.1 Bycatch

The term "bycatch" refers to any fisheries capture that is incidental to the intended or targeted species and can encompass all unwanted, unmanaged, or discarded animals captured. Bycatch in the action area occurs both as a result of nearshore fisheries based in Guam and CNMI as well as large-scale offshore fisheries operated by foreign fishing fleets. Bycatch is likely the most impactful problem presently facing cetaceans worldwide and may account for the deaths of more marine mammals than any other cause (Geijer and Read 2013; Hamer et al. 2010; Northridge 2008; Read 2008). Cetaceans are prone to bycatch in longline, trawl and purse seine fisheries, and large whales are prone to entanglement in trap or pot fisheries. Entanglement may also make whales more vulnerable to additional dangers, such as predation and ship strikes, by restricting agility and swimming speed. Wallace et al. (2010c) estimated that worldwide, 447,000 turtles are killed each year from bycatch in commercial fisheries. It is likely that the majority of individual sea turtles and marine mammals that are killed by commercial fishing gear are never detected, making it very difficult to accurately determine the number and frequency of mortalities.

Fisheries in the action area are likely result in the incidental capture and mortality of green, loggerhead, leatherback and hawksbill sea turtles, though data on sea turtle bycatch in the region are lacking. As greens and hawksbills nest on Guam and the CNMI, they are more likely to be encountered in nearshore waters are therefore more likely to be affected by nearshore fisheries based on Guam and the CNMI. Gill nets generally represent the most problematic fishery for sea turtles because the nets are often left untended, increasing the likelihood of drowning. Guam law prohibits drift gill nets and requires that staked gill nets be moved every six hours; these regulations would be expected to reduce the probability of mortality for any turtles incidentally captured. No such laws regarding gill nets exist in the CNMI that we are aware of. Sea turtles can also be hooked or entangled in hook-and-line fisheries, though the chance of survival is considered higher than if caught in a gill net. Leatherback sea turtles are known to have been occasionally captured offshore by Guam-based fishermen (Karen Frutchey, NMFS PIRO PRD, personal communication to Jordan Carduner, NMFS OPR, September 2014). In a study of stranded green turtles in Hawaii (those that are found on shore either injured, sick, or dead), the second and third most common known causes of stranding were fishing related. Hook-and-line fishing gear-induced trauma accounted for 7 percent, and gillnet fishing gear-induced trauma was responsible for 5 percent (Chaloupka et al. 2008a). However, most turtles that drown in fishing gear are likely never documented, making it very difficult to estimate the total number of turtles killed annually by nearshore fishing interactions, even in Hawaii where turtles are much better monitored and studied than in the Marianas.

We know very little about incidental fisheries interactions with cetaceans in the nearshore waters surrounding Guam and the CNMI. At the time this opinion was written, the NMFS Pacific Islands Regional Office had not received any reports of hookings or entanglements of cetaceans, nor had they observed any sign of fisheries interactions with cetaceans during their surveys (Erin Oleson, NMFS PIRO PRD, personal communication to Jordan Carduner, NMFS OPR, October 2014). As described above, fishing fleets based out of Guam and CNMI are small in scale and there are very few longline vessels or purse seine vessels that operate out of regional ports; thus fisheries interactions with cetaceans would be less likely among these vessels than among the larger scale fishing fleets that operate offshore within the action area. While we suspect that interactions with cetaceans likely occur among the offshore fisheries in the action area, data on these offshore fleets is scarce.

Bycatch of scalloped hammerhead sharks likely occurs in the action area though we are not aware of any data that is specific to the action area. Apra Harbor in Guam is considered an aggregation area for neonate and juvenile scalloped hammerheads, therefore incidental fisheries interactions with scalloped hammerheads near the harbor is a possibility; however we were not able to locate documentation of such bycatch. Guam law prohibits drift gill nets and requires that staked gill nets be moved every six hours; these regulations would be expected to reduce the probability of mortality for any sharks incidentally captured.

5.4.2 Directed Fisheries

The directed hunting of sea turtles in foraging areas and on nesting beaches as well as the harvesting of eggs from nesting beaches represent ongoing threats to sea turtles in the action area. Directed take through harvest of turtles and their nests continues on Guam and the other inhabited islands of the Mariana archipelago. Turtles were traditionally taken by residents of Guam for celebrations, and reports indicate that illegal harvesting still occurs. Poaching also occurs by immigrants, fishing crews, and tourists, especially those from areas where they are accustomed to eating turtles legally.

Between October 2013 and October 2014, the CNMI Department of Lands & Natural Resources reported two cases of attempted poaching of juvenile green sea turtles and one case of recovered juvenile green turtle remains that appeared consistent with poaching activity on Saipan, as well as the confiscation of five juvenile green turtle carapaces at Saipan International Airport. During the 2009 nesting season on Saipan, three out of what is thought to be a total of five nesting turtles were poached as were three nests. On Guam, DAWR has responded to 17 poachings of green sea turtles and 1 hawksbill since 1975. It is likely that the documented cases of poaching of adult sea turtles and sea turtle eggs represent just a fraction of the actual poaching cases that occur in the action area. Despite the evidence of continued poaching of adult sea turtles and sea turtle eggs, the available data on these activities is not adequate to allow for an accurate estimate of the impacts to listed sea turtle species in the action area.

Figure 19: Evidence of attempted poaching in the CNMI: A live juvenile green turtle found flipper-bound by a rubber strap, Saipan, 13 February 2014. Image courtesy of CNMI Department of Lands & Natural Resources.

Guam bans the possession, sale, or distribution of shark products, while the CNMI bans the landing of sharks at all ports within the archipelago. However, as described above, fisheries based in Guam and the CNMI are almost exclusively nearshore fisheries, and it is likely that offshore fisheries in the action area target sharks. Literature and technical reports describing fishing activity in the action area do not indicate that scalloped hammerhead sharks are directly targeted, though they are captured in shark fisheries that do not differentiate by species. Assessing harvest levels of scalloped hammerheads in the action area is difficult because many catch records do not differentiate among hammerhead species, or shark species in general (Miller 2014). For the nearshore fishery, the Western Pacific Fisheries Information Network (WPacFin) houses reported catch data from the fishery agencies of Guam (Division of Aquatic and Wildlife Resources; Bureau of Statistics and Plans) and CNMI (Division of Wildlife). However, reported shark catches for Guam and CNMI are aggregated into “Pelagic fishes” or “Sharks” categories for reporting purposes, making it difficult to differentiate scalloped hammerhead catch from these data. Similarly for the offshore environment, until 2011, the Western and Central Pacific Fisheries Commission (WCPFC) did not require the offshore fishery in the WCPFC Convention Area (inclusive of the action area) to report species specific information for many shark species, including hammerheads.

Observer data appears to be the most useful representation of species-specific catch rates in the offshore fishery of the action area (though it should be noted that these data are from observed fishing trips throughout the western and central Pacific Ocean and are not specific to the action area). Observer data from 1994 to 2009 indicates that hammerhead shark catch accounted for 0.2 percent of the total observed catch, by weight for longline fisheries (Programme 2010). Observer data from the purse seine fishery during the same time period indicated even lower shark catch rates: excluding catches of silky, whale, and oceanic whitetip sharks which were reported separately, catches of "Other sharks and rays" (inclusive of scalloped hammerhead sharks) represented only 0.01 percent of the total catch by weight of observed purse seine catches.

Scalloped hammerheads, like many shark species, are targeted for their fins because they fetch a high commercial value in the Asian shark fin trade. It is thought that scalloped hammerheads, and sharks in general, are likely under-reported in catch records as many records do not account for discards, or finned individuals (Miller 2014). Observer data from the longline fisheries of the Western and Central Pacific Ocean indicates that of the 104 scalloped hammerheads observed discarded from 1994-2009 (on over 3,000 observed trips), 72 percent of those discarded were finned. Also of note, an additional 157 hammerheads were observed discarded over this time, but were not identified to species. Approximately 61 percent of those discarded individuals were finned. It should be noted that only a very small percentage of fishing vessels in the offshore portions of the action area have observer coverage, and many of the vessels that are most likely to be engaged in shark finning activities are also least likely to carry observers onboard. Many countries and fisheries management entities have aimed to restrict shark finning, though the practice continues in many areas. Since 2008, the Western and Central Pacific Fisheries Commission has attempted to discourage shark finning by requiring that fishing vessels retain all parts of the shark excepting head, guts, and skins, to the point of first landing. Further, onboard fins cannot weigh more than 5 percent of the weight of sharks onboard, up to the first point of landing. Despite these restrictions, illegal fishing activity is well documented, particularly on the high seas where enforcement is difficult. Guam banned the possession, sale, offer for sale, take, purchase, barter, transport, export, import, trade, or distribution of shark fins (with exemptions for research and subsistence fishing) in 2011. Similarly, CNMI banned the possession, sale, offer for sale, trade, or distribution of shark fins (with exemptions for research and subsistence fishing) in 2011.

5.5 Whaling

Large whale population numbers in the action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. During the height of global whaling, Guam was an important stopover for whaling ships in the Pacific Ocean. However, we are not aware of any directed hunting of whales that presently occurs in the action area. Prior to current prohibitions on whaling, most large whale species had been significantly depleted. Table 24 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 24. 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 : (Carretta et al. 2014)	1976
Humpback whale	15,000 whales	1919 - 1987	(Tonnessen and Johnsen 1982); C. Allison, IWC unpubl. Data; cited in: (Carretta et al. 2014)	1966
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: (Carretta 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 25). 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 25. 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; originally accessed on January 24, 2015; updated with 2014 information on January 11, 2017).

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
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
2014	0	124	90
Totals	444	3542	1179

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 the humpack whale has 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.

5.6 Ongoing U.S. Military Training and Testing Activities in the Action Area

This section summarizes the United States Pacific Fleet marine species monitoring under the MMPA LOA for at-sea training in the MIRC. This data was provided by the Navy in in the Comprehensive Exercise and Marine Species Monitoring Report for The U.S. Navy's Mariana Islands Range Complex. Department of the Navy, Commander, U.S. Pacific Fleet, Pearl Harbor, Hawaii. 31 October 2014. The majority of the training and testing activities the Navy conducts in the MITT 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.

There were two individual MTEs that took place in the MIRC from 12 August 2010 to 15 July 2014. These MTEs are summarized in Table 26.

Table 26. Summary of Major Training Exercises in MIRC

Exercise Type	12 Aug 2010 – 15 Feb 2011	16 Feb 2011 – 15 Feb 2012	16 Feb 2012 – 15 Feb 2013	16 Feb 2013 – 15 Feb 2014	16 Feb 2014 – 15 Jul 2014	16 Jul 2014 – 3 Aug 2015	Reporting Period Total
Joint Multi-Strike Group Exercise	1	0	1	0	0	1	3
Total	1	0	1	0	0	1	3

There were 11 reported sightings of an estimated 47 marine mammals during MTEs in the MIRC from 12 August 2010 to 15 July 2014. These sightings are summarized in Table 27.

Table 27. Summary of Sightings during Major Training Exercises

Marine Species	12 Aug 2010 – 15 Feb 2011	16 Feb 2011 – 15 Feb 2012	16 Feb 2012 – 15 Feb 2013	16 Feb 2013 – 15 Feb 2014	16 Feb 2014 – 15 Jul 2014	16 Jul 2014 – 3 Aug 2015	Reporting Period Total
	Estimated Number of Animals Sighted While Sonar Active						
Dolphin	0	0	0	0	0	0	0
Whale	4	0	0	0	0	4	8
Pinniped	0	0	0	0	0	0	0
Turtle	0	0	0	0	0	0	0
Generic	1	0	0	0	0	3	4

Marine Species	12 Aug 2010 – 15 Feb 2011	16 Feb 2011 – 15 Feb 2012	16 Feb 2012 – 15 Feb 2013	16 Feb 2013 – 15 Feb 2014	16 Feb 2014 – 15 Jul 2014	16 Jul 2014 – 3 Aug 2015	Reporting Period Total
Subtotal while Active	**5**	**0**	**0**	**0**	**0**	**7**	**12**
	Estimated Number of Animals Sighted While Sonar Passive						
Dolphin	25	0	8	0	0	1	34
Whale	9	0	0	0	0	5	14
Pinniped	0	0	0	0	0	0	0
Turtle	0	0	0	0	0	0	0
Generic	0	0	0	0	0	0	0
Subtotal while	34	0	8	0	0	6	48
Total	**39**	**0**	**8**	**0**	**0**	**13**	**60**

There were 7 total mitigation events where active sonar was powered down or shut down due to the sighting of marine mammals or sea turtles during MTEs from 12 August 2010 to 3 August 2015. These mitigation events are summarized in Table 28. The Navy's unclassified annual exercise reports from 2010 through August 2015 contain tables listing all marine mammals sighted during that reporting year and the range of the sighting.

Table 28.Summary of Mitigation Events During Major Training Exercises

Marine Animal Species	Range of Detection (Yards, <200, 200-500, 500-1,000, 1,000-2,000, >2,000)	Mitigation Measure Implemented	Un-required Mitigation (Yes/No)
12 August 2010 – 15 February 2011			
Generic	Acoustic detection	Sonar shut down	Yes
Whale	<200	Sonar shut down	No
Whale	200-500	Sonar shut down	No
Whale	>2,000	Sonar shut down/maneuvered	Yes
16 February 2011 – 15 February 2012			
No mitigation events during this period			
16 February 2012 – 15 February 2013			
No mitigation events during this period			
16 February 2013 – 15 February 2014			
No mitigation events during this period			
16 February 2014 – 15 July 2014			
No mitigation events during this period			
16 July 2014 – 3 August 2015			
Generic	<200	Sonar shut down*	--
Generic	501-1000	Sonar shut down*	--
Generic	1001-2000	Sonar shut down*	--

*Note: Mitigation measure implemented by a SURTASS LFA sonar vessel participating in Valient Shield.

5.7 Other U.S. Military Activities in the MITT Action Area

The following sections describe other past and ongoing military activities in the MITT action area.

5.7.1 Guam and CNMI Military Relocation

NMFS PIRO issued a biological opinion on 25 August 2010 on the Department of Navy (DoN) Joint Guam Program Office's Guam and CNMI Military Relocation. The action included three primary components: 1) Relocating U.S. Marines from Okinawa to Guam; 2) Increasing and upgrading vessel berthing and related facilities in Apra Harbor to accommodate increased visits by aircraft carriers and their accompanying ships, collectively referred to as a Carrier Strike Group (CSG), as well as supporting increased visits by Amphibious Taskforce (ATF) ships; and 3) Stationing an Army Air and Missile Defense Task Force on Guam. The action included substantial new upland construction and infrastructure improvements to support the incoming Marine and Army forces. That construction was not expected to directly affect ESA-listed species under NMFS jurisdiction. However, it was determined that the work necessary to increase and upgrade vessel berthing and related facilities in Apra Harbor was likely to affect ESA-listed marine species. The action was expected have indirect impacts on ESA-listed marine species through increased wastewater discharge and increased commercial and recreational vessel traffic that were expected to result from the proposed population increase on Guam. Thus, those actions were subject of the consultation. NMFS determined that an undeterminable number of green and hawksbill sea turtles were expected to be adversely affected by the action, primarily through behavioral modification. As of the drafting of this opinion, the majority of in-water work (Inner Apra Harbor Wharf Repairs) associated with the action in the CNMI Military Relocation consultation has been completed.

In February 2012, the DoN initiated a Supplemental EIS to evaluate the environmental consequences of establishing a live-fire training range complex on Guam in support of the relocation of Marine Corps forces to the island. Scoping meetings for the Supplemental EIS were held in March 2012. On 27 April 2012, the U.S.-Japan Security Consultative Committee issued a joint statement announcing its decision to adjust the plans outlined in the May 2006 Realignment Roadmap document. In accordance with the adjustments (the "2012 Roadmap Adjustments"), the DoD adopted a new force posture in the Pacific which provided a substantially smaller Marine Corps relocation to Guam. As a result of the 2012 Roadmap Adjustments, the DoN expanded the scope of the Supplemental EIS to also evaluate the potential environmental consequences from construction and operation of a main cantonment area, including family housing, and associated infrastructure to support the relocation of a substantially reduced number of Marines than previously analyzed. The Supplemental EIS supplements the 2010 Final EIS for the Guam and CNMI Military Relocation. United States Marine forces are not expected to arrive in Guam until facilities are built and operational. Under the currently proposed timeline, this will not occur until 2020 (DoN 2012).

5.7.2 X-Ray Wharf Improvements

The Department of the Navy is planning improvements to its X-Ray wharf facility at Naval Base Guam in Apra Harbor to accommodate two berths for the Navy's new class of supply ships. ESA Section 7 consultation was completed on 9 January 2014 by NMFS PIRO. The temporary displacement of green and hawksbill sea turtles from Apra Harbor was expected as a result of the wharf construction, but no take was anticipated as a result of the action.

5.7.3 Surveillance Towed Array Sensor System Low Frequency Active Sonar

The Navy operates up to four Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar vessels. Based on current Navy national security and operational requirements, use of these sonar systems could occur in the in the Pacific Ocean (including the action area), Atlantic Ocean, Indian Ocean, and Mediterranean Sea. During the operation of SURTASS LFA sonar, the Navy employs a three-part mitigation and monitoring protocol to avoid minimize the risk of injury to protected species: 1) visual monitoring for protected species during daylight hours, 2) passive (low-frequency) SURTASS to listen for sounds generated by marine mammals as an indicator of their presence, and 3) high frequency active sonar to detect potentially affected protected species. If protected species are detected within the mitigation zone while LFA sonar is active, sonar is suspended or delayed.

As a requirement of the ESA and MMPA authorizations for this activity, the Navy submits quarterly and annual mission reports to NMFS detailing the number of missions conducted, number of protected species observed or detected (either passive or active sonar detection), and the number of times LFA sonar was suspended or delayed to the presence of a protected species. Both the historical and the recent results of the mitigation monitoring and effectiveness support the U.S. Navy's and NMFS' assertions that the U.S. Navy's three-part mitigation and monitoring protocols provide an effective means of avoiding risk of injury to protected marine species.

5.7.4 Dredging, Filling and Explosive Clearing

Apra Harbor is a natural deep-water harbor, which has been heavily modified, particularly since World War II (Figure 20). Much of the harbor's current topography and bathymetry is manmade; the result of work begun by the U.S. Government in 1943. Extensive dredging and fill projects resulted in the creation of Inner Apra Harbor and its channel as well as the creation of Dry Dock Peninsula, Polaris Point, and the manmade northeastern and southeastern shorelines and the Glass Breakwater, which extends from Cabras Island, out and across Luminau Reef to provide increased protection for the harbor. Other impacts include the knolls (hard bottom sites that protrude at least 25 ft (7.6 m) above the harbor bottom) that were explosively cleared during WWII, because they were considered navigational hazards. Some of the shallower knolls have been used as anchorage sites since WWII, and some are still used by military and commercial vessels. The Guam and CNMI military relocation involves additional dredging in Inner Apra Harbor (Office 2010).

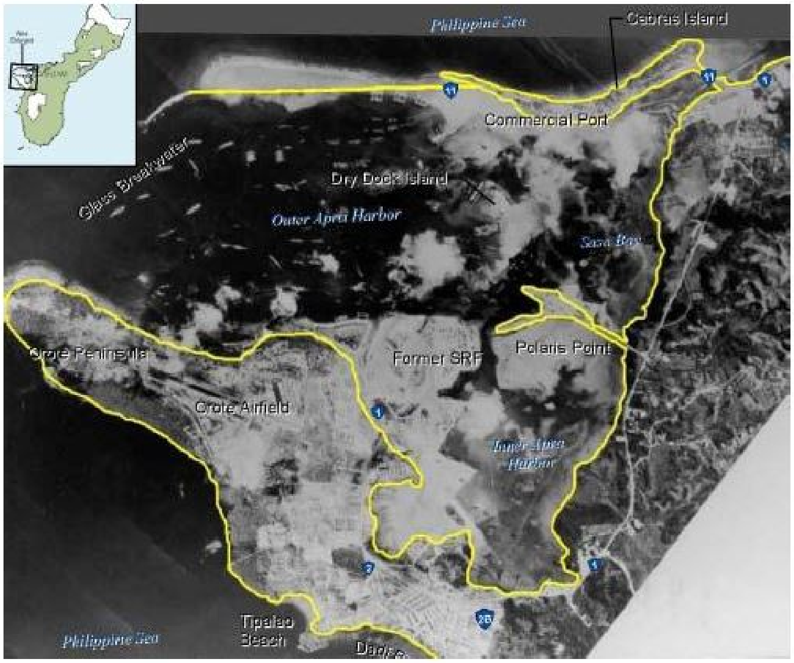

Figure 20. Apra Harbor July 1945. The yellow line indicates the approximate shoreline prior to the dredging and fill projects of the 1940s.

5.8 Commercial and Private Marine Mammal Watching

The ‘whale watching’ industry in the action area remains relatively small and is presently focused on dolphins rather than large whales. 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. Another concern is that preferred habitats may be abandoned if disturbance levels are too high. Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS 2006b). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity (Au and Green 2000; Corkeron 1995; Erbe 2002b; Magalhaes et al. 2002; Richter et al. 2003; Scheidat et al. 2004; Watkins 1986; Williams et al. 2002a; Williams et al. 2002b). Foote et al. (2004) reported that southern resident killer whale call duration in the presence of whale watching boats increased by 10 to 15 percent between 1989 to 1992 and 2001 to 2003 and suggested this indicated compensation for a noisier environment. Disturbance by whale watch vessels has also been noted to cause newborn calves to separate briefly from their mothers' sides, which leads to greater energy expenditures by the calves (NMFS 2006b). Whale-watching vessels are also known to influence sperm whale behavior

(Richter et al. 2006). 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 2006b). 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 Guam or CNMI.

5.9 Vessel Strike

Vessel strike is a significant concern for the recovery of listed whales and sea turtles. Evidence suggests that not all whales killed as a result of vessel strike are detected, particularly in offshore waters, and some detected carcasses are never recovered while those that are recovered may be in advanced stages of decomposition that preclude a definitive cause of death determination (Glass et al. 2010). Therefore, it is likely that the number of documented cetacean mortalities related to ship strikes is much lower than the actual number of mortalities associated with ship strikes. However, the Navy has a policy to report all ship strikes.

Ship strikes are a poorly-studied threat to sea turtles, but have the potential to be highly significant (Work et al. 2010a). All sea turtles must surface to breathe and several species are known to bask at the surface for long periods, including loggerhead sea turtles. Although sea turtles can move rapidly, they apparently are not adept at avoiding vessels that are moving at more than 4 km/hr; most vessels move far faster than this in open water (Hazel and Gyuris 2006; Hazel et al. 2007; Work et al. 2010a). Both live and dead sea turtles are often found with deep cuts and fractures indicative of collision with a boat hull or propeller (Hazel et al. 2007). Hazel et al. (2007) suggested that green sea turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases.

Portions of the action area are heavily traveled by commercial, recreational, and government marine vessels, with several commercial ports occurring in or near the action area. In the western Pacific Ocean, four waterways used by commercial vessels link Guam and the CNMI with major ports to both the east and west (Figure 21). Guam contains one commercial port located within Apra Harbor. The Port of Guam is the largest U.S. deepwater port in the Western Pacific and handles approximately 2 million tons (1,814,369,480 kilograms [kg]) of cargo a year (Port Authority of Guam 2011). The U.S. provides some 60 percent of Guam’s imported goods, with the balance of Guam’s trade coming from the Asian and Pacific markets of Japan, Taiwan, the Philippines, Hong Kong, and—to a lesser extent—Australia, New Zealand, and the islands of Micronesia (Port Authority of Guam 2011). Apra Harbor also provides economical transshipment services from the United States, Hawaii, and East Asia to the entire western

Pacific. Most shipping lanes are located close to the coast but those that are trans-oceanic start and end to the northwest of Guam.

There are three ports within the CNMI. The Port of Rota, or Rota West Harbor, is located on the southwestern tip of the island and is classified as a very small port (World Port Source 2012a) that is mainly used for ferry boats. The Port of Tinian is described by the World Port Source as a small port offering excellent shelter, which allows relatively large vessels to dock there. The Port of Saipan is the largest and most advanced of the three ports, but is nevertheless described as a small seaport with poor shelter by the World Port Source. A number of facilities and services are available at the Port of Saipan, which transferred over 338,000 tons of cargo in 2009 (Commonwealth Ports Authority 2005; Commonwealth Ports Authority 2010).

Major commercial shipping vessels use the shipping lanes for shipping goods between Hawaii, the continental U.S., and Asia. However, there are no direct routes between Guam and the U.S.; stops are made in Asia, and usually Japan or Korea, before continuing on to either Hawaii or the continental U.S. The total number of vessels transiting through the Port of Guam has steadily decreased from 2,924 in 1995 to 1,022 in 2008 (DoN 2010a). The Port Authority of Guam estimates 635 total vessel calls, not counting naval ships, in 2013. The decrease is most pronounced in the number of barges and fishing vessels that transit through the port; however, the number of container ships has increased from a low of 103 in 2003 to a high of 165 in 2008.

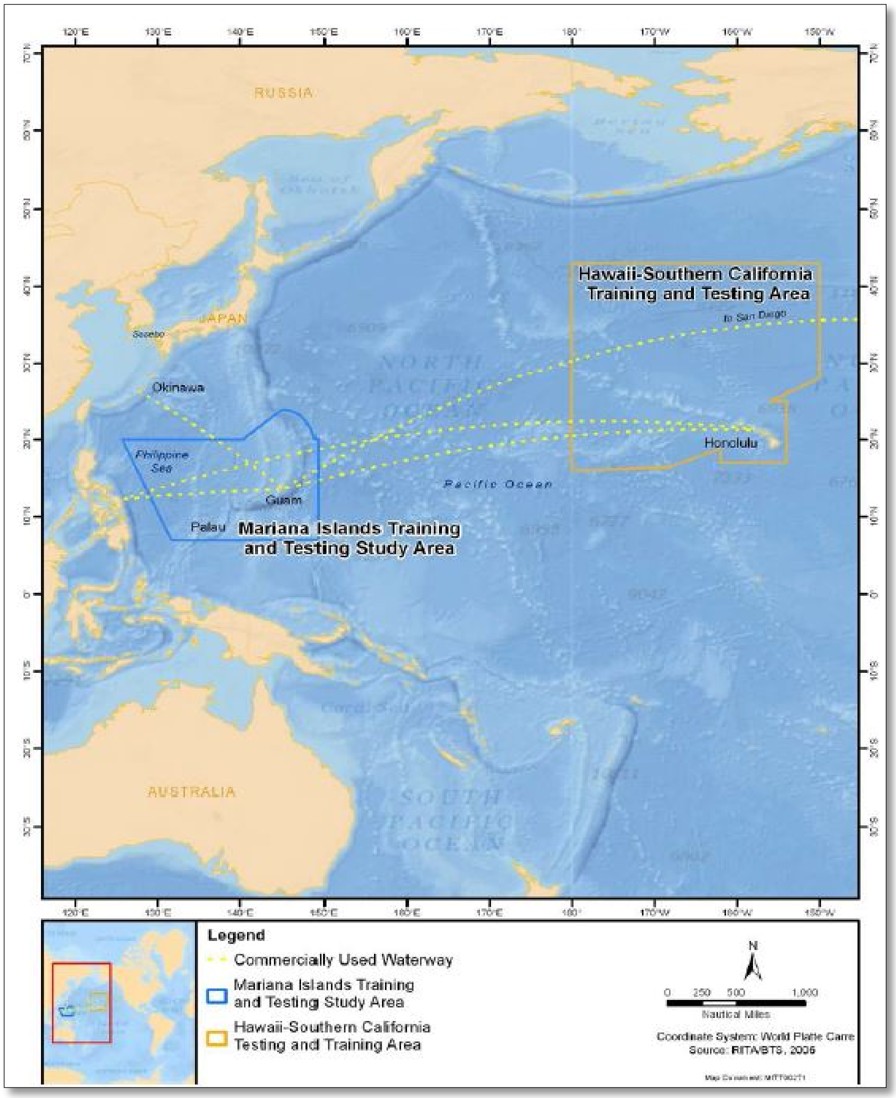

Figure 21: Shipping lanes in the action area.

The magnitude of the risks commercial ship traffic pose to large whales in the action area remains difficult to quantify or estimate. We struggle to estimate the number of whales that are killed or seriously injured in ship strikes on the U.S. Atlantic and Pacific coasts; information on ship strikes in Guam and the CNMI and in the offshore waters within the action area is virtually nonexistent. With the information available, we assume those interactions occur but we cannot estimate their significance to whale species. The Navy has been training and testing in the MITT action area for many years, and there has never been a documented case of a Navy vessel striking an ESA-listed animal.

As in the case of ship strikes to cetaceans, the information on ship strikes to sea turtles in the action area is lacking. Based on Hawaii data for the period of 1998 to 2007 (NMFS 2008), the estimated total number of green turtles killed annually by boat collisions in the Main Hawaiian

Islands (MHI) was between 25 and 50. The number of hawksbills similarly killed was much lower; between 0.2 and 0.4 turtles annually. The nearshore densities of boats and turtles are much lower around Guam and the CNMI than in Hawaiian nearshore waters, thus the number of green and hawksbill turtles killed annually by boat collisions around the Mariana Islands and surrounding waters is likely much lower. Although little information exists to quantify this impact, vessel collision has been implicated as the cause of three green turtle strandings in Apra Harbor between November 2002 and April 2008 (DAWR unpublished data).

5.10 Ocean Noise

A wide variety of anthropogenic and natural sources contribute to ocean noise throughout the world's oceans (Hatch and Wright 2007). Anthropogenic sources of noise that are most likely to contribute to increases in ocean noise are vessel noise from commercial shipping and general vessel traffic, oceanographic research, oil and gas exploration, underwater construction, and naval and other use of sound navigation and ranging.

Any potential for cumulative impact should be put into the context of recent changes to ambient sound levels in the world's oceans as a result of anthropogenic activities. However, there is a large and variable natural component to the ambient noise level as a result of events such as earthquakes, rainfall, waves breaking, and lightning hitting the ocean as well as biological noises such as those from snapping shrimp, other crustaceans, fishes, and the vocalizations of marine mammals (Crawford and Huang 1999; Patek 2002).

Andrew et al. (2002) compared ocean ambient sound from the 1960s to the 1990s from a receiver off the California coast. The data showed an increase in ambient noise of approximately 10 dB in the frequency ranges of 20–80 Hz and 200–300 Hz, and about 3 dB at 100 Hz over a 33-year period. Each 3 dB increase is noticeable to the human ear as a doubling in sound level. A possible explanation for the rise in ambient noise is the increase in shipping noise. There are approximately 11,000 supertankers worldwide, each operating approximately 300 days per year, each producing constant broadband noise at typical source levels of 198 dB (Hildebrand 2004). Generally the most energetic regularly operated sound sources are seismic airgun arrays from approximately 90 vessels with typically 12–48 individual guns per array, firing about every 10 seconds (Hildebrand 2004).

Seismic surveys are typically conducted by towing a sound source behind a research vessel, such as an airgun array that emits acoustic energy in timed intervals. The transmitted acoustic energy is reflected and received by an array of hydrophones. This acoustic information is processed to provide information about geological structure below the seafloor. The oil and gas industry conduct seismic surveys to search for new hydrocarbon deposits. In addition, research geologists conduct seismic surveys to study plate tectonics as well as other topics in marine geology. The underwater sound produced by seismic surveys could affect marine life, including ESA-listed marine species. All seismic surveys conducted by U.S. vessels are subject to the MMPA authorization process administered by the NMFS, as well as the NEPA process associated with issuing MMPA authorizations. There are no current MMPA authorizations for seismic surveys in the action area. In 2012, NMFS issued an MMPA authorization for a seismic survey in CNMI.

Noise is of particular concern to marine mammals because many species use sound as a primary sense for navigating, finding prey, avoiding predators, and communicating with other individuals. As described in greater detail later in this opinion, noise may cause marine mammals to leave a habitat, impair their ability to communicate, or to cause stress. Noise can cause behavioral disturbances, mask other sounds including their own vocalizations, may result in injury and, in some cases, may result in behaviors that ultimately lead to death. The severity of these impacts can vary greatly between minor impacts that have no real cost to the animal, to more severe impacts that may have lasting consequences. A comprehensive discussion of the potential impacts of ocean noise on listed species is included in the *Effects of the Action* section of this Opinion.

Very little data is available on ocean noise and its impacts on listed species in the action area. The extent of commercial and recreational shipping in the action area, which directly influences the extent of ocean noise in a given area, is described above. The extent of noise-producing activities associated with U.S. Navy training and testing in the action area is described in detail in the *Effects of the Action* section of this opinion.

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.

5.11 Invasive Species

Invasive species have been referred to as one of the top four threats to the world's oceans consistently ranked behind habitat degradation and alteration (Pughiuc 2010; Raaymakers 2003; Raaymakers and Hilliard 2002; Terdalkar et al. 2005; Wambiji et al. 2007). In most cases, habitat is directly affected by human alterations, as identified in the baseline section, such as hydromodification, mining, dredging, drilling, and construction. However, invasive species, facilitated by human commerce, have the ability to directly alter ecosystems upon which listed species rely.

Invasive species are a major threat to many ESA-listed species. For species listed by the USFWS, 26 percent were listed partially because of the impacts of invasive species and 7 percent were listed because invasive species were the major cause of listing (Anttila et al. 1998). Pimentel et al. (2004) found that roughly 40 percent of listed species are at risk of becoming endangered or extinct completely or in part due to invasive species, while Wilcove et al. (1998) found this to be 49 percent, with 27 percent of invertebrates, 37 percent of reptiles, 53 percent of fishes, and 57 percent of plants imperiled partly or wholly due to non-native invasions. In some regions of the world, up to 80 percent of species facing extinction are threatened by invasive species (Pimentel et al. 2004; Yan et al. 2002). Clavero and Garcia-Bertro (2005) found that invasive species were a contributing cause to over half of the extinct species in the IUCN database; invasive species were the only cited cause in 20 percent of those cases. Richter et al. (1997) identified invasive species as one of three top threats to threatened and endangered freshwater species in the U.S. as a whole.

5.11.1 Diseases

The impacts of introduced pathogens in the aquatic environment has been poorly explored and we likely know very little about the true frequency and significance of pathogen invasions (Drake et al. 2001). Pathogens have adverse effects to invertebrate communities including reef-building corals. Various species of the genus *Vibrio*, known to cause cholera in humans, white pox and white plague type II diseases in corals have been identified in ports and ballast water of vessels (Aguirremacedo et al. 2008; Anguiano-Beltrán et al. 1998; Ben-Haim and Rosenberg 2002). Oyster species have sustained several outbreaks from invasive pathogens, including *Haplosporidium nelsoni* (the cause of MSX disease, which Chesapeake Bay eastern oysters have shown 75-92 percent mortality to) and *Perkinsus marinus* (the cause of Dermo disease) in California, eastern North America, and Europe (Andrews 1984; Burreson and Ford 2004; Burreson et al. 2000; Ford and Haskin 1982; Renault et al. 2000), *Bonamia ostreae* in Europe (Ciguarria and Elston 1997; Van Banning 1987), and in the northeastern US, respectively (Ford 1996).

5.11.2 Habitat Impacts

In general, species located higher within a food web (including most ESA-listed species under NMFS' jurisdiction) are more likely to become extinct as a result of an invasion; conversely, species that are more centrally or bottom-oriented within a food web are more likely to establish (Byrnes et al. 2007; Harvey and May 1997). Propagule pressure is generally the reason for this trend, as individuals lower in the food web tend to have higher fecundity and lower survival rates (r-selection). This unbalancing of food webs makes subsequent introductions more likely as resource utilization shifts, increasing resource availability, and exploitation success by non-native species (Barko and Smart 1981; Byrnes et al. 2007). Such shifts in the base of food webs fundamentally alters predator-prey dynamics up and across food chains (Moncheva and Kamburska 2002). The number of extinction events seems to be roughly correlated with the number of invasive establishments within an area (Harvey and May 1997).

Pathogens and species with toxic effects not only have direct effects to listed species, but also may affect habitat through ecosystem-mediated impacts. There are a number of non-native species that have the potential to either expel toxins at low levels, only becoming problematic for other members of the ecosystem if their population grows to very large sizes, resulting in very large amounts of toxins being released. In other cases, pathogens introduced to an environment may affect organisms that indirectly affect listed species.

Red tide dinoflagellates have been introduced via ballast water discharges and have the potential to undergo extreme seasonal population fluctuations. During bloom conditions, high levels of neurotoxins are released into local and regional surface water and air that can cause illness and death in fishes, sea turtles, marine mammals, and invertebrates (as well as their larvae) (Hallegraeff and Bolch 1992; Hallegraeff 1998; Hamer et al. 2001; Hamer et al. 2000; Lilly et al. 2002; McMinn et al. 1997). The brown alga, *Aureococcus anophagefferens*, causes brown tide when it blooms, causing diebacks of eelgrass habitat due to blooms decreasing light availability and failure of scallops and mussels to recruit (Doblin et al. 2004).

The most commonly reported impact of non-native species in the coastal environment is competition for limited resources (Nyberg 2007). Molluscs, decapods, and aquatic plants tend to be especially capable invaders and have proven to be disruptive to food webs. The most common impacts are alteration of habitat and nutrient availability as well as altering species composition and diversity within an ecosystem (Strayer 2010). Crabs, polychaetes, and mussels can increase bioturbation and aerate the sediment (Nyberg 2007). Gastropods can alter the biogeochemical cycle through excretion of biogenic silicate in the faeces and pseudofaeces (Ragueneau et al. 2005). Molluscan invasions can also provide substrate for epibionts, shelter for benthic species, remove nutrients from the water, decrease turbidity and increase light penetration, remove sediments, and promote phytoplankton blooms by releasing nutrients from sediments (Bertness 1984; Gutierrez et al. 2003; Hecky et al. 2004). Invasives species may also prey upon ESA-listed species. For example, the crown-of-thorns sea star *Acanthaster planci* can significantly disrupt localized coral reef ecosystems by feeding on live coral (e.g., (Timmers et al. 2012)), including the ESA-listed coral considered in this opinion. At the Tanguisson Reef, Guam, a population explosion of the sea star reduced coral cover to <1% of pre-disturbance levels in 1968-1969. However, the effect of the sea star can be relatively temporary. For example, at the Tanguisson Reef, coral species richness, cover, and composition reached or exceeded measurements of comparable reefs before the disturbance within 12 years (Colgan 1987).

Invasive plants can cause widespread habitat alteration, including native plant displacement, changes in benthic and pelagic animal communities, altered sediment deposition, altered sediment characteristics, and shifts in chemical processes such as nutrient cycling (Grout et al. 1997; Ruiz et al. 1999; Wigand et al. 1997). Introduced seaweeds alter habitat by colonizing previously unvegetated areas, while algae form extensive mats that exclude most native taxa, dramatically reducing habitat complexity and the ecosystem services provided by it (Wallentinus and Nyberg 2007). Invasive algae can alter native habitats through a variety of impacts, including trapping sediment, reducing the number of suspended particles that reach the benthos for benthic suspension and deposit feeders, reduce light availability, and adversely impact foraging for a variety of animals (Britton-Simmons 2004; Gribsholt and Kristensen 2002; Levi and Francour 2004; Sanchez et al. 2005). Invasive fishes can compose a large portion of fish taxa in at least some areas, including New Zealand where 53 percent of fish taxa are exotic, Puerto Rico where invasive fish are 91 percent of the total species, and Brazil where they are 13 percent of the total (Lövei 1997).

5.12 Scientific Research and Permits

Regulations for Section 10(a)(1)(A) of the ESA allow issuance of permits authorizing take of certain ESA-listed species for the purposes of scientific research. Prior to the issuance of such a permit, the proposal must be reviewed for compliance with section 7 of the ESA. Scientific research permits issued by NMFS currently authorize studies on ESA-listed species in the Pacific Ocean, some of which occur in 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. As of August 7, 2017, there were 24 permits in the Pacific Ocean authorizing research on one or more ESA-listed whales considered in this opinion. All take authorized on ESA-listed whales is sub-lethal. Sea turtle research includes capture, handling, restraint, tagging, biopsy, blood sampling, lavage, ultrasound, and tetracycline injection. As of August 7, 2017, there were eight permits in the Pacific Ocean authorizing research on one or more of the ESA-listed sea turtles considered in this opinion. All authorized take is sub-lethal.

5.13 Conclusion on the Impact of the Baseline on Listed Resources

Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed species considered in this opinion. Some of these stressors result in mortality or serious injury to individual animals (e.g., vessel strike, whaling), whereas others result in more indirect (e.g., a fishery that impacts prey availability) or non-lethal (e.g., whale watching, anthropogenic sound) impacts. Assessing the aggregate impacts of these stressors on the species considered in this opinion is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this opinion are wide ranging and subject to stressors in locations well beyond the 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 section 4, 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. Additionally, a brief discussion of the impacts of the Environmental Baseline on Guam and FDM's coral reefs in the MITT action area follows.

The overall health of Guam's reefs has declined over time, with impacts from global and local stressors contributing to a significant decline over recent decades. The average live coral cover was approximately 50 percent in the 1960s, but dwindled to less than 25 percent by the 1990s, with only a few areas having over 50 percent live cover. The health of Guam's coral reefs varies significantly. Reefs unaffected by sediment and nutrient loading, such as those in the northern part of the island and some coastal areas in the south, have healthy coral communities. Guam's reefs have been spared from large-scale bleaching events and coral diseases which are prevalent in many parts of the world. Big Blue Reef in Apra Harbor is considered one of the healthiest reefs in the harbor due to the reef's protection from water quality factors associated with Inner Apra Harbor and ship-induced sediment resuspension that impact other reef systems in the harbor. Reefs off Dry Dock Island, which was artificially created during WWII, are considered to also be among the healthiest reefs in the harbor, primarily due to protection from stressors. In contrast, the coral reef along Polaris Point, which was also constructed during WWII, is of marginal quality and has the greatest signs of stress, including high levels of total suspended solids likely derived from watershed discharge. Recent studies have identified evidence of anchor and/or anchor chain damage to coral in Apra Harbor, including the formation of a rubble field on the southern side of the floating dry dock. Movement of mooring chains on the southern side of the floating dry dock has produced a significant rubble field, although mooring chains on the northern (outer) side of the floating dry dock do not appear to have caused similar damage.

Smith and Marx (2016) conducted underwater biological surveys at FDM between 1997 and 2012. Live coral coverage at FDM ranged between zero to greater than 60 percent in some areas. As described in section 2 of this opinion, FDM is uninhabitated, but is used during Navy training and testing activities as bombing range. Smith and Marx (2016) documented that while impacts to reef habitat did occur (i.e., from ordnance that skipped off the island, from ordnance fragments, and from an in-water detonation), no significant impacts to the physical or biological environment were detected between 1997 and 2012. Instead, the authors suggested that the benefits of restricted access to FDM because it is a bombing range have resulted in a de-facto preserve effect. They noted that marine natural resources at FDM are "comparable or superior to" those at other locations within the Mariana Archipelago.

6 EFFECTS OF THE ACTION ON SPECIES AND CRITICAL HABITAT

Section 7 regulations define “effects of the action” as the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur. This effects analyses section is organized following the stressor, exposure, response, risk assessment framework. As discussed further in section 3.3, our effects analysis relies on the term “significant.” 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).

The ESA defines “take" as “to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct.” 16 U.S.C. § 1532(19). Harm is further defined by regulation to include “an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering.” 50 C.F.R. 222.102. NMFS has not yet defined “harass” under the ESA in regulation. However, on December 21, 2016, NMFS issued interim guidance on the term “harass,” defining it as an action that “creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.” NMFS relied on this interim definition of “harass” to evaluate whether the proposed activities are likely to harass fish, sea turtle, and coral species considered in this opinion.

The number of instances of harassment of ESA-listed sea turtles from acoustic stressors is based on unprocessed exposure estimates provided by NAEMO modeling. The NAEMO model uses acoustic criteria to estimate the number of responses that could qualify as harassment. However, the modeled unprocessed exposure estimates provided by NAEMO 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.

Previous NMFS’ consultations with the Navy and NMFS (Permits and Conservation Division) regarding the effects of the Navy’s training and testing activities and NMFS’ issuance of regulations and LOAs pursuant to the MMPA have relied on outputs from NAEMO modeling to quantify instances of harassment (see, *e.g.*, NMFS’ biological opinions for Hawaii and Southern California training and testing activities, Northwest training and testing activities, 2015 opinion on Mariana Islands training and testing activities) to marine mammals from acoustic stressors (e.g., sonar, explosives). The NAEMO model uses acoustic criteria to estimate the number of responses that could qualify as Level B harassment under the MMPA.[7] Therefore, NMFS has relied on the MMPA definition of Level B harassment in estimating the number of instances of harassment of ESA-listed marine mammals in prior consultations. The Navy requested reinitiation of formal consultation for its Phase II MITT training and testing activities prior to issuance of the interim guidance. Given how far into this consultation the interim guidance was issued and the complexity associated with modeling take estimates of marine mammals, consistent with prior consultations for Navy testing and training activities, NMFS continues to rely on the MMPA definition of Level B harassment and the NAEMO model outputs to evaluate whether the proposed activities are likely to harass ESA-listed marine mammals and to estimate the number of instances of harassment of ESA-listed marine mammals considered in this opinion by acoustic stressors.

We note that as the definition of Level B harassment is currently applied, including in this opinion, a wide range of behavioral reactions may qualify as harassment, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, temporary disruption of feeding, migrating, or reproductive behaviors. The modeled estimates of Level B harassment calculated using the behavioral response function do not differentiate between the different types of potential behavioral reactions. Nor do the estimates provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. We therefore consider the available scientific evidence to determine the likely nature of the modeled behavioral responses and the potential fitness consequences for affected individuals.

For all species considered in this opinion, we rely upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

6.1 Stressors Associated with the Proposed Action

The potential stressors (risks) to ESA-listed species that we analyzed based on the training and testing activities the U.S. Navy proposes to conduct in the MITT action area are summarized in Table 29. A table of the determinations of effect for each species and each stressor is presented in Table 23.

What follows is a brief description of the stressors listed above. More information on each stressor is presented in the FEIS/OEIS, May 2015. Following the descriptions, we present the results of our exposure analyses, followed by the results of our response analyses.

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[7] For military readiness activities, Level B harassment under the MMPA means: "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered." 16 U.S.C. 1362(18)(B).

Table 29. U.S. Navy Stressor Categories Analyzed in This Opinion

Stressor	Description of Stressor
Acoustic (sonar and other active acoustic sources, underwater explosives, weapons firing, launch and impact noise, aircraft noise, and vessel noise)	Effects on species from acoustic sources are dependent on a number of factors, including the type of sound received (non-impulse or impulse), the proximity of the animal to the sound source, and the duration, frequency, and intensity of the sound. Underwater sound propagation is highly dependent upon environmental characteristics such as bathymetry, bottom type, water depth, temperature, and salinity. The sound received at a particular location will be different than near the source due to the interaction of many factors, including propagation loss; how the sound is reflected, refracted, or scattered; the potential for reverberation; and interference due to multi-path propagation. Sonar and other active acoustic sources emit sound waves into the water to detect objects, safely navigate, and communicate. Most systems operate within specific frequencies (although some harmonic frequencies may be emitted at lower sound pressure levels). Most sonar use is associated with anti-submarine warfare (ASW) activities. Sonar use associated with mine warfare (MIW) would also contribute a notable portion of overall acoustic sound. Explosives used during training and testing activities include explosive ordnance, including bombs, missiles, and naval gun shells; torpedoes; demolition charges; and explosive sonobuoys. Depending on the activity, detonations would occur in the air, near the water's surface, or underwater (some torpedoes and sonobuoys). Demolition charges could occur near the surface, in the water column, or on the seafloor. Most detonations would occur in waters greater than 200 ft. (61 m) in depth, and greater than 3 nm from shore, although MIW, demolition, and some testing detonations could occur in shallow water closer to shore. Detonations associated with ASW would typically occur in waters greater than 600 ft. (182.9 m) depth. Noise associated with weapons firing and the impact of non-explosive practice munitions (NEPM) could happen at any location within the action area but generally would occur at locations greater than 12 nm from shore for safety reasons. These training and testing events would occur in areas designated for anti-surface warfare and similar activities. The firing of a weapon may have several components of associated noise. Firing of guns could include sound generated by firing the gun (muzzle blast), vibration from the blast propagating through a ship's hull, and sonic booms generated by the projectile flying through the air. Missiles and targets would also produce noise during launch. In addition, the impact of NEPM at the water surface can introduce noise into the water. Fixed- and rotary-wing aircraft are used for a variety of training and testing activities throughout the action area, contributing both airborne and underwater sound to the ocean environment. Aircraft used in training and testing generally have reciprocating, turboprop, or jet engines. Motors, propellers, and rotors produce the most noise, with some noise contributed by aerodynamic turbulence. Aircraft sounds have more energy at lower frequencies. Takeoffs and landings occur at established airfields as well as on vessels at sea throughout the action

Stressor	Description of Stressor
	area. Most aircraft noise would be produced around air fields in the range complex. Military activities involving aircraft generally are dispersed over large expanses of open ocean but can be highly concentrated in time and location. Vessels (including ships, small craft, and submarines) would produce low-frequency, broadband underwater sound. Overall, naval traffic is often a minor component of total vessel traffic (Mintz and Filadelfo 2011) (Mintz and Parker 2006). Commercial vessel traffic, which included cargo vessels, bulk carriers, passenger vessels, and oil tankers (all over 65 ft. [20 m] in length), was heaviest near and between the major shipping ports.
Energy **(electromagnetic devices)**	Electromagnetic devices are used in towed or unmanned MIW systems that mimic the electromagnetic signature of a vessel passing through the water. None of the devices include any type of electromagnetic "pulse." The devices work by emitting an electromagnetic field and mechanically generated underwater sound to simulate the presence of a ship. The sound and electromagnetic signature cause nearby mines to detonate. The static magnetic field generated by the electromagnetic devices is of relatively minute strength. Typically, the maximum magnetic field generated would be approximately 23 gauss (G). By comparison, magnetic field generated by a refrigerator magnet is between 150 and 200 G. The strength of an electromagnetic field decreases quickly with distance from the device. The magnetic field generated at a distance of 4 m from the source is comparable to the earth's magnetic field, which is approximately 0.5 G.
Physical disturbance and strike **(vessels, in water devices, military expended materials)**	Physical disturbances, including direct strikes on marine animals, may occur in association with vessel movements, the use of in-water devices, and materials expended from vessels and aircraft. Vessels used as part of the action include ships (e.g., aircraft carriers, surface combatants), support craft, small boats, and submarines, ranging in size from 5 to over 300 m. Large Navy ships generally operate at speeds in the range of 10–15 knots, and submarines generally operate at speeds in the range of 8–13 knots. Small craft (for purposes of this discussion, less than 40 ft. [12 m] in length), which are all support craft, have variable speeds. Locations of vessel use in the action area varies with the type of activity taking place, but greater activity would be expected near ports than in other areas of the action area. In-water devices as discussed in this analysis are unmanned vehicles, such as remotely operated vehicles, unmanned surface vehicles and unmanned undersea vehicles, and towed devices. These devices are self-propelled and unmanned or towed through the water from a variety of platforms, including helicopters and surface ships. In-water devices are generally smaller than most participating vessels ranging from several inches to about 15 m. These devices can operate anywhere from the water surface to the benthic zone. Certain devices do not have a realistic potential to strike marine animals because they either move slowly through the water column (e.g., most unmanned undersea vehicles) or are closely monitored by observers manning the towing platform (e.g., most towed devices).

Stressor	Description of Stressor
	Military expended materials include: (1) all sizes of NEPM; (2) fragments from explosive munitions; and (3) expended materials other than munitions, such as sonobuoys, ship hulks, and expendable targets. Activities using NEPM (e.g., small-, medium-, and large-caliber gun ammunitions, missiles, rockets, bombs, torpedoes, and neutralizers), explosive munitions (generating munitions fragments), and materials other than munitions (e.g., flares, chaff, sonobuoys, decelerators/parachutes, aircraft stores and ballast, and targets) have the potential to contribute to the physical disturbance and strike stressor.
Entanglement **(fiber optic cables and guidance wires, and decelerators/parachutes)**	The only type of cable expended during training and testing are fiber optic cables. Fiber optic cables are flexible, durable, and abrasion or chemical-resistant. The physical characteristics of the fiber optic material render the cable brittle and easily broken when kinked, twisted, or bent sharply (i.e., to a radius greater than 360 degrees). The fiber optic cable would be suspended within the water column during the activity, and then be expended to sink to the sea floor. The only types of wires expended during training and testing activities are guidance wires from heavy-weight torpedoes. Guidance wires are used to help the firing platform control and steer the torpedo. They trail behind the torpedo as it moves through the water or air. Finally, the guidance wire is released from both the firing platform and the torpedo then sinks to the ocean floor. The torpedo guidance wire is a single-strand, thin gauge, coated copper alloy. The tensile breaking strength of the wire is a maximum of 42 pounds (lb.) (19 kilograms [kg]) and can be broken by hand (Group 2005). The length of wire dispensed would generally be equal to the distance the torpedo travels to impact the target and any undispensed wire would be contained in the dispensers upon impact. Degradation rates for the wire may vary because of changing environmental conditions in seawater, but are likely to take between 12 and 45 months Aircraft-launched sonobuoys, lightweight torpedoes (such as the MK 46 and MK 54), illumination flares, and targets use nylon parachutes or decelerators ranging in size from 18 to 48 in. (46 to 122 cm) in diameter. Decelerators are made of cloth and nylon, and many have weights attached to the lines for rapid sinking. At water impact, the decelerator assembly is expended, and it sinks away from the unit. The decelerator assembly may remain at the surface for 5–15 seconds before the decelerator and its housing sink to the seafloor, where it becomes flattened (Group 2005). Some decelerators are weighted with metal clips that facilitate their descent to the seafloor. Once settled on the bottom the canopy may temporarily billow if bottom currents are present.

Stressor	Description of Stressor
Ingestion **(munitions and military expended materials other than munitions)**	The only munitions small enough for a marine mammal or sea turtles to ingest are small- and medium-caliber projectiles. These projectiles include all sizes up to and including 2.25 in. (57 mm) in diameter. Projectiles are composed of solid metal materials and would quickly move through the water column and settle on the seafloor where they are most likely to be encountered by bottom foraging animals. Sinking projectiles are unlikely to be encountered in the water column by marine mammals or sea turtles. Many different types of explosive munitions can result in fragments that are expended at sea during training and testing activities. Types of explosive munitions that can result in fragments include demolition charges, grenades, projectiles, missiles, and bombs. Fragments would result from fractures in the munitions casing and would vary in size depending on the size of the net explosive weight and munition type; however, typical sizes of fragments are unknown. These solid metal materials would quickly sink through the water column and settle to the seafloor. Military expended materials other than munitions include target fragments, chaff, and flares. At-sea targets are usually remotely operated airborne, surface, or subsurface traveling units, most of which, but not all, are designed to be recovered for re-use. However, if they are used during activities that utilize explosives then they may result in fragments. Expendable targets that may result in fragments would include air-launched decoys, surface targets (such as marine markers, paraflares, cardboard boxes, and 10 ft. [3.05 m] diameter red balloons), and mine shapes. Most target fragments would sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time Chaff consists of reflective, aluminum-coated glass fibers used to obscure ships and aircraft from radar-guided systems. Chaff, which is stored in canisters, is either dispensed from aircraft or fired into the air from the decks of surface ships when an attack is imminent. The glass fibers create a radar cloud that mask the position of the ship or aircraft. Chaff is composed of an aluminum alloy coating on glass fibers of silicon dioxide (Force 1997). Chaff is released or dispensed in cartridges or projectiles that contain millions of fibers. When deployed, a diffuse cloud of fibers is formed that is undetectable to the human eye. Chaff is a very light material, similar to fine human hair. It can remain suspended in air anywhere from 10 minutes to 10 hours and can travel considerable distances from its release point, depending on prevailing atmospheric conditions (Arfsten et al. 2002; Force 1997). Chaff cartridge plastic end caps and pistons would also be released into the marine environment, where they would persist for long periods and could be ingested by marine animals. Chaff end caps and pistons sink in saltwater (Spargo 2007). Flares are pyrotechnic devices used to defend against heat-seeking missiles, where the missile seeks out the heat signature from the flare rather than the aircraft's engines. Similar to chaff, flares are also dispensed from aircraft and fired from ships. The flare device consists of a cylindrical cartridge approximately 1.4 in. (3.6 cm) in diameter and 5.8 in. (14.7 cm) in length. Flares are designed to

Stressor	Description of Stressor
	burn completely. The only material that would enter the water would be a small, round, plastic end cap (approximately 1.4 in. [3.6 cm] in diameter).
Secondary **(explosion byproducts, metals, chemicals, sedimentation, and transmission of marine mammal diseases and parasites)**	Secondary stressors associated with some training and testing activities could pose indirect impacts to ESA-listed marine species through habitat degradation or alteration or an effect on prey availability. Secondary stressors include (1) explosives, (2) explosion byproducts and unexploded ordnance, (3) metals, (4) chemicals, and (5) transmission of marine mammal diseases and parasites. In addition to directly impacting marine species, underwater explosions could impact other species in the food web, including prey species that ESA-listed marine species feed upon. The impacts of explosions would differ depending upon the type of prey species in the area of the detonation. Indirect impacts of explosives and unexploded ordnance to marine species via degradation of sediment or water quality is possible in the immediate vicinity of the ordnance. Explosion byproducts are not toxic to marine organisms at realistic exposure levels (Rosen and Lotufo 2010). Relatively low solubility of most explosives and their degradation products means that concentrations of these contaminants in the marine environment are relatively low and readily diluted. Metals are introduced into seawater and sediments as a result of training and testing activities involving ship hulks, targets, ordnance, munitions, and other military expended materials. Several training and testing activities introduce potentially harmful chemicals into the marine environment; principally, flares and propellants for rockets, missiles, and torpedoes. Properly functioning flares missiles, rockets, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures allow propellants and their degradation products to be released into the marine environment. The greatest risk to marine species would be from perchlorate released from flares, missile, and rockets that operationally fail. Perchlorate is highly soluble in water, persistent, and impacts metabolic processes in many plants and animals. The U.S. 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. When deployed,

Stressor	Description of Stressor
	the animals are part of what the Navy refers to as Marine Mammal Systems. These Marine Mammal Systems include one or more motorized small boats, several crew members, and a trained marine mammal. Based on the standard operating procedures with which these systems are deployed, it is not reasonably foreseeable that use of these marine mammals systems would result in the transmission of disease or parasites to ESA-listed cetacea in the action area.

6.2 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.

6.2.1 Direct Injury of Marine Mammals From Acoustic Stressors

The potential for direct injury of marine mammals has been inferred from terrestrial mammal experiments and from post-mortem examination of marine mammals believed to have been exposed to underwater explosions (Ketten et al. 1993; Richmond et al. 1973; Yelverton et al. 1973). Additionally, non-injurious effects on marine mammals (e.g., TTS) are extrapolated to injurious effects (e.g., PTS) based on data from terrestrial mammals to derive the criteria serving as the potential for injury (Southall et al. 2007b). Actual effects on marine mammals may differ from terrestrial animals due to anatomical and physiological adaptations to the marine environment, such as a reinforced trachea and flexible thoracic cavity (Ridgway and Dailey 1972) that may decrease the risk of lung injury.

Potential non-auditory direct injury from non-impulsive sound sources, such as sonar, is unlikely due to relatively lower peak pressures and slower rise times than potentially injurious impulsive sources such as explosives. Non-impulsive sources also lack the strong shock waves associated with explosions. Therefore, primary blast injury and barotrauma (i.e., injuries caused by large pressure changes; discussed below) would not occur from exposure to non-impulsive sources such as sonar. Further, though there have been marine mammal strandings associated with use of sonar, as Ketten (2012) has recently summarized, "to date, there has been no demonstrable evidence of acute, traumatic, disruptive, or profound auditory damage in any marine mammal as the result [of] anthropogenic noise exposures, including sonar." The theories of sonar induced acoustic resonance and sonar induced bubble formation are discussed below. These phenomena, if they were to occur, would require the co-occurrence of a precise set of circumstances that are unlikely to occur.

6.2.2 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue effects is primary blast injury and barotrauma after exposure to high amplitude impulsive sources, such as explosions. Primary blast injuries result from the initial compression of a body exposed to a blast wave. Primary blast injury is usually limited to gas-containing structures (e.g., lung and gut) and the auditory system (Craig Jr. 2001a; Craig Jr. and Hearn 1998). Barotrauma refers to injuries caused when large pressure changes occur across tissue interfaces, normally at the boundaries of air-filled tissues such as the lungs. Primary blast injury to the respiratory system may be fatal depending upon the severity of the trauma. Rupture of the lung may introduce air into the vascular system, producing air emboli that can restrict oxygen delivery to the brain or heart. Though often secondary in life-threatening severity to pulmonary blast trauma, the gastrointestinal tract can also suffer contusions and lacerations from blast exposure, particularly in air-containing regions of the tract. Although hemorrhage of solid organs (e.g., liver, spleen, and kidney) from blast exposure is possible, rupture of these organs is rarely encountered.

The only known occurrence of mortality or injury to a marine mammal due to 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 the MITT Final EIS/OEIS for more information on the topic of stranding. Since this incident, the Navy, in consultation with NMFS, developed additional mitigation measures to minimize the potential for similar incidents in the future.

6.2.3 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).

6.2.4 Acoustic Resonance

Acoustic resonance has been proposed as a hypothesis suggesting that acoustically induced vibrations (sound) from sonar or sources with similar operating characteristics could be damaging tissues of marine mammals. In 2002, NMFS convened a panel of government and private scientists to investigate the issue (NMFS 2002). They modeled and evaluated the likelihood that Navy mid-frequency sonar caused resonance effects in beaked whales that eventually led to their stranding in the Bahamas (DoN 2015). 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.

6.2.5 Bubble Formation (Acoustically Induced)

A suggested cause of injury to marine mammals is rectified diffusion (Crum and Mao 1996), the process of increasing the size of a bubble by exposing it to a sound field. The process is dependent upon a number of factors including the sound pressure level and duration. Under this hypothesis, one of three things could happen: (1) bubbles grow to the extent that tissue hemorrhage (injury) occurs, (2) bubbles develop to the extent an immune response is triggered or nervous tissue is subjected to enough localized pressure that pain or dysfunction occurs (a stress response without injury), or (3) the bubbles are cleared by the lung without negative consequence to the animal. The probability of rectified diffusion, or any other indirect tissue effect, will necessarily be based upon what is known about the specific process involved. Rectified diffusion is more likely if the environment in which the ensonified bubbles exist is supersaturated with gas. Repetitive diving by marine mammals can cause the blood and some tissues to accumulate gas to a greater degree than is supported by the surrounding environmental pressure (Ridgway and Howard 1979). The dive patterns of some marine mammals (e.g., beaked whales) are theoretically predicted to induce greater supersaturation (Houser 2010; Houser et al. 2001b). If rectified diffusion were possible in marine mammals exposed to high-level sound, conditions of tissue supersaturation could theoretically speed the rate of bubble growth and increase the size of the bubbles. Subsequent effects due to tissue trauma and emboli would presumably mirror those observed in humans suffering from decompression sickness. It is unlikely the short duration of sonar or explosion sounds would be long enough to drive bubble growth to any substantial size, if such a phenomenon occurs.

An alternative but related hypothesis has also been suggested: stable microbubbles could be destabilized by high-level sound exposures such that bubble growth then occurs through static diffusion of gas out of the tissues. In such a scenario, the marine mammal would need to be in a gas-supersaturated state for enough time for bubbles to become a problematic size. Recent research with *ex vivo* supersaturated bovine tissues suggested that for a 37 kHz signal, a sound exposure of approximately 215 dB re 1 μPa would be required before microbubbles became destabilized and grew (Crum et al. 2005). Assuming spherical spreading loss and a nominal sonar source level of 235 dB re 1 μPa at 1 m, a whale would need to be within 10 m (33 ft.) of the sonar dome to be exposed to such sound levels. Furthermore, tissues in the study were supersaturated by exposing them to pressures of 400 to 700 kilopascals for hours and then releasing them to ambient pressures. Assuming the equilibration of gases with the tissues occurred when the tissues were exposed to the high pressures, levels of supersaturation in the tissues could have been as high as 400 to 700 percent. These levels of tissue supersaturation are substantially higher than model predictions for marine mammals (Houser et al. 2001a; Saunders et al. 2008). It is improbable this mechanism is responsible for stranding events or traumas associated with beaked whale strandings. Both the degree of supersaturation and exposure levels observed to cause microbubble destabilization are unlikely to occur.

There is considerable disagreement among scientists as to the likelihood of this phenomenon (Evans and Miller 2004; Piantadosi and Thalmann 2004). Although it has been argued that traumas from recent beaked whale strandings are consistent with gas emboli and bubble-induced tissue separations (Fernandez et al. 2005a; Jepson et al. 2003), bubble formation as the cause of the traumas has not been verified. The presence of bubbles postmortem, particularly after decompression, is not necessarily indicative of bubble pathology (Bernaldo de Quiros et al. 2012; Dennison et al. 2011; Moore et al. 2009). Prior experimental work has also demonstrated the post-mortem presence of bubbles following decompression in laboratory animals can occur as a result of invasive investigative procedures (Stock et al. 1980).

6.2.6 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 (Hooker et al. 2012; Saunders et al. 2008). It is speculated if exposure to a startling sound elicits a rapid ascent to the surface, tissue gas saturation sufficient for the evolution of nitrogen bubbles might result (Fernandez et al. 2005a; Hooker et al. 2012; Jepson et al. 2003). In this scenario, the rate of ascent would need to be sufficiently rapid to compromise behavioral or physiological protections against nitrogen bubble formation.

Previous modeling suggests even unrealistically rapid rates of ascent from normal dive behaviors are unlikely to result in supersaturation to the extent bubble formation would be expected in beaked whales (Zimmer and Tyack 2007). Tyack et al. (2006) suggested emboli observed in animals exposed to mid-frequency active (MFA) sonar (Fernandez et al. 2005a; Jepson et al. 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth at which lung collapse would occur. A bottlenose dolphin was trained to repetitively dive to to elevate nitrogen saturation to the point that asymptomatic nitrogen bubble formation was predicted to occur. However, inspection of the vascular system of the dolphin via ultrasound did not demonstrate the formation of any nitrogen gas bubbles (Houser 2010).

More recently, modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of tissues (e.g., fat, bone lipid) to the point that they are supersaturated when the animals are at the surface (Hooker et al. 2009; Saunders et al. 2008). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al. 2006; Hooker et al. 2009). Since bubble formation is facilitated by compromised blood flow, it has been suggested that rapid stranding may lead to bubble formation in animals with supersaturated tissues because of the stress of stranding and the cardiovascular collapse that can accompany it (Houser 2010).

A fat embolic syndrome was identified by Fernandez et al. (2005b) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals, and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream. Recently, Dennison et al. (2011) reported on investigations of dolphins stranded in 2009 and 2010.Using ultrasound the authors identified gas bubbles in kidneys from 21 of 22 live-stranded dolphins and in the liver from two of 22. The authors postulated stranded animals are unable to recompress by diving, and thus may retain bubbles that are otherwise re-absorbed in animals that can continue to dive. The researchers concluded minor bubble formation can be tolerated since the majority of stranded dolphins released did not re-strand (Dennison et al. 2011). Recent modeling by Kvadsheim (2012) determined behavioral and physiological responses to sonar have the potential to result in bubble formation. However, the observed behavioral responses of cetaceans to sonar do not imply any significantly increased risk over what may otherwise occur normally in individual marine mammals. As a result, no marine mammals addressed in this analysis are given differential treatment due to the possibility for acoustically mediated bubble growth.

6.2.7 Hearing Loss

The most familiar effect of exposure to high intensity sound is hearing loss, meaning an increase in the hearing threshold. Both auditory injury and auditory fatigue may result in hearing loss. The meaning of the term “hearing loss” does not equate to “deafness.” Hearing loss is a noise-induced threshold shift, or simply a threshold shift. If high-intensity sound over stimulates tissues in the ear, causing a threshold shift, the impacted area of the ear (associated with and limited by the sound’s frequency band) no longer provides the same auditory impulses to the brain as before the exposure (Ketten 2012). The distinction between PTS and TTS is based on whether there is a complete recovery of a threshold shift following a sound exposure. If the threshold shift eventually returns to zero (the threshold returns to the pre-exposure value), the threshold shift is a TTS.

For TTS, full recovery of the hearing loss (to the pre-exposure threshold) has been determined from studies of marine mammals, and this recovery occurs within minutes to hours for the small amounts of TTS that have been experimentally induced (Finneran et al. 2005; Finneran and Schlundt 2010; Nachtigall et al. 2004). The recovery time is related to the exposure duration, sound exposure level, and the magnitude of the threshold shift, with larger threshold shifts and longer exposure durations requiring longer recovery times (Finneran et al. 2005; Finneran and Schlundt 2010; Mooney et al. 2009a; Mooney et al. 2009b). In some cases, threshold shifts as large as 50 dB (loss in sensitivity) have been temporary, although recovery sometimes required as much as 30 days (Ketten 2012). If the threshold shift does not return to zero but leaves some finite amount of threshold shift, then that remaining threshold shift is a PTS. Again for clarity, PTS, as discussed in this document, is not the complete loss of hearing, but instead is the loss of hearing sensitivity over a particular range of frequency. Figure 22 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).

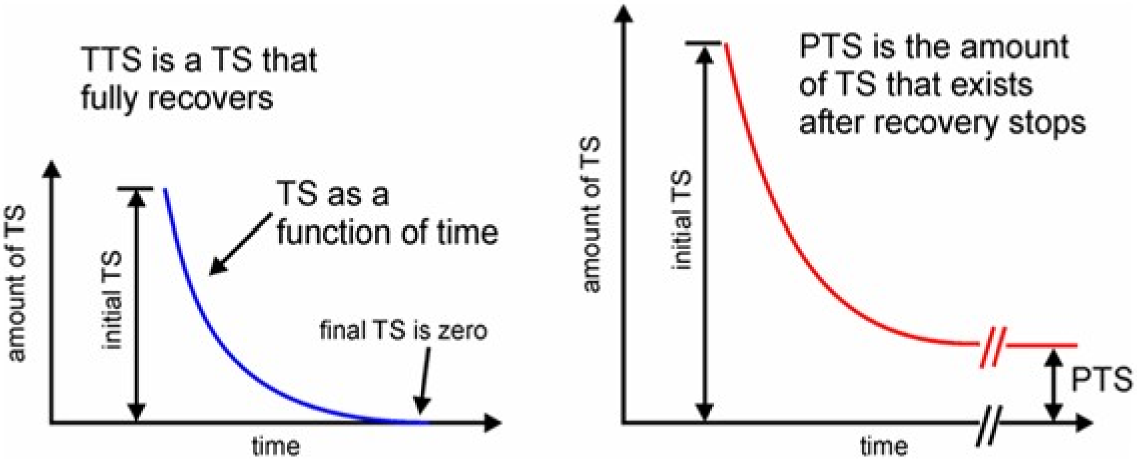

Figure 22. Two Hypothetical Threshold Shifts, Temporary and Permanent

Hearing loss, or auditory fatigue, in marine mammals has been studied by a number of investigators (Finneran et al. 2005; Finneran and Schlundt 2010; Finneran et al. 2007; Finneran et al. 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. 2014).

- Sound exposure level is correlated with the amount of TTS and is a good predictor for onset-TTS from single, continuous exposures with similar durations. This agrees with human TTS data presented by (Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b). However, for longer duration sounds beyond 16 to 32 seconds, the relationship between TTS and sound exposure level breaks down and duration becomes a more important contributor to TTS (Finneran and Schlundt 2010). Still, for a wide range of exposure durations, wound exposure level correlates reasonably well to TTS growth (Popov et al. 2014).

- The maximum TTS after tonal exposures occurs one-half to one octave above the exposure frequency (Finneran et al. 2007; Schlundt et al. 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 6.8.3.2.

6.2.8 Auditory Masking

Auditory masking occurs when a sound, or noise in general, limits the perception of another sound. As with hearing loss, auditory masking can effectively limit the distance over which a marine mammal can communicate, detect biologically relevant sounds, and echolocate (odontocetes). Unlike hearing loss, which likely results in a behavioral stress response, behavioral changes resulting from auditory masking may not be coupled with a stress response. Another important distinction between masking and hearing loss is that masking only occurs in the presence of the sound stimulus, whereas hearing loss can persist after the stimulus is gone.

Critical ratios, a measure of the relative ability of an animal to extract signals from noise, have been determined for pinnipeds (Southall et al. 2000; Southall et al. 2003) and bottlenose dolphins (Johnson 1967) and detections of signals under varying masking conditions have been determined for active echolocation and passive listening tasks in odontocetes (Au and Pawloski 1989a; Au and Pawloski 1989b; Branstetter 2013; Erbe 2000; Johnson 1971). These studies provide baseline information from which the probability of masking can be estimated.

Clark et al. (2009) developed a methodology for estimating masking effects on communication signals for low frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, their technique calculates that in Stellwagen Bank National Marine Sanctuary, when two commercial vessels pass through a North Atlantic right whale’s (a baleen whale like blue, fin, sei, and humpback whales) optimal communication space (estimated as a sphere of water with a diameter of 20 km), that space is decreased by 84 percent. This methodology relies on empirical data on source levels of calls (which is unknown for many species), and requires many assumptions about ancient ambient noise conditions and simplifications of animal behavior. However, it is an important step in determining the impact of anthropogenic noise on animal communication. Subsequent research for the same species and location estimated that an average of 63 to 67 percent of North Atlantic right whale’s communication space has been reduced by an increase in ambient noise levels, and that noise associated with transiting vessels is a major contributor to the increase in ambient noise (Hatch et al. 2012).

Vocal changes in response to anthropogenic noise can occur across sounds produced by marine mammals, such as whistling, echolocation click production, calling, and singing. Changes to vocal behavior and call structure may result from a need to compensate for an increase in background noise. In cetaceans, vocalization changes have been reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying. Vocalizations may also change in response to variation in the natural acoustic environment (e.g., from variation in sea surface motion)(Dunlop et al. 2014).

In the presence of low frequency active sonar, humpback whales have been observed to increase the length of their songs (Fristrup et al. 2003; Miller et al. 2000), possibly due to the overlap in frequencies between the whale song and the low frequency active sonar. North Atlantic right whales have been observed to increase the frequency and amplitude (intensity) (Parks 2009) of their calls while reducing the rate of calling in areas of increased anthropogenic noise (Parks et al. 2007). In contrast, both sperm and pilot whales potentially ceased sound production during experimental sound exposure (Bowles et al. 1994), although it cannot be absolutely determined whether the inability to acoustically detect the animals was due to the cessation of sound production or the displacement of animals from the area.

Different vocal responses in marine mammals have been documented in the presence of seismic survey noise. An overall decrease in vocalization during active surveying has been noted in large marine mammal groups (Potter et al. 2007). In contrast, blue whale feeding and social calls increased when seismic exploration was underway (Di Lorio and Clark 2010), indicative of a potentially compensatory response to the increased noise level. (Melcon et al. 2012) recently documented that blue whales decreased the proportion of time spent producing certain types of calls when simulated mid-frequency sonar was present. At present it is not known if these changes in vocal behavior corresponded to changes in foraging or any other behaviors. Controlled exposure experiments in 2007 and 2008 in the Bahamas recorded responses of false killer whales, short-finned pilot whales, and melon-headed whales to simulated MFA sonar (Deruiter et al. 2013a). The responses to exposures between species were variable. After hearing each MFA signal, false killer whales were found to “increase their whistle production rate and made more-MFA-like whistles” (Deruiter et al. 2013a). In contrast, melon-headed whales had “minor transient silencing” after each MFA signal, while pilot whales had no apparent response. Consistent with the findings of other previous research (see, for example, (Southall et al. 2007b)), Deruiter et al. (2013a) found the responses were variable by species and with the context of the sound exposure.

Evidence suggests that at least some marine mammals have the ability to acoustically identify predators. For example, harbor seals that reside in the coastal waters off British Columbia are frequently targeted by certain groups of killer whales, but not others. The seals discriminate between the calls of threatening and non-threatening killer whales (Deecke et al. 2002), a capability that should increase survivorship while reducing the energy required for attending to and responding to all killer whale calls. Auditory masking may prevent marine mammals from responding to the acoustic cues produced by their predators. The effects of auditory masking on the predator-prey relationship depends on the duration of the masking and the likelihood of encountering a predator during the time that predator cues are impeded.

6.2.9 Physiological Stress

Marine mammals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to diseases and naturally occurring toxins, lack of prey availability, social interactions with members of the same species, and interactions with predators all contribute to the stress a marine mammal experiences. In some cases, naturally occurring stressors can have profound impacts on marine mammals; for example, chronic stress, as observed in stranded animals with long-term debilitating conditions (e.g., disease), has been demonstrated to result in an increased size of the adrenal glands and an increase in the number of epinephrine-producing cells (Clark et al. 2006).

Anthropogenic activities have the potential to provide additional stressors above and beyond those that occur naturally. For example, various efforts have investigated the impact of vessels on marine mammals (both whale-watching and general vessel traffic noise) and demonstrated that impacts do occur (Bain 2002; Erbe 2002b; Noren et al. 2009; Pirotta et al. 2015; Williams and Ashe 2006; Williams and Noren 2009). In an analysis of energy costs to killer whales, Williams et al. (2009) suggested that whale-watching in the Johnstone Strait resulted in lost feeding opportunities due to vessel disturbance. Ayres et al. (2012) measured fecal hormones of southern resident killer whales in the Salish Sea to assess the lack of prey (salmon) and vessel traffic on species recovery. Ayres et al. (2012) suggested that the lack of prey overshadowed any population-level physiological impacts on southern resident killer whales from vessel traffic.

Marine mammals may exhibit a behavioral response or combinations of behavioral responses upon exposure to anthropogenic sounds. If a sound is detected by a marine mammal, a stress response (e.g., startle or annoyance) or a cueing response (based on a past stressful experience) can occur. Although preliminary because of the small numbers of samples collected, different types of sounds have been shown to produce variable stress responses in marine mammals. Belugas demonstrated no catecholamine (hormones released in situations of stress) response to the playback of oil drilling sounds (Thomas et al. 1990) but showed an increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al. 2004). A bottlenose dolphin exposed to the same seismic water gun signals did not demonstrate a catecholamine response, but did demonstrate an elevation in aldosterone, a hormone that may be a significant indicator of stress in odontocetes (St. Aubin et al. 2001; St. Aubin and Geraci 1989). Increases in heart rate were observed in bottlenose dolphins to which conspecific calls were played, although no increase in heart rate was observed when tank noise was played back (Miksis et al. 2001). Collectively, these results suggest a variable response that depends on the characteristics of the received signal and prior experience with the received signal.

Other types of stressors include the presence of vessels, fishery interactions, acts of pursuit and capture, the act of stranding, and pollution. In contrast to the limited amount of work performed on stress responses resulting from sound exposure, a considerably larger body of work exists on stress responses associated with pursuit, capture, handling and stranding. A study compared pathological changes in organs/tissues of odontocetes stranded on beaches or captured in nets over a 40-year period (Cowan and Curry 2008). The type of changes observed indicate multisystemic harm caused in part by an overload of catecholamines into the system, as well as a restriction in blood supply capable of causing tissue damage or tissue death. This response to a stressor or stressors is thought be mediated by the over-activation of the animal's normal physiological adaptations to diving or escape. Pursuit, capture and short-term holding of belugas have been observed to result in decreased thyroid hormones (St. Aubin and Geraci 1988) and increases in epinephrine (St. Aubin and Dierauf 2001). In dolphins, the trend is more complicated with the duration of the handling time potentially contributing to the magnitude of the stress response (Ortiz and Worthy 2000; St. Aubin 2002; St. Aubin et al. 1996). Male grey seals subjected to capture and short-term restraint showed an increase in cortisol levels accompanied by an increase in testosterone (Lidgard et al. 2008). This result may be indicative of a compensatory response that enables the seal to maintain reproduction capability in spite of stress. Elephant seals demonstrate an acute cortisol response to handling, but do not demonstrate a chronic response; on the contrary, adult females demonstrate a reduction in the adrenocortical response following repetitive chemical immobilization (Engelhard et al. 2002). Similarly, no correlation between cortisol levels and heart/respiration rate changes were seen in harbor porpoises during handling for satellite tagging (Eskesen et al. 2009). Taken together, these studies illustrate the wide variations in the level of response that can occur when faced with these stressors.

Factors to consider when trying to predict a stress or cueing response include the mammal's life history stage and whether they are experienced with the stressor. Prior experience with a stressor may be of particular importance as repeated experience with a stressor may dull the stress response via acclimation (St. Aubin and Dierauf 2001).

The sound characteristics that correlate with specific stress responses in marine mammals are poorly understood. Therefore, in practice and for the purposes of this opinion, a stress response is assumed if a physical injury such as hearing loss or trauma is predicted; or if a significant behavioral response is predicted.

6.2.10 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. 1995c). More recent reviews (Ellison et al. 2012b; Nowacek et al. 2007; Southall et al. 2009) (Southall et al. 2007b) address studies conducted since 1995 and focus on observations where the received sound level of the exposed marine mammal(s) was known or could be estimated.

Except for some vocalization changes that may be compensating for auditory masking, all behavioral reactions are assumed to occur due to a preceding stress or cueing response, however stress responses cannot be predicted directly due to a lack of scientific data (see preceding section). Responses can overlap; for example, an increased respiration rate is likely to be coupled with a flight response. Differential responses are expected among and within species since hearing ranges vary across species, the behavioral ecology of individual species is unlikely to completely overlap, and individuals of the same species may react differently to the same, or similar, stressor.

Southall et al. (2007a) synthesized data from many past behavioral studies and observations to determine the likelihood of behavioral reactions to specific sound levels exposures. While in general, the louder the sound source the more intense the behavioral response, it was clear that the proximity of a sound source and the animal's experience, motivation, and conditioning were also critical factors influencing the response (Southall et al. 2007b). After examining all of the available data, the authors felt that the derivation of thresholds for behavioral response based solely on exposure level was not supported because context of the animal at the time of sound exposure was an important factor in estimating response. Nonetheless, in some conditions consistent avoidance reactions were noted at higher sound levels dependent on the marine mammal species or group allowing conclusions to be drawn. Most low-frequency cetaceans (mysticetes) observed in studies usually avoided sound sources at levels of greater than or equal to 160 dB re 1 µPa. Published studies of mid-frequency cetaceans analyzed include sperm whales, belugas, bottlenose dolphins, and river dolphins. These groups showed no clear tendency, but for non-impulsive sounds, captive animals tolerated levels in excess of 170 dB re 1 µPa before showing behavioral reactions, such as avoidance, erratic swimming, and attacking the test apparatus. High-frequency cetaceans (observed from studies with harbor porpoises) exhibited changes in respiration and avoidance behavior at levels between 90 and 140 dB re 1 µPa, with profound avoidance behavior noted for levels exceeding this. Phocid seals showed avoidance reactions at or below 190 dB re 1 µPa, thus seals may actually receive levels adequate to produce TTS before avoiding the source. Recent studies with beaked whales have shown them to be particularly sensitive to noise, with animals during 3 playbacks of sound breaking off foraging dives at levels below 142 dB re 1 µPa, although acoustic monitoring during actual sonar exercises revealed some beaked whales continuing to forage at levels up to 157 dB re 1 µPa (Tyack et al. 2011b). Passive acoustic monitoring of beaked whales, classified as Blainville's beaked whales and Cross-seamount type beaked whales, at Pacific Missile Range Facility, Kauai, Hawaii showed statistically significant differences in dive rates, diel occurrence patterns, and spatial distribution of dives after the initiation of a training event. However, for the beaked whale dives that continued to occur during MFAS activity, differences from normal dive profiles and click rates were not detected with estimated receive levels up to 137 dB re 1 uPa while the animals were at depth during their dives (Manzano-Roth et al. 2013).

6.2.10.1 *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.

6.2.10.1.1 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). While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al. 1995a), some whales avoided vessels by more than 20 km at received levels as low as 120 dB re 1 µPa root mean square. Additionally, Malme et al. (1988) observed clear changes in diving and respiration patterns in bowheads at ranges up to 73 km from seismic vessels, with received levels as low as 125 dB re 1 µPa.

Gray whales migrating along the U.S. west coast showed avoidance responses to seismic vessels at 164 dB re 1 µPa (10 percent of animals showed avoidance response), and at 190 dB re 1 µPa (90 percent of animals showed avoidance response), with similar results for whales in the Bering Sea (Malme et al. 1986; Malme et al. 1988). In contrast, noise from seismic surveys was not found to impact Western North Pacific gray whale feeding behavior or exhalation rates off the coast of Russia (Gailey et al. 2007; Yazvenko et al. 2007).

Humpback whales showed avoidance behavior at ranges of 5 to 8 km from a seismic array during observational studies in western Australia (McCauley et al. 1998; Todd et al. 1996) found no clear short-term behavioral responses by foraging humpbacks to explosions associated with construction operations in Newfoundland, but did see a trend of increased rates of net entanglement and a shift to a higher incidence of net entanglement closer to the noise source.

Seismic pulses at average received levels of 131 dB re 1 µPa^2-s caused blue whales to increase call production (Di Lorio and Clark 2010). McDonald et al. (1995) tracked a blue whale with seafloor seismometers and reported that it stopped vocalizing and changed its travel direction at a range of 10 km from the seismic vessel (estimated received level 143 dB re 1 µPa peak-to-peak). Castellote et al. (2012a) found that vocalizing fin whales in the Mediterranean left the area where a seismic survey was being conducted and that their displacement persisted beyond the completion of the survey. These studies demonstrate that even low levels of noise received far from the noise source can induce behavioral responses.

6.2.10.1.2 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).

6.2.10.2 *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.

6.2.10.2.1 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. 2001b). However, five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives, although the alarm signal was long in duration, lasting several minutes, and purposely designed to elicit a reaction from the animals as a prospective means to protect them from ship strikes (Nowacek et al. 2004). Although the animal's received sound pressure level was similar in the latter two studies (133 to 150 dB re 1 μPa), the frequency, duration, and temporal pattern of signal presentation were different. Additionally, the right whales did not respond to playbacks of either right whale social sounds or vessel noise, highlighting the importance of the sound characteristics, species differences, and individual sensitivity in producing a behavioral reaction.

Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were not found to affect dive times of humpback whales in Hawaiian waters (Frankel and Clark 2000) or to overtly affect elephant seal dives off California (Costa et al. 2003). However, they did produce subtle effects that varied in direction and degree among the individual seals, again illustrating the uncertain nature of behavioral effects and consequent difficulty in defining and predicting them.

Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007b; Stafford and Moore 2005a), preliminary results from the 2010 and 2011 field seasons of an ongoing behavioral response study in Southern California waters indicated that in some cases and at low received levels, tagged blue whales responded to mid-frequency sonar. However, those responses were mild and there was a quick return to their baseline activity (Southall et al. 2011a). Blue whales appeared to ignore sonar transmissions at received levels lower than approximately 150 dB and generally ignored received levels greater than these when they were engaged in feeding behavior (Southall et al. 2011a).

Goldbogen et al. (2013) used DTAGs to test the response of blue whales in the Southern California Bight to playbacks of simulated MFA sonar. Source levels of simulated sonar and control sounds (pseudo-random noise or PRN) in the 3.5 to 4.0 kHz range were ramped up in 3 dB increments from 160 to 210 dB re 1 µPa. Responses varied depending on the whales' prior behavioral state: surface feeding whales showed no response, while deep feeding whales and whales that were not feeding were affected. Responses among affected animals ranged from termination of deep foraging dives to prolonged mid-water dives. The reactions were context dependent, leading the authors to conclude a combination of received sound level and the behavioral state of the animal are likely to influence behavioral response. The authors note that whales responded even at low SPLs, suggesting that received level alone may not be a reliable predictor of behavior. Goldbogen et al. (2013) noted that behavioral responses observed were temporary and whales typically resumed normal behavior quickly after the cessation of sound exposure. Perhaps the most significant 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.

Melcon et al. (2012) tested whether MFA sonar and other anthropogenic noises in the mid-frequency band affected the “D-calls” produced by blue whales in the Southern California Bight. The authors used passive acoustic monitoring data recorded with stationary High-frequency Acoustic Recording Packages in the Southern California Bight. The likelihood of an animal calling decreased with the increased received level of mid-frequency sonar, beginning at a sound pressure level of approximately 110 to 120 dB re 1 µPa. Observations indicated that after sonar cessation, blue whales start producing D calls again. The authors concluded that blue whales heard and devoted attention to the sonar, despite its high frequency (relative to their putative hearing sensitivity) and its low received level. However, the authors noted that while D calls are typically associated with blue whale foraging behavior, they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior.

Martin et al. (2015) 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.

6.2.10.2.2 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; Southall et al. 2009; Tyack et al. 2011c). Reaction to mid-frequency sounds included premature cessation of clicking, termination of a foraging dive, and a slower ascent rate to the surface. Preliminary results have been presented for the behavioral response study in Southern California waters (e.g., (Boyd et al. 2008; Southall et al. 2013; Southall et al. 2012; Southall et al. 2011a).

For example, Stimpert et al. (2014) tagged a Baird’s beaked whale and exposed it to simulated mid-frequency sonar. Some changes in the animal’s dive behavior and locomotion were observed when received level reached 127 dB re 1µPa. Deruiter et al. (2013b) presented results from two Cuvier’s beaked whales that were tagged and exposed to simulated MFA sonar during the 2010 and 2011 field seasons of the southern California behavioral response study. The 2011 whale was also incidentally exposed to MFA sonar from a distant naval exercise. Received levels from the MFA sonar signals from the controlled and incidental exposures were calculated as 84 to 144 and 78 to 106 dB re 1 µPa root mean square, respectively. Both whales showed responses to the controlled exposures, ranging from initial orientation changes to avoidance responses characterized by energetic fluking and swimming away from the source. However, the authors did not detect similar responses to incidental exposure to distant naval sonar exercises at comparable received levels, indicating that context of the exposures (e.g., source proximity, controlled source ramp-up) may have been a significant factor. Cuvier's beaked whale responses suggested particular sensitivity to sound exposure as consistent with results for Blainville’s beaked whale. Similarly, beaked whales exposed to sonar during British training exercises stopped foraging (DSTL 2007).

Miller et al. (2011) reported on behavioral responses of pilot whales, killer whales, and sperm whales off Norway to Norwegian Navy mid-frequency sonar sources (a 3-year effort called the 3S experiments) (see also (Antunes et al. 2014); (Kuningas et al. 2013); (Kvadsheim et al. 2011); (Miller et al. 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 (Miller et al. 2014). Some aspects of the experiment differ from typical Navy actions and may have exacerbated observed reactions; for example, animals were directly approached by the source vessel, researchers conducted multiple approaches toward the same animal groups over the course of each session, some exposures were conducted in bathymetrically restricted areas, and, in some cases, researchers "leapfrogged" the boat to repeatedly move ahead of the animals in order to repeatedly approach animals on their travel path. For example, separation of a killer whale calf from its mother occurred during the fifth vessel approach towards a killer whale group in a fjord. In contrast, 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. 2011c). In contrast, preliminary analyses suggest that none of the pilot whales or false killer whales in the Bahamas showed an avoidance response to controlled exposure playbacks (Southall et al. 2009).

Through analysis of the behavioral response studies, a preliminary overarching effect of greater sensitivity to all anthropogenic exposures was seen in beaked whales compared to the other odontocetes studied (Southall et al. 2009). Therefore, recent studies have focused specifically on beaked whale responses to MFA sonar transmissions or controlled exposure playback of simulated sonar on various military ranges (Claridge and Durban 2009; DSTL 2007; McCarthy et al. 2011; Moretti 2009; Tyack et al. 2011c). In the Bahamas, Blainville's beaked whales located on the range will move off-range during MFA sonar use and return only after the sonar transmissions have stopped, sometimes taking several days to do so (Claridge and Durban 2009; McCarthy et al. 2011; Moretti 2009; Tyack et al. 2011c). Moretti et al. (2014) used recordings from seafloor mounted hydrophones at the Atlantic Undersea Test and Evaluation Center (AUTEC) to analyze the probability of Blainsville's beaked whale dives before, during, and after Navy sonar exercises. They developed an empirical risk function and predicted a 0.5 probability of disturbance at received levels of 150 dB.

Claridge (2013) used photo-recapture methods to estimate population abundance and demographics of Blainville's beaked whale (Mesoplodon densirostris) in the Bahamas at two sample locations; one within the bounds of the AUTEC where sonar training occurs and the second along the edge of Abaco Island approximately 170 km to the north. To investigate the potential effect of beaked whale exposure to MFA sonar, Claridge assumed that the two sample sites should have equal potential abundances and hypothesized that a lower abundance found at the AUTEC was due to either reduced prey availability at AUTEC or due to population level effects from the exposure to MFA sonar at AUTEC.

There are two major issues with this study. First, all of the re-sighted whales during the 5-year study at both sites were female. Claridge acknowledges that this can lead to a negative bias in the estimation of abundances. It has been shown in other cetacean species that females with calves may prefer "nursery" habitats or form nursery groups with other mother-calf pairs (e.g., (Claridge 2006; Scott et al. 1990; Weir et al. 2008)). It may be that the site at Abaco is a preferred site for females with calves, while the site at AUTEC is not, and therefore over the 5-year study period fewer females with calves were observed at AUTEC as these females went elsewhere in the area during the 3-year weaning period. In addition, Marques et al. (2009) estimated the Blainville's beaked whale population at AUTEC to be between 22.5 and 25.3 animals per 1,000 km^2 . This density was estimated over 6 days using passive acoustic methods, which is a method Claridge identified as one that may be better for estimating beaked whale densities than visual methods. The results at AUTEC are also biased by reduced effort and a shorter overall study period that did not capture some of the emigration/immigration trends Claridge identified at Abaco. For these reasons among others, it is unclear whether there are significant differences in the abundances between the two sites.

Second, Claridge assumed that the two sites are identical and therefore should have equal potential abundances; Abaco is a "control" site with the difference being the use of sonar at AUTEC. Although the sample boundaries at each location were drawn to create samples "of comparable size," there are differences between the two sample area locations as follows: the Abaco site is along a leeward shore, AUTEC is windward; the Abaco sample area is a long narrow margin along a canyon wall, the rectangular AUTEC sample site is a portion of a deep and landlocked U-shaped trough. In addition to the physical differences, Claridge notes that it remains unclear whether or not variation in productivity between sites influenced what she refers to as the substantial differences in abundance. Claridge reports that a study investigating prey distributions at her sample locations was unable to sample prey at the beaked whale foraging depth. Claridge dismisses the possibility of differences in prey availability between the sites noting that there is no supporting evidence that prey availability differs between the two sites. As this study illustrates, the multiple and complex factors required by investigations of potential long-term cause and effect from actions at sea require a comprehensive assessment of all factors influencing potential trends in species abundances that are not likely attributable to a single cause and effect.

In the Caribbean, research on sperm whales in 1983 coincided with the U.S. intervention in Grenada. Sperm whales interrupted their activities by stopping echolocation and leaving the area. This response was assumed to be the result of underwater sounds originating from submarine mid to high-frequency sonar signals (Watkins et al. 1985b; Watkins and Schevill 1975a). The authors did not provide any sound levels associated with these observations, although they did note getting a similar reaction from banging on their boat hull. It was unclear if the sperm whales were reacting to the sonar signal itself or to a potentially new unknown sound as had been demonstrated on another occasion during which sperm whales in the Caribbean stopped vocalizing when presented with sounds from nearby acoustic pingers (Watkins and Schevill 1975a).

Researchers at the Navy's Marine Mammal Program facility in San Diego, California have conducted a series of controlled experiments on bottlenose dolphins and beluga whales to study TTS (Finneran 2010; Finneran 2011; Finneran et al. 2001; Finneran et al. 2003; Finneran et al. 2010; Finneran and Schlundt 2004; Schlundt et al. 2000). 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. 2007b). Henderson et al. (2014) observed behavioral responses of delphinids to MFA sonar in the Southern California Bight from 2004 to 2008. The authors observed responses ranging from changes in behavioral state or direction of travel, to changes in vocalization activity. Behavioral responses were generally observed at received sound pressure levels ranging from 107 to 117 dB_{rms} re: 1 µPa. We are not reasonably certain that exposure to such sound pressure levels will elicit a substantive behavioral reaction and rise to the level of take per the ESA.

6.2.10.3 *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; Kerosky et al. 2013; Melcon et al. 2012; Richardson et al. 1995c). 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 2000; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams and Noren 2009). Noren et al. (2009) conducted research in the San Juan Islands in 2005 and 2006 and their findings suggested that close approaches by vessels impacted the whales' behavior and that the whale-watching guideline minimum approach distance of 100 m may be insufficient in preventing behavioral responses. Most studies of this type are opportunistic and have only examined the short-term response to vessel sound and vessel traffic (Magalhaes et al. 2002; Noren et al. 2009; Richardson and Wursig 1995; Watkins 1981c).

Long-term and cumulative implications of vessel sound on marine mammals remains largely unknown. Clark et al. (2009) provided a discussion on calculating the cumulative impacts of anthropogenic noise on baleen whales and estimated the noise from the passage of two vessels could reduce the optimal communication space for North Atlantic right whales by 84 percent (see also (Hatch et al. 2012)).

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

6.2.10.3.1 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 1981a). Only when vessels approached more closely did the fin whales in this study alter their behavior by increasing time at the surface and exhibiting avoidance behaviors. Other studies have shown when vessels are near, some but not all fin whales change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au and Green 2000; Castellote et al. 2012b; Richter et al. 2003; Williams et al. 2002b).

Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012b) demonstrated that fin whales' songs had shortened duration and decreased bandwidth, center frequency, and peak frequency in the presence of high shipping noise levels. It is not known if these changes in vocal behavior corresponded to other behaviors.

In the Watkins (1981a) study, humpback whales did not exhibit any avoidance behavior but did react to vessel presence. In a study of regional vessel traffic, Baker et al. (1983) found that when vessels were in the area, the respiration patterns of the humpback whales changed. The whales also exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 1.24 and 2.48 mi. (2,000 and 4,000 m) away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were within approximately 1.2 mi. (2,000 m; (Baker and Herman 1983)). Similar findings were documented for humpback whales when approached by whale watch vessels in Hawaii (Au and Green 2000).

Gende et al. (2011) reported on observations of humpback whales in inland waters of Southeast Alaska subjected to frequent cruise ship transits (i.e., in excess of 400 transits in a 4-month season in 2009). The study was focused on determining if close encounter distance was a function of vessel speed. The reported observations, however, seem in conflict with other reports of avoidance at much greater distance so it may be that humpback whales in those waters are more tolerant of vessels (given their frequency) or are engaged in behaviors, such as feeding, that they are less willing to abandon. This example again highlights that context is critical for predicting and understanding behavioral reactions as concluded by Southall et al. (2007b) and Ellison et al. (2012b).

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. 2009). 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 MITT action area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004) and therefore might provide insight to behavioral responses of other baleen whales. North Atlantic right whales continue to use habitats in high vessel traffic areas (Nowacek et al. 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Nowacek et al. 2004; Terhune and Verboom 1999). Although this may minimize potential disturbance from passing ships, it does increase the whales' vulnerability to potential ship strike. The regulated approach distance for North Atlantic right whales is 500 yards (yd.) (457 m) (NMFS 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 (Watkins 1986).

Mysticetes have been shown to both increase and decrease calling behavior in the presence of vessel noise. An increase in feeding call rates and repetition by humpback whales in Alaskan waters was associated with vessel noise (Doyle et al. 2008). Melcon et al. (2012) also recently documented that blue whales increased the proportion of time spent producing certain types of calls when vessels were present. Conversely, decreases in singing activity by humpback whales have been noted near Brazil due to boat traffic (Sousa-Lima and Clark 2008). The Central North Pacific stock of humpback whales is the focus of whale-watching activities in both its feeding grounds (Alaska) and breeding grounds (Hawaii). Regulations addressing minimum approach distances and vessel operating procedures are in place in Hawaii, however, there is still concern that whales may abandon preferred habitats if the disturbance is too high (Allen and Angliss 2010b).

6.2.10.3.2 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. (2011b) noted the result of a controlled exposure to pseudorandom noise suggests beaked whales would respond to vessel noise at similar received levels to those noted previously for mid-frequency sonar.

Most delphinids react neutrally to vessels, although both avoidance and attraction behavior is known (Hewitt 1985a; Wursig et al. 1998). Avoidance reactions include a decrease in resting behavior or change in travel direction (Bejder et al. 2006a). Incidence of attraction includes harbor porpoises approaching a vessel and common, rough-toothed, and bottlenose dolphins bow riding and jumping in the wake of a vessel (Norris and Prescott 1961; Shane et al. 1986) (Ritter 2002; Wursig et al. 1998). A study of vessel reactions by dolphin communities in the eastern tropical Pacific found that populations that were often the target of tuna purse-seine fisheries (spotted, spinner and common dolphins) show evasive behavior when approached; however populations that live closer to shore (within 100 nm; coastal spotted and bottlenose dolphins) that are not set on by purse-seine fisheries tend to be attracted to vessels (Archer et al. 2010a; Archer et al. 2010b). Pirotta et al. (2015) quantified the effect of boat disturbance on bottlenose dolphin foraging and determined that foraging activity was more affected by boat presence than noise level.

Killer whales, the largest of the delphinids, are targeted by numerous small whale-watching vessels in the Pacific Northwest. For the 2012 season, it was reported that 1,590 vessel incidents were possible violations of the federal vessel approach regulations or MMPA and ESA laws as well (Eisenhardt 2013). Research suggests that whale-watching distances may be insufficient to prevent behavioral disturbances due to vessel noise (Noren et al. 2009). In 2012, there were 79 U.S. and Canadian commercial whale watch vessels in the Haro Strait region (Eisenhardt 2013). These vessels have measured source levels that ranged from 145 to 169 dB re 1 μPa at 1 m. The sound they produce underwater has the potential to result in behavioral disturbance, interfere with communication, and affect the killer whales' hearing (Erbe 2002b). Killer whales foraged significantly less and traveled significantly more when boats were within 328 ft. (100 m) (Kruse 1991a; Lusseau et al. 2009; Trites and Bain 2000; Williams and Noren 2009; Williams et al. 2002b). These short-term feeding activity disruptions may have important long-term population-level effects (Lusseau et al. 2009; Noren et al. 2009). The reaction of the killer whales to whale-watching vessels may be in response to the vessel pursuing them, rather than to the noise of the vessel itself, or to the number of vessels in their proximity.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity (Holt et al. 2008a) as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado and Wartzok 2008). Likewise, modification of multiple vocalization parameters has been shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al. 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al. 2005). Killer whales off the northwestern coast of the United States have been observed to increase the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which has been suggested as a response to increased masking noise produced by the vessels (Foote et al. 2004). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. For example, the source level of killer whale vocalizations has been shown to increase with higher background noise levels associated with vessel traffic (Hotchkin and Parks 2013). In addition, calls with a high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al. 2008a).

6.2.10.4 *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. (1995c), 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 (Holst et al. 2011; Manci et al. 1988). Richardson et al. (1995c) 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. 1995c). These factors could include aircraft type (single engine, multi-engine, jet turbine), flight path (centered on the animal, off to one side, circling, level and slow), environmental factors such as wind speed, sea state, cloud cover, and locations where native subsistence hunting continues.

6.2.10.4.1 Mysticetes

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Efroymson et al. 2000; Koski et al. 1998). Richardson et al. (1995c) 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. 1995c). Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 492 ft. (150 m) or higher. It should be noted that bowhead whales may have more acute responses to anthropogenic activity than many other marine mammals since these animals are often presented with limited egress due to limited open water between ice floes. Additionally, many of these animals may be hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

6.2.10.4.2 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. 1995c).

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. 1992; Richter et al. 2006; Richter et al. 2003; Smultea et al. 2008; Wursig et al. 1998). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al. 1995c). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 ft. [244 to 335 m]) by moving closer together and forming a defensive fan-shaped semicircle, with their heads facing outward. Several individuals in the group turned on their sides, apparently to look up toward the aircraft (Smultea et al. 2008). Whale-watching aircraft apparently caused sperm whales to turn more sharply but did not affect blow interval, surface time, time to first click, or the frequency of aerial behavior (Richter et al. 2003). Navy aircraft do not fly at low altitude, hover over, or follow whales and so are not expected to evoke this type of response.

Smaller delphinids generally react to overflights either neutrally or with a startle response (Wursig et al. 1998). The same species that show strong avoidance behavior to vessel traffic (*Kogia* species and beaked whales) also react to aircraft (Wursig et al. 1998). Beluga whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns to a greater extent than mysticetes in the same area (Patenaude et al.

2002). These reactions increased in frequency as the altitude of the helicopter dropped below 492 ft. (150 m).

6.2.11 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 2013). For example, a typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. For sonar devices that are stationary (e.g. dipped sonar), due to the duty cycle, duration of active transmission in a specific location, and mitigation measures (e.g. avoidance of visible marine mammals), we would not expect repeated exposures.

Some individuals may be exposed to multiple sound-producing activities over a season, year, or life stage. Repeated exposure to acoustic and other anthropogenic stimuli has been studied in several cases, especially as related to vessel traffic and whale watching. Common dolphins in New Zealand responded to dolphin-watching vessels by interrupting foraging and resting bouts, and took longer to resume behaviors in the presence of the vessel (Stockin 2008). Bejder et al. (2006a) studied responses of bottlenose dolphins to vessel approaches and found shorter lasting reactions in populations exposed to higher levels of vessel traffic. The authors indicated that lesser reactions in populations of dolphins regularly subjected to high levels of vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity.

Marine mammals exposed to high levels of human activities may leave the area, habituate to the activity, or tolerate the disturbance and remain in the area. Individual marine mammals that are more tolerant may stay in a disturbed area, whereas individuals that are more sensitive may leave for areas with less human disturbance. Animals that remain throughout the disturbance may be unable to leave the area for a variety of physiological or environmental reasons. However, given the highly migratory, wide ranging life histories, and open ocean environments of the species considered in this opinion, we do not believe this will result from Navy training and testing activities in the MITT action area. Longer-term displacement can lead to changes in abundance or distribution patterns of the species in the affected region if they do not become acclimated to the presence of the sound (Bejder et al. 2006b; Blackwell et al. 2004; Teilmann et al. 2006). Gray whales in Baja California abandoned a historical breeding lagoon in the mid-1960s due to an increase in dredging and commercial shipping operations. Whales did repopulate the lagoon after shipping activities had ceased for several years (Bryant et al. 1984). Over a shorter time scale, studies on the Atlantic Undersea Test and Evaluation Center instrumented range in the Bahamas have shown that some Blaineville's beaked whales may be resident during all or part of the year in the area, and that individuals may move off of the range for several days during and following a sonar event. However animals are thought to continue feeding at short distances (a few kilometers) from the range out of the louder sound fields (less than 157 dB re 1 µPa) (McCarthy et al. 2011; Tyack et al. 2011c). Mysticetes in the northeast tended to adjust to vessel traffic over a number of years, trending towards more neutral responses to passing vessels (Watkins 1986) indicating that some animals may habituate or otherwise learn to cope with high levels of human activity. Nevertheless, the long-term consequences of these habitat utilization changes are unknown, and likely vary depending on the species, geographic areas, and the degree of acoustic or other human disturbance.

Moore and Barlow (2013) have noted a decline in beaked whales in a broad area of the Pacific Ocean area out to 300 nm from the coast and extending from the Canadian-U.S. border to the tip of Baja Mexico. There are scientific caveats and limitations to the data used for that analysis, as well as oceanographic and species assemblage changes not thoroughly addressed in Moore and Barlow (2013), although the authors suggest Navy sonar as one possible explanation for the apparent decline in beaked whale numbers over that broad area. In the small portion of the Pacific coast overlapping the Navy's SOCAL Range Complex, long-term residency by individual Cuvier's beaked whales and documented higher densities of beaked whales provide indications that the proposed decline in numbers elsewhere along the Pacific coast is not apparent where the Navy has been intensively training with sonar and other systems for decades. While it is possible that a downward trend in beaked whales may have gone unnoticed at the range complex (due to a lack of survey precision) or that beaked whale densities may have been higher before the Navy began using sonar more than 60 years ago, there is no data available to suggest that beaked whale numbers have declined on the range where Navy sonar use has routinely occurred. As Moore and Barlow (2013) point out, it remains clear that the Navy range in SOCAL continues to support high densities of beaked whales. Furthermore, a large part of the U.S. West Coast action area used by Moore and Barlow (2013) in their assessment of possible reasons for the decline include vast areas where the Navy does not conduct in-water training with sonar or explosives.

Establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013) (Read et al. 2014b). 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; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Williams et al. 2014) (Read et al. 2014b). McGregor (2013) summarized sound impacts and described two types of possible effects based on the studies they reviewed: 1) an apparent effect of noise on communication, but with a link between demonstrated proximate cost and ultimate cost in survival or reproductive success being inferred rather than demonstrated, and 2) studies showing a decrease in population density or diversity in relation to noise, but with a relationship that is usually a correlation, so factors other than noise or its effect on communication might account for the relationship. Within the ocean environment, aggregate anthropogenic impacts have to be considered in context of natural variation and climate change (Boyd and Hutchins 2012). These contexts can include additive effects from two or more factors, multiplicity where response from two or more factors is greater than the sum of individual effects, synergism between factors and response, antagonism as a negative feedback between factors, acclimation as a short-term individual response, and adaptation as a long-term population change (Boyd and Hutchins 2012). To address aggregate impacts and responses from any changes due to processes such as habituation, tolerance, and sensitization, future experiments over an extended period of time still need further research (Bejder et al. 2009; Blickley et al. 2012) (Read et al. 2014b).

Some, including (Goldbogen et al. 2013) and (Stockin et al. 2008) have speculated that repeated interruptions of a marine mammal's normal activity could lead to fitness consequences and eventually, long-term implications for the population. However, to our knowledge, empirical data has not confirmed this to be the case. For example, Goldbogen et al. (2013) suggested that if a blue whale responded to MFA sonar by temporarily interrupting feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since unconsumed prey would still be available in the environment following the cessation of acoustic exposure.

If sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks 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.

6.2.12 Stranding

When a marine mammal swims or floats (live or dead) onto shore and becomes "beached" or incapable of returning to sea, the event is termed a "stranding" (Geraci et al. 1999; Geraci and Lounsbury 2005). Animals outside of their "normal" habitat are also sometimes considered "stranded" even though they may not have beached themselves. The legal definition for a stranding within the United States is that: (A) a marine mammal is dead and is (i) on a beach or shore of the United States; or (ii) in waters under the jurisdiction of the United States (including any navigable waters); or (B) a marine mammal is alive and is (i) on a beach or shore of the United States and is unable to return to the water; (ii) on a beach or shore of the United States and, although able to return to the water, is apparently in need of medical attention; or (iii) in the waters under the jurisdiction of the United States (including any navigable waters), but is unable to return to its natural habitat under its own power or without assistance" (16 United States Code Section 1421h).

Marine mammals are subjected to a variety of natural and anthropogenic factors, acting alone or in combination, which may cause a marine mammal to strand on land or die at-sea (Geraci et al. 1999; Geraci and Lounsbury 2005). Even for the fractions of more thoroughly investigated strandings involving post-stranding data collection and necropsies, the cause (or causes) for the majority of strandings remain undetermined. Natural factors related to strandings include the availability of food, predation, disease, parasitism, climatic influences, and aging (Bradshaw et al. 2006; Culik 2004; Geraci et al. 1999; Geraci and Lounsbury 2005; Hoelzel 2003; NRC 2003; Perrin and Geraci 2002; Walker et al. 2005). Anthropogenic factors include pollution (Anonmyous 2010; Elfes et al. 2010; Hall et al. 2006a; Hall et al. 2006b; Jepson et al. 2005; Tabuchi et al. 2006), vessel strike (Berman-Kowalewski et al. 2010; De Stephanis and Urquiola 2006; Geraci and Lounsbury 2005; Jensen and Silber 2003b; Laist et al. 2001), fisheries interactions (Look 2011; Read et al. 2006), entanglement (Baird and Gorgone 2005; Johnson et al. 2005; Saez et al. 2013), and noise (Cox et al. 2006; NRC 2003; Richardson et al. 1995c).

Along the coasts of the continental United States and Alaska between 2001 and 2009, there were approximately 1,400 cetacean strandings and 4,300 pinniped strandings (5,700 total) per year (NMFS 2011b; NMFS 2011c; NMFS 2011d). Several "mass stranding" events—strandings that involve two or more individuals of the same species (excluding a single cow-calf pair)—that have occurred over the past two decades have been associated with naval operations, seismic surveys, and other anthropogenic activities that introduced sound into the marine environment. An in-depth discussion of strandings is presented in DoN (2013c).

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 (MMC 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 (ICES 2005a; ICES 2005b; ICES 2005c). The U.S.-Navy-funded research involving Behavioral Response Studies in SOCAL and the Bahamas discussed previously were motivated by the desire to understand any links between the use of mid-frequency sonar and cetacean behavioral responses, including the potential for strandings. Although these events have served to focus attention on the issue of impacts resulting from the use of sonar, as Ketten (2012) recently pointed out, “ironically, to date, there has been no demonstrable evidence of acute, traumatic, disruptive, or profound auditory damage in any marine mammal as the result [of] anthropogenic noise exposures, including sonar.”

In these previous circumstances, exposure to non-impulsive acoustic energy has been considered a potential indirect cause of the death of marine mammals (Cox et al. 2006). One hypothesis regarding a potential cause of the strandings is tissue damage resulting from “gas and fat embolic syndrome” (Fernandez et al. 2005a; Jepson et al. 2003; Jepson et al. 2005). Models of nitrogen saturation in diving marine mammals have been used to suggest that altered dive behavior might result in the accumulation of nitrogen gas such that the potential for nitrogen bubble formation is increased (Houser 2010; Houser et al. 2001b; Zimmer and Tyack 2007). If so, this mechanism might explain the findings of gas and bubble emboli in stranded beaked whales. It is also possible that stranding is a behavioral response to a sound under certain contextual conditions and that the subsequently observed physiological effects (e.g., overheating, decomposition, or internal hemorrhaging from being on shore) were the result of the stranding rather than direct physical impact from exposure to sonar (Cox et al. 2006).

In May 2003 there was an incident involving the use of mid-frequency sonar by the USS SHOUP, which was portrayed in some media reports at the time as having potentially causing harbor porpoise strandings in the region. On May 5, 2003, in the area of Admiralty Inlet, the USS SHOUP began the use of mid-frequency sonar as part of a training event, which continued until later that afternoon and ended as the USS SHOUP transited Haro Strait heading north. Between May 2 and June 2, 2003, approximately 16 strandings involving 15 harbor porpoises (*Phocoena phocoena*) and 1 Dall’s porpoise (*Phocoenoides dalli*) had been reported to the Northwest Marine Mammal Stranding Network, and allegations were made that these strandings had been caused by the USS SHOUP’s use of sonar. A comprehensive review of all strandings and the events involving USS SHOUP on May 5, 2003, were subsequently presented in a report by U.S. Department of Navy (DON 2004).

Additionally NMFS undertook a series of necropsy analyses on the stranded animals to determine the cause of the strandings (NMFS 2005a; Norman et al. 2004a). Necropsies were performed on 10 of the porpoises and two heads were selected for computed tomographic imaging (Norman et al. 2004a).

None of the 11 harbor porpoises demonstrated signs of acoustic trauma. A putative cause of death was determined for five of the porpoises based only on the necropsy results; two animals had blunt trauma injuries and three animals had indication of disease processes. A cause of death could not be determined in the remaining animals, which is consistent with the expected percentage of marine mammal necropsies conducted within the northwest region. It is important to note, that these determinations were based only on the evidence from the necropsy to avoid bias with regard to determinations of the potential presence or absence of acoustic trauma. For example, the necropsy investigators had no knowledge of other potential external causal factors, such as Specimen 33NWR05005 having been found tangled in a fishing net, which may have otherwise assisted in their determination regarding the likely cause of death for that animal. Additionally, seven of the porpoises collected and analyzed died prior to SHOUP departing to sea on May 5, 2003. Of these seven, one, discovered on May 5, 2003, was in a state of moderate decomposition, indicating it died before May 5; the cause of death was determined, most likely, to be Salmonella septicemia. Another porpoise, discovered at Port Angeles on May 6, 2003, was in a state of moderate decomposition, indicating that this porpoise also died prior to May 5. One stranded harbor porpoise discovered fresh on May 6 is the only animal that could potentially be linked to the USS SHOUP's May 5 active sonar use. Necropsy results for this porpoise found no evidence of acoustic trauma. The remaining eight strandings were discovered 1 to 3 weeks after the USS SHOUP's May 5 use of sonar. Two of the eight porpoises died from blunt trauma injury and a third suffered from parasitic infestation, which possibly contributed to its death (Norman et al. 2004a). For the remaining five porpoises, NMFS was unable to identify the causes of death.

NMFS concluded from a retrospective analysis of stranding events that the number of harbor porpoise stranding events in the approximate month surrounding the USS SHOUP's use of sonar was higher than expected based on annual strandings of harbor porpoises (Norman et al. 2004a). This conclusion in the NMFS report also conflicts with data from The Whale Museum, which has documented and responded to harbor porpoise strandings since 1980 (Osborne 2003). According to The Whale Museum, the number of strandings as of May 15, 2003, was consistent with what was expected based on historical stranding records and was less than that occurring in certain years. For example, since 1992, the San Juan Stranding Network has documented an average of 5.8 porpoise strandings per year. In 1997, there were 12 strandings in the San Juan Islands, with more than 30 strandings throughout the general Puget Sound area. In reporting their findings, NMFS acknowledged that the intense level of media attention to the 2003 strandings likely resulted in increased reporting effort by the public over that which is normally observed (Norman et al. 2004a). NMFS also noted in its report that the "sample size is too small and biased to infer a specific relationship with respect to sonar usage and subsequent strandings." It was also clear that in 2003, the number of strandings from May to June was also higher for the outer coast, indicating a much wider phenomena than use of sonar by USS SHOUP in Puget Sound for one day in May. It was later determined by NMFS that the number of harbor porpoise strandings in the northwest had been increased beginning in 2003 and through 2006. On November 3, 2006, an Unusual Mortality Event in the Pacific Northwest was declared by NMFS (see (DoN 2013c), Cetacean Stranding Report for more detail on this Unusual Mortality Event).

The speculative association of the harbor porpoise strandings to the use of sonar by the USS SHOUP was inconsistent with prior stranding events linked to the use of mid-frequency sonar. Specifically, in prior events strandings occurred shortly after the use of sonar (less than 36 hours) and stranded individuals were spatially co-located. Although MFA sonar was used by the USS SHOUP, the distribution of harbor porpoise strandings by location and with respect to time surrounding the event do not support the suggestion that MFA sonar was a cause of harbor porpoise strandings. Rather, a lack of evidence of any acoustic trauma within the harbor porpoises, and the identification of probable causes of stranding or death in several animals, supports the conclusion that harbor porpoise strandings in 2003 in the Pacific Northwest were unrelated to the sonar activities by the USS SHOUP.

As the ICES (2005c) noted, taken in context of marine mammal populations in general, sonar is not a major threat, or significant portion of the overall ocean noise budget. This has also been demonstrated by monitoring in areas where the Navy operates (Bassett et al. 2010; Baumann-Pickering et al. 2010; Hildebrand et al. 2011; McDonald et al. 2006a; Tyack et al. 2011a). Regardless of the direct cause, the Navy considers potential sonar related strandings important and continues to fund research and work with scientists to better understand circumstances that may result in strandings. During a Navy training event on March 4, 2011, at the Silver Strand Training Complex in San Diego, California, four long-beaked common dolphins were killed by the detonation of an underwater explosive (Danil and St. Leger 2011). This area has been used for underwater demolitions training for at least 3 decades without incident. During this underwater detonation training event, a pod of 100 to 150 long-beaked common dolphins were moving towards the explosive’s 700-yd. (640 m) exclusion zone monitored by a personnel in a safety boat and participants in a dive boat. Within the exclusion zone, approximately 5 minutes remained on a time-delayed firing device connected to a single 8.76 lb (3.8 kg) explosive charge set at a depth of 48 ft. (14.6 m), approximately 0.5 to 0.75 nm from shore. Although the dive boat was placed between the pod and the explosive in an effort to guide the dolphins away from the area, that effort was unsuccessful and three long-beaked common dolphins died as a result of being in proximity to the explosion. In addition, to the three dolphins found dead on March 4th at the event site, the remains of a fourth dolphin were discovered on March 7th (3 days later and approximately 42 mi. (68 km) from the location where the training event occurred), which was assessed as being related to this event (Danil and St. Leger 2011). Details such as the dolphins’ depth and distance from the explosive at the time of the detonation could not be estimated from the 250-yd (229 m) standoff point of the observers in the dive boat or the safety boat.

These dolphin mortalities are the only known occurrence of a 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 (Hooker et al. 2007; O'Shea and Brownell Jr. 1994), and annual fishery-related entanglement, bycatch, injury, and mortality (Baird and Gorgone 2005; Forney and Kobayashi 2007; Saez et al. 2013), have been estimated worldwide to be orders of magnitude greater (hundreds of thousands of animals versus tens of animals; (Culik 2004; ICES 2005c; Read et al. 2006)) than the few potential injurious impacts that could be possible as a result of Navy activities. This does not negate the potential influence of mortality or additional stress to small, regionalized sub-populations which may be at greater risk from human related mortalities (fishing, vessel strike, sound) than populations with larger oceanic level distributions, but overall the Navy's impact in the oceans and inland water areas where training occurs is small by comparison to other human activities. Nonetheless, the focus of our analysis is to determine, considering the status of the resources, the environmental baseline and effects from future non-federal activities, whether the Navy's activities are likely to jeopardize listed species or are likely to destroy or adversely modify critical habitat.

6.2.13 Long-term Consequences to the Individual and the Population

Long-term consequences to a population are determined by examining changes in the population growth rate. Individual effects that could lead to a reduction in the population growth rate include mortality or injury (that removes animals from the reproductive pool), hearing loss (which depending on severity could impact navigation, foraging, predator avoidance, or communication), chronic stress (which could make individuals more susceptible to disease), displacement of individuals (especially from preferred foraging or mating grounds), and disruption of social bonds (due to masking of conspecific signals or displacement). However, the long-term consequences of any of these effects are difficult to predict because individual experience and time can create complex contingencies, especially for intelligent, long-lived animals like marine mammals. While a lost reproductive opportunity could be a measureable cost to the individual, the outcome for the animal, and ultimately the population, can range from insignificant to significant. Any number of factors, such as maternal inexperience, years of poor food supply, or predator pressure, could produce a cost of a lost reproductive opportunity, but these events may be "made up" during the life of a normal healthy individual. The same holds true for exposure to human-generated noise sources. These biological realities must be taken into consideration when assessing risk, uncertainties about that risk, and the feasibility of preventing or recouping such risks. The long-term consequence of relatively trivial events like short-term masking of a conspecific's social sounds, or a single lost feeding opportunity, can be exaggerated beyond its actual importance by focusing on the single event and not the important variable, which is the individual and its lifetime parameters of growth, reproduction and survival.

Population models are well known from many fields in biology including fisheries and wildlife management. These models accept inputs for the population size and changes in vital rates of the population such as the mean values for survival age, lifetime reproductive success, and recruitment of new individuals into the population. The time-scale of the inputs in a population model for long-lived animals such as marine mammals is on the order of seasons, years, or life stages (e.g., neonate, juvenile, reproductive adult), and are often concerned only with the success of individuals from one time period or stage to the next. Unfortunately, information is not available to accurately assess the impact of acoustic and explosive exposure on individual marine mammal vitals rates. Further for assessing the impact of acoustic and explosive impacts to marine mammal populations, many of the inputs required by population models are not known.

Recently, efforts have been made to understand the linkage between a stressor, such as anthropogenic sound, and its immediate behavioral or physiological consequences for the individual, and then the subsequent effects on that individual's vital rates (growth, survival and reproduction), and the consequences, in turn, for the population. In 2005, a panel convened by the National Research Council of the United States National Academy of Sciences published a report on 'Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects'. The panel developed what it called "a conceptual model" that outlined how marine mammals might be affected by anthropogenic noise and how population level effects could be inferred on the basis of observed behavioral changes. They called this model 'Population Consequences of Acoustic Disturbance' (PCAD). In 2009 the US Office of Naval Research (ONR) set up a working group to transform this framework into a formal mathematical structure and determine how that structure could be parameterized using data from a number of case studies. The ONR working group extended the PCAD framework so that it could be used to consider other forms of disturbance and to address the impact of disturbance on physiology as well as behavior. Their current version of that framework is now known as PCoD (Population Consequences of Disturbance) (New et al. 2014). It is important to note that PCoD is ongoing and is an exploratory project to determine how an interim PCoD approach might inform analysis. It is not intended to provide an actual assessment of the population-level consequences of disturbance for beaked whale populations on Navy ranges.

New et al. (2013) developed a mathematical model simulating a functional link between feeding energetics and a species' requirements for survival and reproductions for 21 species of beaked whale. The authors report "reasonable confidence" in their model, although approximately 29 percent (6 of 21 beaked whale species modeled) failed to survive or reproduce, which the authors attribute to possible inaccuracies in the underlying parameters. Based on the model simulation, New et al. (2013) determined that if habitat quality and "accessible energy" (derived from the availability of either plentiful prey or prey with high energy content) are both high, then survival rates are high as well. If these variables are low, then adults may survive but calves will not. For the 29 percent of beaked whale species for which the model failed (within the assumed range of current inputs), the assumption was a 2-year calving period (or inter-calf interval), however, for species with longer gestation periods (such as the 17-month gestation period of Baird's beaked whale (Berardius bairdii), this inter-calf interval may be too short. For Blainville's beaked whale, (Claridge 2013) has shown that calf age at separation is at least 3 years, and that the inter-calf interval at Abaco in the Bahamas may be 4 years. New et al. (2013) acknowledge that an assumed 2-year calving period in the modeling may not be long enough to build up the energetic resources necessary for mother and calf survival.

As another critical model assumption, prey preferences were modeled based on stomach content analyses of stranded animals, which the authors acknowledge are traditionally poor estimates of the diets of healthy animals, as stranded animals are often sick prior to stranding. Stomach content remnants of prey species do not digest equally, as only the hard parts of some prey types remain (e.g., fish otoliths, beaks of cephalopods) and thus often provide an incomplete picture of diet. Given these unknowns and the failure of the simulation to work for 29 percent of beaked whale species, the modeled survival rates of all beaked whales, particularly those modeled with prey having low energy content, may be better than simulated if higher-energy prey makes up a larger part of the diet than assumed by the model simulations.

In short, for the model output New et al. (2013) created to correctly represent links between the species and their environment, that model must identify all the critical and relevant ecological parameters as input variables, provide the correct values for those parameters, and then the model must appropriately integrate modeling functions to duplicate the complex relationships the model intends to represent. If an assumption (model input) such as calving period or prey preferences is incorrect (and there is presently no way to know), then the model would not be representing what may actually be occurring. New et al. (2013) report that their simulations suggest that adults will survive but not reproduce if anthropogenic disturbances result in being displaced to areas of “impaired foraging.” Underlying this suggestion is the additional unstated assumption that habitat capable of sustaining a beaked whale is limited in proximity to where any disturbance has occurred and there are no data to indicate that is a valid assumption.

While the New et al. (2013) model provides a test case for future research, this pilot study has very little of the critical data necessary to form any conclusions applicable to current management decisions. The authors note the need for more data on prey species and reproductive parameters including gestation and lactation duration, as the model results are particularly affected by these assumptions. Therefore, any suggestion of biological sensitivity to the simulation’s input parameters is uncertain.

New et al. (2014) used a simulation model to assess how behavioral disruptions (e.g., significant disruption of foraging behavior) may affect southern elephant seal health, offspring survival, individual fitness, and population growth rate. They suggested their model can determine the population consequences of disturbance from short-term changes in individual animals. Their model assumed that disturbance affected behavior by reducing the number of drift dives in which the animals were feeding and increasing the time they spent in transit. For example, they suggested a disturbance lasting 50 percent of an average annual foraging trip would reduce pup survival by 0.4 percent. If this level of disturbance continued over 30 years and the population did not adapt, the authors found that the population size would decrease by approximately 10 percent.

The findings of New et al. (2014) are not applicable to the temporary behavioral disruptions that may be caused by Navy training and testing activities for a number of reasons. First, the model assumed that individuals would be unable to compensate for lost foraging opportunities. As described previously, available empirical data does not confirm this would be the case. For example, elephant seals are unlikely to be affected by short-term variations in prey availability because they take long foraging trips, allowing for some margin of error in prey availability

(Costa 1993), as cited in New et al. (2014). We expect the species considered in this opinion to be similarly unaffected. We have no information to suggest animals eliciting a behavioral response (e.g., temporary disruption of feeding) to Navy training and testing activities would be unable to compensate for this temporary disruption in feeding activity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Additionally, the behavioral disruption of ESA-listed species reasonably expected to occur due to Navy training and testing activities will not have as long of a duration as those considered in the New et al. (2014) study. As discussed in Section 6.2.11, 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).

In summary, it remains challenging to assess the relative biological significance of, and the potential for long term or population consequences to result from, short-term behavioral responses in marine mammals due to Navy sonar (Southall et al. 2016). While it is important to understand the ways in which behavioral responses may result in long-term impacts to individuals or populations, we do not yet have the data to underpin the link between behavioral response and population consequences (Harris et al. 2017). 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 and testing 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 MITT action area. All monitoring reported are available to the public and posted at www.navymarinespeciesmonitoring.us/.

6.2.14 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).

6.2.14.1 *Frequency Weighting*

Frequency-weighting functions are used to adjust the received sound level based on the sensitivity of the animal to the frequency of the sound. The weighting functions de-emphasize sound exposures at frequencies to which marine mammals are not particularly sensitive. This effectively makes the acoustic thresholds frequency-dependent, which means they are applicable over a wide range of frequencies and therefore applicable for a wide range of sound sources. Frequency-weighting functions, called "M-weighting" functions, were proposed by Southall et al. (2007b) to account for the frequency bandwidth of hearing in marine mammals. These M-weighting functions were derived for each marine mammal hearing group based on an algorithm using the range of frequencies that are within 80 kHz of an animal or group's best hearing. The Southall et al. (2007b) M-weighting functions are nearly flat between the lower and upper cutoff frequencies, and thus were believed to represent a conservative approach to assessing the effects of noise (Figure 23). 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).

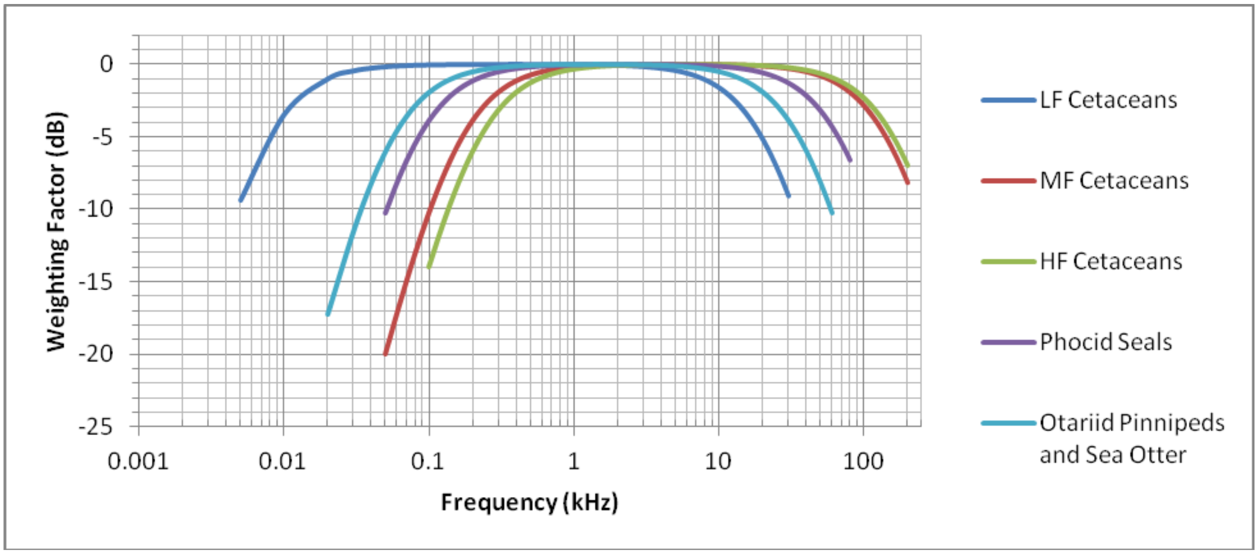

Figure 23. Type I Auditory Weighting Functions Modified from the Southall et al. (2007) M-Weighting Functions

Finneran and Jenkins (2012) considered data since Southall et al. (2007b) and determined two published studies suggested adjustments to the weighting functions were appropriate. The first experiment measured TTS in a bottlenose dolphin after exposure to pure tones with frequencies from 3 to 28 kHz (Finneran and Schlundt 2010). These data were used to derive onset-TTS values as a function of exposure frequency, and demonstrate that the use of a single numeric threshold for onset-TTS, regardless of frequency, is not correct. The second experiment examined how subjects perceived the loudness of sounds at different frequencies to derive equal loudness contours (Finneran and Schlundt 2011a). These data are important because human auditory weighting functions are based on equal loudness contours. The dolphin equal loudness contours provide a means to generate auditory weighting functions in a manner directly analogous to the approach used to develop safe exposure guidelines for people working in noisy environments (NIOSH 1998).

Taken together, the recent higher-frequency TTS data and equal loudness contours provide the underlying data necessary to develop new weighting functions, referred to as Type II auditory weighting functions. Type II auditory weighting functions improve accuracy and avoid underestimating the impacts to animals at higher frequencies as shown in Figure 24. To generate the new Type II weighting functions, Finneran and Schlundt (2011b) substituted lower and upper frequency values which differ from the values used by Southall et al. (2007b).

The new weighting curve predicts appreciably higher (almost 20 dB) susceptibility for frequencies above 3 kHz for bottlenose dolphins, a mid-frequency cetacean. Since data below 3 kHz are not available, the original weighting functions from Southall et al. (2007b) were substituted below this frequency. Low- and high-frequency cetacean weighting functions were extrapolated from the dolphin data as well, because of the suspected similarities of greatest susceptibility at best frequencies of hearing. Similar Type II weighting curves were not developed for pinnipeds since their hearing is markedly different from cetaceans, and because they do not hear as well at higher frequencies. Their weighting curves do not require the same adjustment (see Finneran and Jenkins (2012) for additional details).

The Type II auditory cetacean weighting functions (Figure 24) 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 24) 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.

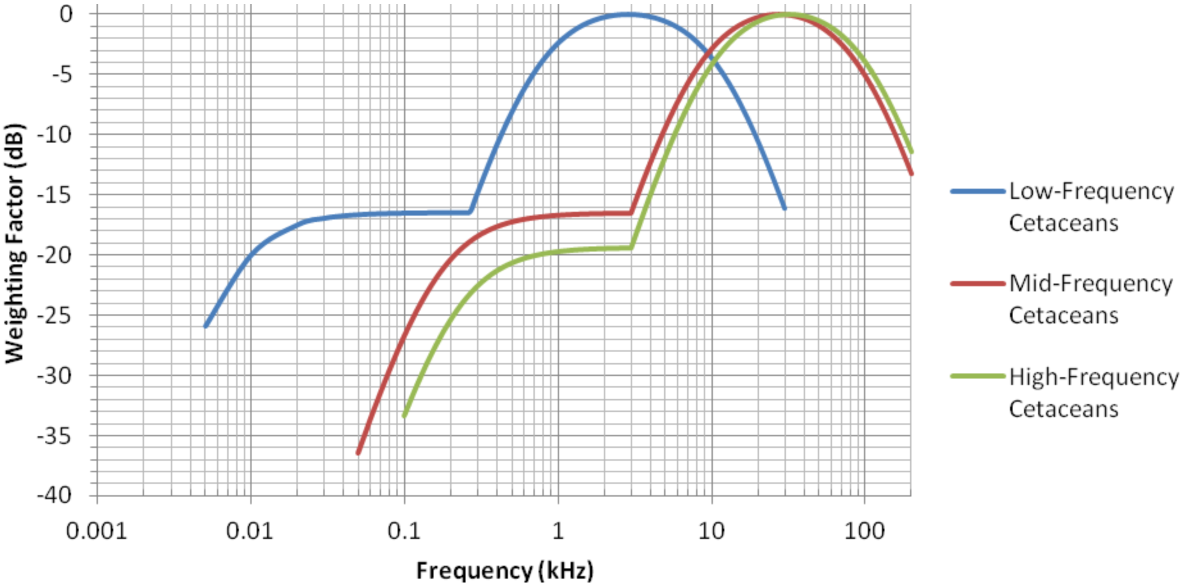

Figure 24. Type II Weighting Functions for Low-, Mid-, and High-Frequency Cetaceans

6.2.14.2 *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.

6.2.14.3 *Hearing Loss – Temporary and Permanent Threshold Shift*

Criteria for physiological effects (Table 30) 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 30. Acoustic Criteria and Thresholds for Predicting Physiological Effects to Marine Mammals Underwater from Sonar and Other Active Acoustic Sources

Hearing Group	Species	Onset temporary	Onset permanent
Low-Frequency Cetaceans	All mysticetes	178 dB re 1 µPa^2-s SEL (Type II weighting)	198 dB re 1 µPa^2-s SEL (Type II weighting)
Mid-Frequency Cetaceans	Dolphins, beaked whales, and medium and large toothed whales	178 dB re 1 µPa^2-s SEL (Type II weighting)	198 dB re 1 µPa^2-s SEL (Type II weighting)
High-Frequency Cetaceans	Porpoises and *Kogia* spp.	152 dB re 1 µPa^2-s SEL (Type II weighting)	172 dB re 1 µPa^2-s SEL (Type II weighting)
Phocid Seals (underwater)	Northern Elephant & Harbor Seals	183 dB re 1 µPa^2-s SEL (Type I weighting)	197 dB re 1 µPa^2-s SEL (Type I weighting)
Otariidae (underwater)	Sea Lion & Fur Seals	206 dB re 1 µPa^2-s SEL (Type I weighting)	220 dB re 1 µPa^2-s SEL (Type I weighting)
Mustelidae (underwater)	Sea Otters		

Notes: dB = decibels, SEL = Sound Exposure Level, dB re 1 µPa^2-s = decibels referenced to 1 micropascal squared second

6.2.14.3.1 Temporary Threshold Shift – Non-Impulsive Sources

Temporary threshold shift involves no tissue damage, is by definition temporary, and therefore is not considered injury. The onset of TTS in mid-frequency cetaceans exposed to non-impulsive sound are derived from multiple studies (Finneran et al. 2010; Finneran et al. 2005; Finneran and Schlundt 2010; Mooney 2009; Schlundt et al. 2000) from two species, bottlenose dolphins and beluga whales. Especially notable are data for frequencies above 3 kHz, where bottlenose dolphins have exhibited lower TTS onset thresholds than at 3 kHz (Finneran 2011; Finneran and

Schlundt 2010). This difference in TTS onset at higher frequencies is incorporated into the weighting functions.

Lucke et al. (2009) measured TTS in a harbor porpoise exposed to a small seismic air gun and those results are reflected in the current impulsive sound TTS thresholds described below. The beluga whale, which had been the only species for which both impulsive and non-impulsive TTS data exist, has a non-impulsive TTS onset value about 6 dB above the (weighted) impulsive threshold (Finneran et al. 2002; Schlundt et al. 2000). Therefore, 6 dB was added to the harbor porpoise's impulsive TTS threshold demonstrated by Lucke et al. (2009) to derive the non-impulsive TTS threshold used in the current Navy modeling for high frequency cetaceans. The first direct measurements of TTS from non-impulsive sound was presented by Kastelein et al. (2012b) for harbor porpoise. These data are fully consistent with the current harbor porpoise thresholds used in the modeling of effects from non-impulsive sources.

There are no direct measurements of TTS or hearing abilities for low-frequency cetaceans. The Navy uses mid-frequency cetacean thresholds to assess PTS and TTS for low-frequency cetaceans, since mid-frequency cetaceans are the most similar to the low frequency group (see Finneran and Jenkins (2012) on the development of the thresholds and criteria).

Pinniped TTS criteria are based on data provided by Kastak et al. (2005) for representative species of both of the pinniped hearing groups: harbor seals (Phocidae) and California sea lions (Otariidae). Kastak et al. (2005) used octave band noise centered at 2.5 kHz to extrapolate an onset TTS threshold. More recently Kastelein et al. (2012a) used octave band noise centered at 4 kHz to obtain TTS thresholds in the same two species resulting in similar levels causing onset-TTS as those found in Kastak et al. (2005). For sea otters, the otariid TTS threshold and weighting function are applied due to similarities in taxonomy and auditory performance. The appropriate frequency weighting function for each species group is applied when using the sound exposure level-based thresholds to predict TTS.

Table 31 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 31**. 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

6.2.14.3.2 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 (NMFS 2008a; NMFS 2008c). TTS values for impulse sound criteria have not been obtained for pinnipeds, but there are TTS data for octave band sound from representative species of both major pinniped hearing groups (Kastak et al. 2005). Impulsive sound TTS criteria for pinnipeds were estimated by applying the difference between mid-frequency cetacean TTS onset for impulsive and non-impulsive sounds to the pinniped non-impulsive TTS data (Kastak et al. 2005), a methodology originally developed by Southall et al. (Southall et al. 2007b). Therefore, the TTS criteria for impulsive sounds from explosions for pinnipeds is 6 dB less than the non-impulsive onset-TTS criteria derived from Kastak et al. (2005).

6.2.14.3.3 Permanent Threshold Shift – Non-Impulsive Sources

There are no direct measurements of PTS onset in marine mammals. Well understood relationships between terrestrial mammalian TTS and PTS have been applied to marine mammals. Threshold shifts up to 40 to 50 dB have been induced in terrestrial mammals without resultant PTS (Miller et al. 1963; Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b). These data would suggest that a PTS criteria of 40 dB would be reasonable for conservatively predicting (overestimating) PTS in marine mammals. Data from terrestrial mammal testing (Ward et al. 1958; Ward et al. 1959a; Ward et al. 1959b) show growth of TTS by 1.5 to 1.6 dB for every 1 dB increase in exposure level. The difference between measureable TTS onset (6 dB) and the selected 40 dB upper safe limit of TTS yields a difference in TTS of 34 dB which, when divided by a TTS growth function of 1.6 indicates that an increase in exposure of 21 dB would result in 40 dB of TTS. For simplicity and additional conservatism we have rounded that number down to 20 dB (Southall et al. 2007b).

Therefore, exposures to sonar and other active acoustic sources with levels 20 dB above those producing TTS are assumed to produce a PTS. For example, an onset-TTS criteria of 195 dB re 1 μPa^2-s would have a corresponding onset-PTS criteria of 215 dB re 1 μPa^2-s. This extrapolation process is identical to that recently proposed by Southall et al. (Southall et al. 2007b). The method overestimates or predicts greater effects than have actually been observed in tests on a bottlenose dolphin (Finneran and Schlundt 2010; Schlundt et al. 2006) and is therefore protective.

Kastak et al. (2007) obtained different TTS growth rates for pinnipeds than Finneran and colleagues obtained for mid-frequency cetaceans. NMFS recommended reducing the estimated PTS criteria for both groups of pinnipeds, based on the difference in TTS growth rate reported by Kastak et al. (2007) (14 dB instead of 20 dB).

The appropriate frequency weighting function for each species group is applied when using the sound exposure level-based thresholds to predict PTS.

Table 32 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 32. 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

6.2.14.3.4 Permanent Threshold Shift – Impulsive Sources

Since marine mammal PTS data from impulsive exposures do not exist, onset PTS levels for these animals are estimated by adding 15 dB to the sound exposure level-based TTS threshold and by adding 6 dB to the peak pressure based thresholds. These relationships were derived by Southall et al. (Southall et al. 2007b) from impulsive noise TTS growth rates in chinchillas. The appropriate frequency weighting function for each species group is applied when using the resulting sound exposure level-based thresholds, as shown in Figure 24, 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. 2001a).

Criteria and thresholds for predicting injury and mortality to marine mammals from explosive sources are listed in Table 33. 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 33. 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 weighting) or 195 dB Peak SPL	161 dB re 1 μPa^2-s (mid-freq weighting) or 201 dB Peak SPL			
Phocid Seals (In-Water)	Harbor, beared, hooded common, spotted, ringed, harp,	177 dB re 1 μPa^2-s (phocid weighting)	192 dB re 1 μPa^2-s (phocid weighting)			

Group	Species	Onset TTS	Onset PTS	Onset Slight GI Tract Injury	Onset Slight Lung Injury	Onset Mortality
	ribbon and gray seals	or 212 dB Peak SPL	or 218 dB Peak SPL			

Equations:

(1) $$\boxed{-39.1M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2}Pa-\sec}$$

(2) $$\boxed{=91.4M^{1/3}\left(1+\frac{D_{Rm}}{10.081}\right)^{1/2}Pa-\sec}$$

D_{rm} = depth of the receiver (animal) in meters; M = mass of the animals in kg; SPL = sound pressure level

[1] Impulse calculated over a delivery time that is the lesser of the initial positive pressure duration or 20 percent of the natural period of the assumed-spherical lung adjusted for animal size and depth.

Notes: TTS = temporary threshold shift, PTS = permanent threshold shift, GI = gastrointestinal, M = mass of animals in kilograms, DRm = depth of receiver (animal) in meters, SEL = Sound Exposure Level, SPL = Sound Pressure Level (re 1 µPa), dB = decibels, dB re 1 µPa = decibels referenced to 1 micropascal, dB re 1 μPa^2-s = decibels referenced to 1 micropascal squared second

6.2.14.3.5 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 (Finneran and Jenkins (2012); 63 FR 230, 66 FR 87, 73 FR 143).

6.2.14.3.6 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. 2001a). After correction for atmospheric and hydrostatic pressures and based on the cube root scaling of body mass, as used in the Goertner injury model (Goertner 1982), the minimum impulse for predicting onset of extensive (50 percent) lung injury for “1 percent Mortality” (defined as most survivors had moderate blast injuries and should survive on their own) and slight lung injury for “zero percent Mortality” (defined as no mortality, slight blast injuries) (Yelverton and Richmond 1981) were derived for each species. As the mortality threshold, the Navy chose to use the minimum impulse level predictive of 50 percent lung injury, even though this injury is likely to result in mortality to only 1 percent of exposed animals. Because the mortality criteria represents a threshold at which 99 percent of exposed animals would be expected to recover, this analysis overestimates the impact on individuals and populations from exposure to impulse sources.

6.2.14.4 ***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.

6.2.14.4.1 Sonar and other active acoustic sources

Potential behavioral effects to marine mammals from non-impulse sound sources underwater were predicted using a behavioral response function. The received sound level is weighted with Type I auditory weighting functions (Southall et al. 2007b) before the behavioral response function is applied. The behavioral response functions estimate the percentage of an exposed population that is likely to exhibit reactions to the sound source. This effects analysis assumes that the potential response from an exposure to non-impulsive sound on individual animals would be a function of the received sound pressure level (SPL; dB re 1 µPa). For example, at 165 dB SPL (dB re 1µPa root mean square), the risk (or probability) of response is defined according to this function as 50 percent. This means that 50 percent of the individuals exposed at that received level would be predicted to exhibit behavioral response.

The behavioral response functions are based on three sources of data: behavioral observations during TTS experiments conducted at the US Navy Marine Mammal Program (Finneran and Schlundt 2004); reconstruction of sound fields produced by the USS Shoup associated with the behavioral responses of killer whales observed in Haro Strait (DoN 2003c; Fromm 2009); and observations of the behavioral response of North Atlantic right whales exposed to alert stimuli containing mid-frequency components (Nowacek et al. 2004). For a detailed discussion of the derivation of the BRFs, see the 2013 Atlantic Fleet Training and Testing FEIS (DoN 2013b). The behavioral response function applied to mysticetes (Figure 25) differs from that used for odontocetes and pinnipeds (Figure 26) in having a shallower slope, which results in the inclusion of more behavioral events at lower amplitudes, consistent with observational data from North Atlantic right whales (Nowacek and Tyack 2007). Although the response functions differ, the intercepts on each figure highlight that each function has a 50 percent probability of harassment at a received level of 165 dB SPL. These analyses assume that sound poses a negligible risk to marine mammals if they are exposed to sound pressure levels below a certain basement value. For both behavioral response functions, the basement received level is 120 dB re 1 µPa.

In some circumstances, some individuals will continue normal behavioral activities in the presence of high levels of human-made noise. In other circumstances, the same individual or other individuals may avoid an acoustic source at much lower received levels (Richardson et al. 1995b; Southall et al. 2007c; Wartzok et al. 2003). These differences within and between individuals appear to result from a complex interaction of experience, motivation, and learning that are difficult to quantify and predict. Therefore, the behavioral response functions represent a relationship that is deemed generally accurate, but may not be true in specific circumstances. Specifically, the behavioral response function treats the received level as the only variable that is relevant to a marine mammal’s behavioral response. However, many other variables such as the marine mammal’s gender, age, and prior experience; the activity it is engaged in during a sound exposure; its distance from a sound source; the number of sound sources; and whether the sound sources are approaching or moving away from the animal can be critically important in determining whether and how a marine mammal will respond to a sound source (Southall et al. 2007c). At present, available data do not allow for incorporation of these other variables in the current behavioral response functions; however, the response function represents the best use of the data that are available.

The Navy uses the behavioral response function to quantify the number of behavioral responses that could qualify as Level B behavioral harassment under the MMPA. As the statutory definition is currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated using the behavioral response functions do not differentiate between the different types of potential reactions nor the significance of those potential reactions. These estimates also do not provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. Therefore, our analysis considers the available scientific evidence to determine the likely nature of modeled behavioral responses and potential fitness consequences for affected individuals.

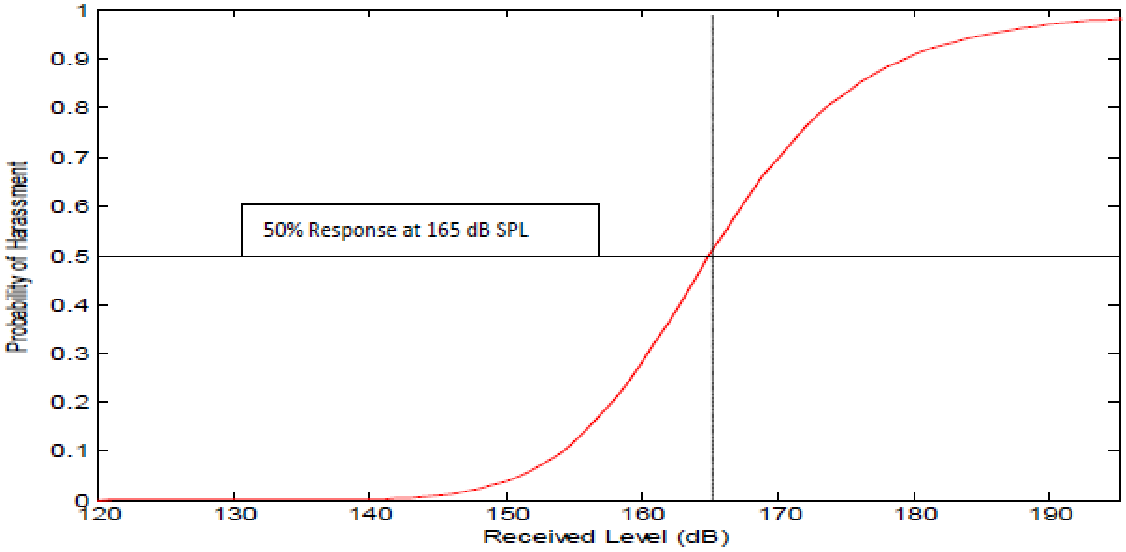

Figure 25. Behavioral response function applied to mysticetes.

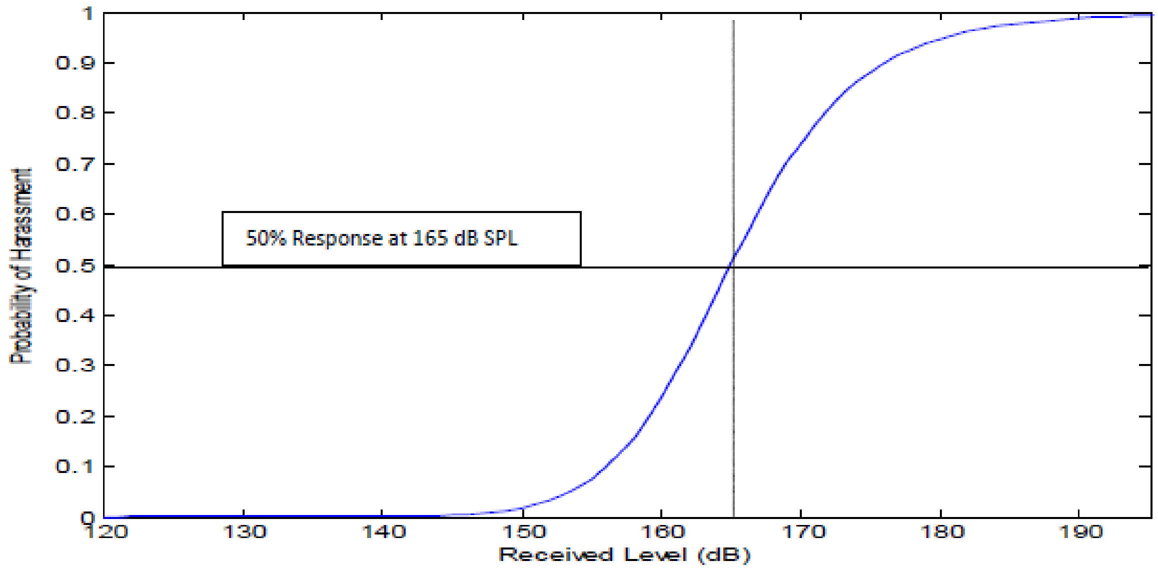

Figure 26. Behavioral response function applied to odontocetes and pinnipeds.

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 34 and Table 35 respectively.

Table 34. 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 35. 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%

6.2.14.4.2 Explosives

The thresholds for a behavioral response from explosives are listed in Table 36. 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 36) are applied to the received sound level before being compared to the threshold.

Table 36. 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.

6.3 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.

6.3.1 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue impacts to sea turtles is primary blast injury and barotrauma after exposure to the shock waves of high-amplitude impulsive sources, such as explosions. Primary blast injuries result from the initial compression of a body exposed to the high pressure of a blast or shock wave. Primary blast injury is usually limited to gas-containing structures (e.g., lung and gut) and the pressure-sensitive components of the auditory system (discussed below) (Craig and Hearn 1998; SG 1991), although additional injuries could include concussive brain damage and cranial, skeletal, or shell fractures (Ketten 1995). Barotrauma refers to injuries caused when large pressure changes occur across tissue interfaces, normally at the boundaries of air-filled tissues such as the lungs. Primary blast injury to the respiratory system may be fatal depending on the severity of the trauma. Rupture of the lung may introduce air into the vascular system, producing air blockages that can restrict oxygen delivery to the brain and heart. Although often secondary in life-threatening severity to pulmonary blast trauma, the gastrointestinal tract can also suffer bruising and tearing from blast exposure, particularly in air-containing regions of the tract. Potential traumas include internal bleeding, bowel perforation, tissue tears, and ruptures of the hollow abdominal organs. Although hemorrhage of solid organs (e.g., liver, spleen, and kidney) from blast exposure is possible, rupture of these organs is rarely encountered. Non-lethal injuries could increase a sea turtle's risk of predation, disease, or infection.

6.3.2 Auditory Trauma

Components of the auditory system that detect smaller or more gradual pressure changes can also be damaged when overloaded at high pressures with rapid rise times. Rupture of the tympanic membrane, while not necessarily a serious or life-threatening injury, may lead to permanent hearing loss (Ketten 1993; Ketten 1995). No data exist to correlate the sensitivity of the tympanic membrane and middle and inner ear to trauma from shock waves from underwater explosions (Viada et al. 2008).

The specific impacts of bulk cavitation (the collapse of air spaces created by explosive detonations) on sea turtles are unknown. The presence of a sea turtle within the cavitation region created by the detonation of small charges could annoy, injure, or increase the severity of the injuries caused by the shock wave. The area of cavitation from a large charge, such as those used in ship shock trials, is expected to result in mortality for smaller animals (Craig 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.

6.3.3 Hearing Loss

Hearing loss could effectively reduce the distance over which sea turtles can detect biologically relevant sounds. Both auditory trauma (a direct injury discussed above) and auditory fatigue may result in hearing loss, but the mechanisms responsible for auditory fatigue differ from auditory trauma. Hearing loss due to auditory fatigue is also known as threshold shift, a reduction in hearing sensitivity at certain frequencies. Threshold shift is the difference between hearing thresholds measured before and after an intense, fatiguing sound exposure. Threshold shift occurs when hair cells in the ear fatigue, causing them to become less sensitive over a small range of frequencies related to the sound source to which an animal was exposed. The actual amount of threshold shift depends on the amplitude, duration, frequency, and temporal pattern of the sound exposure. No studies are published on inducing threshold shift in sea turtles; therefore, the potential for the impact on sea turtles is inferred from studies of threshold shift in other animals.

Temporary threshold shift is a hearing loss that recovers to the original hearing threshold over a period. An animal may not even be aware of a TTS. It does not become deaf, but requires a louder sound stimulus (relative to the amount of TTS) to detect a sound within the affected frequencies. Temporary threshold shift may last several minutes to several days, depending on the intensity and duration of the sound exposure that induced the threshold shift (including multiple exposures).

Permanent threshold shift is a permanent hearing loss at a certain frequency range. Permanent threshold shift is non-recoverable due to the destruction of tissues within the auditory system. The animal does not become deaf, but requires a louder sound stimulus (relative to the amount of PTS) to detect a sound within the affected frequencies.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003a). 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.

6.3.4 Auditory Masking

Auditory masking occurs when a sound prevents or limits the distance over which an animal detects other biologically relevant sounds. When a noise has a sound level above the sound of interest, and in a similar frequency band, auditory masking could occur. Any sound above ambient noise levels and within an animal’s hearing range could cause masking. The degree of masking increases with increasing noise levels; a noise that is just-detectable over ambient levels is unlikely to actually cause any substantial masking, whereas a louder noise may mask sounds over a wider frequency range. In addition, a continuous sound would have more potential for masking than a sound with a low duty cycle. In the open ocean, ambient noise levels are between about 60 and 80 dB re 1 μPa (NRC 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 (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann

2003a). 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.

6.3.5 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 MITT action area, we would expect the turtles considered in this opinion to have a similar physiological stress response. Captive olive ridley hatchlings showed heightened blood glucose levels indicating physiological stress (Zenteno et al. 2008). Repeated exposure to stressors, including human disturbance such as vessel disturbance and anthropogenic sound, may result in negative consequences to the health and viability of an individual or population (Gregory and Schmid 2001). Factors to consider when predicting a stress or cueing response is whether an animal is naïve or has prior experience with a stressor. Prior experience with a stressor may be of particular importance as repeated experience with a stressor may dull the stress response via acclimation (Hazel et al. 2007).

6.3.6 Behavioral Reactions

Little is known about the hearing ability of sea turtles and their response to acoustic disturbance and thus analogous species for which data are available are used to estimate the potential behavioral reactions to sound. The response of a sea turtle to an anthropogenic sound will depend on the frequency, duration, temporal pattern, and amplitude of the sound, as well as the animal's prior experience with the sound and the context in which the sound is encountered (i.e., what the animal is doing at the time of the exposure). Distance from the sound source and whether it is perceived as approaching or moving away could also affect the way a sea turtle responds. Potential behavioral responses to anthropogenic sound could include startle reactions, disruption of feeding, disruption of migration, changes in respiration, alteration of swim speed, alteration of swim direction, and area avoidance.

It is also possible that behavioral reactions could lead to negative physiological consequences. For example, Garcia-Parraga et al. (2014) reported evidence of decompression sickness (DCS; e.g., gas embolism) in sea turtles following capture in trawls or gillnets, with a higher incidence of DCS when caught in deeper waters. It is possible that a sea turtle could have an extreme behavioral avoidance reaction (e.g., surfacing too quickly in an attempt to avoid noise) that could lead to DCS-like symptoms and fitness consequences. However, it should be noted that this is the first, and to our knowledge, only study that has documented DCS-like symptoms in sea turtles. Previous research has suggested sea turtles are protected against DCS through anatomical, physiological, and behavioral adaptations (Berkson 1967; Castellini 2012; Fossette et al. 2010b; 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.

6.3.6.1 *Behavioral Reactions to Impulsive Sound Sources*

Studies of sea turtle responses to sounds are limited, though a few studies examined sea turtle reactions to airguns, which produce broadband impulse sound. O'Hara and Wilcox (1990) attempted to create a sound barrier at the end of a canal using seismic airguns. They reported that loggerhead turtles kept in a 984 ft by 148 ft (300 m by 45 m) enclosure in a 10 m deep canal maintained a distance of 98 ft (30 m) from airguns fired every 15 seconds, with the strongest sound components within the 25 Hz to 1,000 Hz frequency range. McCauley et al. (2000) estimated that the received level at which turtles avoided sound in the O'Hara and Wilcox (1990) experiment was 175 to 176 dB re 1 μPa root mean square.

Moein Bartol et al. (1995) investigated the use of air guns to repel juvenile loggerhead sea turtles from hopper dredges. Sound frequencies of the airguns ranged from 100 Hz to 1,000 Hz at three levels: 175, 177, and 179 dB re 1 μPa at 1 m. The turtles avoided the airguns during the initial exposures (mean range of 24 m), but additional trials several days afterward did not elicit statistically significant avoidance. They concluded that this was due to either habituation or a temporary shift in the turtles' hearing capability.

McCauley et al. (2000) exposed caged green and loggerhead sea turtles to an approaching-departing single air gun to gauge behavioral responses. The trials showed that above a received level of 166 dB re 1 μPa root mean square, the turtles noticeably increased their swimming activity compared to non-operational periods, with swimming time increasing as air gun levels increased during approach. Above 175 dB re 1 μPa root mean square, behavior became more erratic, possibly indicating the turtles were in an agitated state (McCauley et al. 2000). The authors noted that the point at which the turtles showed erratic behavior and exhibited possible agitation would be expected to approximately equal the point at which active avoidance would occur for unrestrained turtles (McCauley et al. 2000).

No obvious avoidance reactions by free-ranging sea turtles, such as swimming away, were observed during a multi-month seismic survey using airgun arrays, although fewer sea turtles were observed when the seismic airguns were active than when they were inactive (Weir 2007). The author noted that sea state and the time of day affected both airgun operations and sea turtle surface basking behavior, making it difficult to draw conclusions from the data. Further, DeRuiter and Larbi Doukara (2012) noted diving behavior following airgun shots in loggerhead turtles, and noted a decreased dive probability with increasing distance from the airgun array.

6.3.6.2 *Behavioral Reactions to Non-Impulsive Sources*

No studies have been performed to examine the response of sea turtles to sonar. However, based on the limited range of hearing, they may respond to sources operating below 2 kHz but are unlikely to sense higher frequency sounds.

6.3.6.3 *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).

6.3.6.4 *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.

6.3.7 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 2013). For example, a typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. For sonar devices that are stationary (e.g. dipped sonar), due to the duty cycle, duration of active transmission in a specific location, and mitigation measures (e.g. avoidance of visible sea turtles), we would not expect repeated exposures.

Establishing a causal link between anthropogenic noise and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013) (Read et al. 2014b). 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 2013) (Read et al. 2014b) (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. 2009; Blickley et al. 2012) (Read et al. 2014b).

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003a). Additionally, they are not known to produce sounds underwater for communication. Therefore, repeated interruptions of a sea turtle’s normal activity due to acoustic stressors is unlikely to lead to fitness consequences and long-term implications for the population. To result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since foraging habitat would still be available in the environment following the cessation of acoustic exposure.

Sea turtles may habituate to, or become tolerant of, repeated exposures over time, such as ambient noise found in areas of high vessel traffic (Hazel et al. 2007). After initial avoidance reactions, loggerhead sea turtles habituated to repeated experimental exposures to airguns of up to a source level of 179 dB re 1 μPa in an enclosure. The habituation behavior was retained by the sea turtles when exposures were separated by several days (Moein Bartol et al. 1995). Individual sea turtles that are more tolerant may stay in a disturbed area, whereas individuals that are more sensitive may leave for areas with less human disturbance. Animals that remain throughout the disturbance may be unable to leave the area for a variety of physiological or environmental reasons. However, given the highly migratory and wide ranging life histories of the species considered in this opinion, we do not believe this will result from Navy training and testing activities in the MITT 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. 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. 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.

6.3.8 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.

6.3.8.1 *Frequency weighting*

Animals generally do not hear equally well across their entire hearing range. Several studies using green, loggerhead, and Kemp's ridley turtles suggest sea turtles are most sensitive to low-frequency sounds, although this sensitivity varies slightly by species and age (Bartol et al. 1999a; Bartol and Ketten 2006; Lenhardt et al. 1994a; Ridgeway et al. 1969). Sea turtles possess an overall hearing range of approximately 100 Hz to 1 kHz, with an upper limit of 2 kHz (Bartol et al. 1999a; Bartol and Ketten 2006; Lenhardt et al. 1994a; Ridgeway et al. 1969). Because hearing thresholds are frequency-dependent, an auditory weighting function was developed for sea turtles (turtle-weighting, or T-weighting). The T-weighting function simply defines lower and upper frequency boundaries beyond which sea turtle hearing sensitivity decreases. The single frequency cutoffs at each end of the frequency range where hearing sensitivity begins to decrease are based on the most liberal interpretations of sea turtle hearing abilities (10 Hz and 2 kHz). These boundaries are precautionary and exceed the demonstrated or anatomy-based hypothetical upper and lower limits of sea turtle hearing. The T-weighting function adjusts the received sound level, based on sensitivity to different frequencies, emphasizing frequencies to which sea turtles are most sensitive and reducing emphasis on frequencies outside of their estimated useful range of hearing. For example, a 160 dB re 1 μPa tone at 10 kHz, far outside sea turtle best range of hearing, is estimated to be perceived by a sea turtle as a 130 dB re 1 μPa sound (i.e., 30 dB lower). Stated another way, a sound outside of the range of best hearing would have to be more intense to have the same impact as a sound within the range of best hearing.

6.3.8.2 ***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 6.3.8.2). 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 (Lenhardt et al. 1985; Moein Bartol and Musick 2003; Wartzok and Ketten 1999; Wever 1978; Wyneken 2001), almost certainly have poorer absolute sensitivity (i.e., higher thresholds) across much of their hearing range than do the mid-frequency cetacean species. Therefore, applying TTS and PTS criteria derived from mid-frequency cetaceans to sea turtles provides a protective approach to estimating acoustic impacts to sea turtles (PTS and TTS data are not available for low-frequency cetaceans). Criteria for hearing loss due to onset of TTS and PTS are based on sound exposure level (for non-impulsive and impulsive sources) and peak pressure (for impulsive sources only).

To determine the sound exposure level, the turtle weighting function is applied to the acoustic exposure to emphasize only those frequencies within a sea turtle's hearing range. Multiple exposures within any 24-hour period are considered one continuous exposure for the purposes of calculating the received sound exposure level for a given individual. This conservatively assumes no recovery of hearing between exposures during a 24-hour period. The weighted sound exposure level is then compared to weighted threshold values for TTS and PTS. If the weighted exposure level meets or exceeds the weighted threshold, then the physiological impact (TTS or PTS) is assumed to occur. For impacts from exposures to impulsive sources, the metric (peak pressure or sound exposure level) and threshold level that results in the longest range to impact is used to predict impacts. Exposures are not calculated for sound sources with a nominal frequency outside the upper and lower frequency hearing limits for sea turtles.

In addition to being discussed below, thresholds for onset of TTS and PTS for impulsive and non-impulsive sounds are summarized in 6.3.8.2.2, 6.3.8.2.1, 6.3.8.2.4, and 6.3.8.2.3, 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 37. 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 38. 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 39 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 39. 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.

6.3.8.2.1 Temporary Threshold Shift – Non-impulsive sources

Based on TTS in other marine vertebrates (Finneran et al. 2002; Southall et al. 2007c) and the lack of information regarding TTS in sea turtles, the total T-weighted sound exposure level of 178 dB re 1 micro Pascal squared second (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.

However, 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).

6.3.8.2.2 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).

6.3.8.2.3 Permanent Threshold Shift – Non-impulsive sources

Since no studies were designed to intentionally induce PTS in sea turtles, levels for onset of PTS for these animals must be estimated using TTS data and relationships between TTS and PTS established in terrestrial mammals. Permanent threshold shift can be estimated based on the growth rate of a threshold shift and the level of threshold shift required to potentially become non-recoverable. A variety of terrestrial and marine mammal data show that threshold shifts up to 40 to 50 dB may be recoverable, and that 40 dB is a reasonable upper limit of a threshold shift that does not induce PTS (Southall et al. 2007b) (Ward et al. 1958; Ward et al. 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).

6.3.8.2.4 Permanent Threshold Shift – Impulsive sources

Because marine mammal and sea turtle PTS data from impulsive exposures do not exist, onset of PTS levels for these animals are estimated by adding 15 dB to the sound exposure level-based TTS threshold and adding 6 dB to the peak pressure-based thresholds. These relationships were derived by Southall et al. (Southall et al. 2007b) from impulsive noise TTS growth rates in chinchillas. This results in onset of PTS thresholds of total weighted sound exposure level of 187 dB re 1 μPa^2-s or peak pressure of 230 dB re 1 μPa for sea turtles. The T-weighting function is applied when using the sound exposure level-based thresholds to predict PTS.

The T-weighted impulsive PTS threshold of 187 dB re 1 μPa^2-s sound exposure level was inadvertently based on Type II weighted cetacean TTS data rather than Type I weighted cetacean TTS data. This resulted in incorrectly lowering the turtle TTS threshold. The sea turtle impulsive PTS threshold, based on Type I mid-frequency cetacean data, should be 198 dB re 1 μPa^2-s. Because an incorrectly lowered threshold was used to quantitatively analyze acoustic impacts to sea turtles, the quantitative impacts presented herein for impulsive PTS are conservative (i.e., over-predicted).

6.3.8.3 ***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.

6.3.8.3.1 Criteria for Mortality and Slight Lung Injury

In air or submerged, the most commonly reported internal bodily injury to sea turtles from explosive detonations is hemorrhaging in the fine structure of the lungs. The likelihood of internal bodily injury is related to the received impulse of the underwater blast (pressure integrated over time), not peak pressure or energy (Richmond et al. 1973; Yelverton and Richmond 1981; Yelverton et al. 1973; Yelverton et al. 1975). Therefore, impulse is used as a metric upon which internal organ injury can be predicted. Onset mortality and onset slight lung injury are defined as the impulse level that would result in one percent mortality (most survivors have moderate blast injuries and should survive) and zero percent mortality (recoverable, slight blast injuries) in the exposed population, respectively. Criteria for onset mortality and onset slight lung injury were developed using data from explosive impacts on mammals (Yelverton and Richmond 1981).

The impulse required to cause lung damage is related to the volume of the lungs. The lung volume is related to both the size (mass) of the animal and compression of gas-filled spaces at increasing water depth. Turtles have relatively low lung volume to body mass and a relatively stronger anatomical structure compared to mammals; therefore application of the criteria derived from studies of impacts of explosions on mammals may be conservative. Table 40 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 40. Species-Specific Sea Turtle Masses for Determining Onset of Extensive and Slight Lung Injury Thresholds

Common Name	Juvenile Mass (kg)	Reference
Loggerhead sea turtle	8.4	Southwood et al. (2007)
Green sea turtle	8.7	Wood and Wood (1993)
Hawksbill sea turtle	7.4	Okuyama et al. (2010)
Olive ridley sea turtle	6.3	McVey and Wibbels (1984) and Caillouet et al. (1986)
Leatherback sea turtle	34.8	Jones (2009)

The scaling of lung volume to depth is conducted because data come from experiments with terrestrial animals held near the water's surface. The calculation of impulse thresholds consider depth of the animal to account for compression of gas-filled spaces that are most sensitive to impulse injury. The impulse required for a specific level of injury (impulse tolerance) is assumed to increase proportionally to the square root of the ratio of the combined atmospheric and hydrostatic pressures at a specific depth with the atmospheric pressure at the (Goertner 1982).

Very little information exists about the impacts of underwater detonations on sea turtles. Impacts of explosive removal operations on sea turtles range from non-injurious impacts (e.g., acoustic annoyance, mild tactile detection, or physical discomfort) to varying levels of injury (i.e., non-lethal and lethal injuries) (Klima et al. 1988; Viada et al. 2008). Often, impacts of explosive events on turtles must be inferred from documented impacts on other vertebrates with lungs or other-gas containing organs, such as mammals and most fishes (Viada et al. 2008). The methods used by Goertner (1982) to develop lung injury criteria for marine mammals may not be directly applicable to sea turtles, as it is not known what degree of protection to internal organs from the shock waves is provided to sea turtles by their shell (Viada et al. 2008). However, the general principles of the Goertner model are applicable, and should provide a protective approach to assessing potential impacts on sea turtles. The Goertner method predicts a minimum primary positive impulse value for onset of slight lung injury and onset of mortality, adjusted for assumed lung volume (correlated to animal mass) and depth of the animal. These equations are shown in Table 37.

6.3.8.3.2 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.

6.3.8.4 ***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 6.3.6, 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 MITT 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.

6.4 Risk Assessment Framework – Scalloped Hammerhead Shark

This section is largely based on a technical report prepared for the Navy: Effects of Mid- and High-Frequency Sonars on Fish (Popper 2008b). Additionally, Popper and Hastings (Popper and Hastings 2009a; Popper and Hastings 2009b) provide a critical overview of some of the most recent research regarding potential effects of anthropogenic sound on fish.

Studies of the effects of human-generated sound on fish have been reviewed in numerous places (e.g., (Hastings and Popper 2005; NRC 1994; Popper 2003; Popper 2008a; Popper and Hastings 2009a; Popper and Hastings 2009b; Popper et al. 2004)). Most investigations, however, have been in the gray literature (non-peer-reviewed reports—see (Hastings and Popper 2005; Popper 2008a; Popper and Hastings 2009a) for extensive critical reviews of this material). Studies have been published assessing the effect on fish of short-duration, high-intensity signals such as might be found near high-intensity sonar, pile driving, or seismic air guns. The investigators in such studies examined short-term effects that could result in death to the exposed fish, as well as hearing loss and long-term consequences (Doksaeter et al. 2009; Govoni et al. 2003; McCauley et al. 2003; Popper et al. 2007; Popper et al. 2005). Information is also discussed from a technical report that resulted from a working group established by the Acoustical Society of America on sound exposure guidelines for fishes and sea turtles (Popper et al. 2014).

6.4.1 Direct Injury of Scalloped Hammerhead Sharks from Non-Impulsive Acoustic Stressors

Potential direct injuries from non-impulsive sound sources, such as sonar, are unlikely because of the relatively lower peak pressures and slower rise times than potentially injurious sources such as explosives. Non-impulsive sources also lack the strong shock wave such as that associated with an explosion. The theories of sonar induced acoustic resonance, bubble formation, neurotrauma, and lateral line system injury are discussed below, although these would likely occur only in fish very close to the sound source and are therefore unlikely to impact entire populations of fish or have an impact in a large area.

No studies have indicated any physiological damage to adult fish from mid-frequency active sonar. Kvadsheim and Sevaldsen (2005) showed that intense sonar activities in herring spawning areas affected less than 0.3 percent of the total juvenile stock. Jørgensen et al. (2005) exposed fish larvae and juveniles representing four species (of three families) to sounds that were designed to simulate mid-frequency sonar transmissions (1 to 6.5 kHz) to study the effects of the exposure on the survival, development, and behavior of the larvae and juveniles (the study used larvae and juveniles of Atlantic herring (*Clupea harengus*) Atlantic cod (*Gadus morhua*), saithe (*Pollachius virens),* and spotted wolffish (*Anarhichas minor*)). The researchers placed the fish in plastic bags three meters from the sound source and exposed them to between four and 100 pulses of one-second duration of pure tones at 1.5, 4, and 6.5 kHz. The fish in only two groups out of the 42 tested exhibited adverse effects beyond a behavioral response. These two groups were both composed of herring, a hearing specialist, and were tested with sound pressure levels of 189 dB re 1 µPa, which resulted in a post-exposure mortality of 20 to 30 percent. In the remaining 40 tests, there were no observed effects on behavior, growth (length and weight), or the survival of fish that were kept as long as 34 days post exposure. While statistically significant losses were documented in the two groups impacted, the researchers only tested that particular sound level once, so it is not known if this increased mortality was due to the level of the test signal or to other unknown factors.

Halvorsen et al. (2012) exposed rainbow trout to simulated MFA (2.8 to 3.8 kHz) sonar at received sound pressure levels of 210 dB re 1 uPa, resulting in cumulative sound exposure levels of 220 dB re 1 uPa. The authors did not observe any mortality or hearing sensitivity changes in rainbow trout and suggested that the frequency range of MFA sonar may be above the most sensitive hearing range of the species. Similarly, Kane et al. (2010) found that low-and mid-frequency exposure caused no acute, gross or histopathology, nor any mortality to rainbow trout, *I. punctatus*, or *Lepomis* sp. Popper et al. (2008c; 2007; Popper and Hastings 2009b) investigated the effects of exposing several fish species to the 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.

Swim bladder resonance is a function of the size and geometry of the air cavity, depth of the fish, and frequency of the transmitted signal. Wavelengths associated with mid-frequency sounds are shorter than wavelengths associated with lower frequency sounds. It is the lower frequencies that are expected to produce swim bladder resonance in adult fishes. Resonance frequencies for juvenile fish are 1 to 8 kHz and can escalate physiological impact (Kvadsheim and Sevaldsen 2005; Løvik and Hovem 1979). Scalloped hammerhead sharks do not have a swim bladder; therefore, they are likely less susceptible to acoustic stressors than many marine fish.

High sound pressure levels may cause bubbles to form from micronuclei in the blood stream or other tissues of animals, possibly causing embolism damage (Ketten 1998). Fish have small capillaries where these bubbles could be caught and lead to the rupturing of the capillaries and internal bleeding. It has also been speculated that this phenomena could also take place in the eyes of fish due to potentially high gas saturation within the fish’s eye tissues (Popper and Hastings 2009b) (Popper and Hastings 2009a). As reviewed in Popper and Hastings (Popper and Hastings 2009a), Hastings (Hastings 1990; Hastings 1995) found ‘acoustic stunning’ (loss of consciousness) in blue gouramis (*Trichogaster trichopterus*) following an 8-minute exposure to a 150 Hz pure tone with a peak sound pressure level (SPL) of 198 dB re 1 μPa. This species of fish has an air bubble in the mouth cavity directly adjacent to the animal’s braincase that may have caused this injury. Hastings (Hastings 1990; Hastings 1995) also found that goldfish exposed to two hours of continuous wave sound at 250 Hz with peak pressures of 204 dB re 1 μPa, and fathead minnows exposed to 0.5 hours of 150 Hz continuous wave sound at a peak level of 198 dB re 1 μPa did not survive. The only study on the effect of exposure of the lateral line system to continuous wave sound (conducted on one freshwater species) suggests no effect on these sensory cells by intense pure tone signals (Hastings et al. 1996).

Popper et al. (2014) developed sound exposure guidelines for fishes exposed to low and mid-frequency naval sonar. The authors did not did not provide evidence that injury or mortality could occur from naval sonar, and indicated that if injury or mortality occurs, it is thought to begin at higher sound levels than have been tested to date. The authors concluded that the relative risk of injury or mortality to fish with no swim bladders (e.g., scalloped hammerhead shark) exposed to low and mid-frequency sonar was low, no matter the distance from the source.

6.4.2 Primary Blast Injury and Barotrauma

The greatest potential for direct, non-auditory tissue effects in fish is primary blast injury and barotrauma following exposure to explosions. Primary blast injury refers to those injuries that result from the initial compression of a body exposed to a blast wave. Primary blast injury is usually limited to gas-containing structures (e.g., swim bladder) and the auditory system. Barotrauma refers to injuries caused when the swim bladder or other gas-filled structures vibrate in response to the signal, particularly if there is a relatively sharp rise-time and the walls of the structure strike near-by tissues and damage them. The relative risk of injury or mortality to scalloped hammerhead sharks with no swim bladders exposed to explosions is anticipated to be much lower than fish with swim bladders.

An underwater explosion generates a shock wave that produces a sudden, intense change in local pressure as it passes through the water (DoN 1998; DoN 2001). Pressure waves extend to a greater distance than other forms of energy produced by the explosion (i.e., heat and light) and are therefore the most likely source of negative effects to marine life from underwater explosions (Craig Jr. 2001b; DoN 2006; SIO 2005). The shock wave from an underwater explosion is lethal to fish at close range causing massive organ and tissue damage and internal bleeding (Keevin and Hempen 1997). At greater distance from the detonation point, the extent of mortality or injury depends on a number of factors including fish size, body shape, orientation, and species (Keevin and Hempen 1997; Wright 1982). Additional factors include the current physical condition of the fish and the presence of a swim bladder. Again, scalloped hammerhead sharks do not have swim bladders. At the same distance from the source, larger fish are generally less susceptible to death or injury, elongated forms that are round in cross-section are less at risk than deep-bodied forms, and fish oriented sideways to the blast suffer the greatest impact (Edds-Walton and Finneran 2006; O'Keefe and Young 1984; Wiley et al. 1981; Yelverton et al. 1975). Species with gas-filled organs have higher mortality than those without them (Continental Shelf Associates Inc. 2004; Goertner et al. 1994).

Two aspects of the shock wave appear most responsible for injury and death to fish: the received peak pressure and the time required for the pressure to rise and decay (Dzwilewski and Fenton 2002). Higher peak pressure and abrupt rise and decay times are more likely to cause acute pathological effects (Wright and Hopky 1998). Rapidly oscillating pressure waves might rupture the kidney, liver, spleen, and sinus and cause venous hemorrhaging (Keevin and Hempen 1997). They can also generate bubbles in blood and other tissues, possibly causing embolism damage (Ketten 1998). Oscillating pressure waves might also burst gas-containing organs. The swim bladder, the gas-filled organ used by most bony fish to control buoyancy, is the primary site of damage from explosives (Wright 1982; Yelverton et al. 1975). Gas-filled swim bladders resonate at different frequencies than surrounding tissue and can be torn by rapid oscillation between high- and low-pressure waves. The range over which damage may occur in a fish without a swim bladder, such as the scalloped hammerhead shark, is on the order of 100 times less than that for swim bladder fish (Popper et al. 2014).

Studies that have documented fish killed during planned underwater explosions indicate that most fish that die do so within one to four hours, and almost all die within a day (Hubbs and Rechnitzer 1952; Yelverton et al. 1975). Fitch and Young (1948) found that the type of fish killed changed when blasting was repeated at the same marine location within 24 hours of previous blasting. They observed that most fish killed on the second day were scavengers, presumably attracted by the victims of the previous day’s blasts. However, fishes collected during these types of studies have mostly been recovered floating on the water’s surface.

Gitschlag et al. (2001) collected both floating fish and those that were sinking or lying on the bottom after explosive removal of nine oil platforms in the northern Gulf of Mexico. They found that 3 to 87 percent (46 percent average) of the specimens killed during a blast might float to the surface. Other impediments to accurately characterizing the magnitude of fish mortality included currents and winds that transported floating fishes out of the sampling area and predation by seabirds or other fishes. Popper et al. (2014) developed sound exposure guidelines for fishes and generally estimated that mortality and mortal injury would occur when the peak sound pressure level from a single explosion exceeds 229 to 234 dB re 1 μPa.

There have been few studies of the impact of underwater explosions on early life stages of fishes (eggs, larvae, juveniles). Fitch and Young (1948) reported the demise of larval anchovies exposed to underwater blasts off California, and Nix and Chapman (1985) found that anchovy and eulachon larvae died following the detonation of buried charges. It has been suggested that impulsive sounds, such as that produced by seismic airguns, may cause damage to the cells of the lateral line in fish larvae and fry when in close proximity (15 ft. [5 m]) to the sound source (Booman et al. 1996). Similar to adult fishes, the presence of a swim bladder contributes to shock wave-induced internal damage in larval and juvenile fishes (Settle et al. 2002). Shock wave trauma to internal organs of larval pinfish and spot from shock waves was documented by Govoni et al. (2003). These were laboratory studies, however, and have not been verified in the field.

Interim criteria for injury of fish were discussed in Stadler and Woodbury (2009). The onset of physical injury would be expected if either the peak sound pressure level exceeds 206 dB re 1 μPa, or the cumulative sound exposure level, accumulated over all pile strikes generally occurring within a single day, exceeds 187 dB re 1 micropascal squared second (μPa^2-s) for fish two grams or larger, or 183 dB re 1 μPa^2-s for smaller fish (Stadler and Woodbury 2009). A more recent study by Halvorsen et al. (2011) used carefully controlled laboratory conditions to determine the level of pile driving sound that may cause a direct injury to the fish tissues (barotrauma). The investigators found that juvenile Chinook salmon (*Oncorhynchus tshawytscha*) received less than a single strike sound exposure level of 179 to 181 dB re 1μ Pa^2-s and cumulative sound exposure level of less than 211 dB re 1 μPa^2-s over the duration of the pile driving activity would sustain no more than mild, non-life-threatening injuries.

6.4.3 Hearing Loss

Exposure to high intensity sound can cause hearing loss, also known as a noise-induced threshold shift, or simply a threshold shift (Miller 1974). A TTS is a temporary, recoverable loss of hearing sensitivity. A TTS may last several minutes to several weeks and the duration may be related to the intensity of the sound source and the duration of the sound (including multiple exposures). A permanent threshold shift (PTS) is non-recoverable, results from the destruction of tissues within the auditory system, and can occur over a small range of frequencies related to the sound exposure. As with temporary threshold shift, the animal does not become deaf but requires a louder sound stimulus (relative to the amount of PTS) to detect a sound within the affected frequencies; however, in this case, the effect is permanent.

Permanent hearing loss, or permanent threshold shift has not been documented in fish. The sensory hair cells of the inner ear in fish can regenerate after they are damaged, unlike in mammals where sensory hair cells loss is permanent (Lombarte et al. 1993; Smith et al. 2006a). As a consequence, any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells.

As we discussed in the *Status of Listed Resources*, Scalloped hammerhead sharks have an inner ear capable of detecting high-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 Hz to 1000 Hz with the highest sensitivity to sounds at the lower ranges (Casper et al. 2003; Casper and Mann 2006; Myrberg Jr. 2001). Sharks generally react to low-frequency tones in this range and are particularly responsive to sounds lower than 375 Hz, which makes them able to perceive sounds including underwater movement and thrashing over long distances.

The “otoliths” or fish ear bones in teleosts (ray-finned, bony fishes) are generally composed of a solid calcium carbonate matrix, while elasmobranch ear structures are calcium carbonate, with exogenous siliceous material, in a gelatinous matrix. It has been suggested that ears with otoliths of a higher density are more sensitive to accelerations (Lychakov and Rebane 2005). Therefore, the solid, dense otoliths of ray-finned fishes should result in a more sensitive ear than the less dense, gelatinous otoliths of sharks including scalloped hammerhead sharks.

Sound is composed of two major components, the propagating sound pressure wave and particle motion. All fishes detect particle motion with their inner ear otoliths (otoconia in elasmobranchs) which act as accelerometers (Casper and Mann 2006). Sound pressure, however, can only be detected by fishes which have a pressure-to-displacement transducer, usually the swim bladder in some teleost fishes (Casper and Mann 2006). Some fishes have evolved a specialized connection between the swim bladder and the inner ear which can transmit the sound pressure signal being detected by the bladder. In the case of the otophysans such as goldfish, modified vertebrae known as the Weberian ossicles have evolved for this function. Scalloped hammerhead sharks (an elasmobranch) and other fishes without swimbladders or any other kind of hearing specialization can only detect the particle motion component of sound (Casper and Mann 2006).

6.4.3.1 Non-impulsive Sound Sources

Studies of the effects of long-duration sounds with sound pressure levels below 170 to 180 dB re 1 μPa indicate that there is little to no effect of long-term exposure on species such as the scalloped hammerhead shark that lack notable anatomical hearing specialization (Amoser and Ladich 2003; Scholik and Yan 2001; Smith et al. 2004a; Smith et al. 2004b; Wysocki et al. 2007a). The longest of these studies exposed young rainbow trout (*Onorhynchus mykiss*), to a level of noise equivalent to one that fish would experience in an aquaculture facility (e.g., on the order of 150 dB re 1 μPa) for about 9 months. The investigators found no effect on hearing (i.e., TTS) as compared to fish raised at 110 dB re 1 μPa.

In contrast, studies on fish with hearing specializations (i.e., greater sensitivity to lower sound pressures and higher frequencies) have shown that there is some hearing loss after several days or weeks of exposure to increased background sounds, although the hearing loss seems to recover (e.g., (Scholik and Yan 2002; Smith et al. 2006a; Smith et al. 2004b)). Smith et al. (Smith et al. 2006a; Smith et al. 2004b) exposed goldfish to noise at 170 dB re 1 μPa and found a clear relationship between the amount of hearing loss (TTS) and the duration of exposure until maximum hearing loss occurred after 24 hours of exposure. A 10-minute exposure resulted in a 5 dB TTS, whereas a 3-week exposure resulted in a 28 dB TTS that took over 2 weeks to return to pre-exposure baseline levels (Smith et al. 2004b) (Note: recovery time not measured by investigators for shorter exposure durations).

Similarly, Wysocki and Ladich (2005) investigated the influence of noise exposure on the auditory sensitivity of two freshwater fish with notable hearing specializations, the goldfish and the lined Raphael catfish (*Platydoras costatus*), and on a freshwater fish without notable specializations, the pumpkinseed sunfish (*Lepomis gibbosus*). Baseline thresholds showed greatest hearing sensitivity around 0.5 kHz in the goldfish and catfish and at 0.1 kHz in the sunfish. For the goldfish and catfish, continuous white noise of approximately 130 dB re 1 μPa at 1 m resulted in a significant TTS of 23 to 44 dB. In contrast, the auditory thresholds in the sunfish declined by 7 to 11 dB. The duration of exposure and time to recovery was not addressed in this study. Scholik and Yan (2001) demonstrated TTS in fathead minnows *(Pimephales promelas*) after a 24-hour exposure to white noise (0.3 to 2.0 kHz) at 142 dB re 1 μPa that did not recover as long as 14 days post-exposure.

Studies have also examined the effects of the sound exposures from Surveillance Towed Array Sensor System Low-Frequency Active sonar on fish hearing (Kane et al. 2010; Popper et al. 2007). Hearing was measured both immediately post exposure and for several days thereafter. Maximum received sound pressure levels were 193 dB re 1 μPa for 324 or 628 seconds. Catfish and some specimens of rainbow trout showed 10 to 20 dB of hearing loss immediately after exposure to the low-frequency active sonar when compared to baseline and control animals; however, another group of rainbow trout showed no hearing loss. Recovery in trout took at least 48 hours, but studies were not completed. The different results between rainbow trout groups is difficult to understand, but may be due to developmental or genetic differences in the various groups of fish. Catfish hearing returned to, or close to, normal within about 24 hours after exposure to low-frequency active sonar. Furthermore, examination of the inner ears of the fish during necropsy (note: maximum time fish were held post exposure before sacrifice was 96 hours) revealed no differences from the control groups in cilliary bundles or other features indicative of hearing loss (Kane et al. 2010).

The study of mid-frequency active sonar by the same investigators also examined potential effects on fish hearing and the inner ear (Halvorsen et al. 2012; Kane et al. 2010). Out of the four species tested (rainbow trout, channel catfish, largemouth bass, and yellow perch) only one group of channel catfish, tested in December, showed any hearing loss after exposure to mid-frequency active sonar. The signal consisted of a 2 second (s) long, 2.8 kHz to 3.8 kHz frequency sweep followed by a 3.3 kHz tone of 1 s duration.

The stimulus was repeated five times with a 25 second interval. The maximum received sound pressure level was 210 dB re 1 μPa. These animals, which have the widest hearing range of any of the species tested, experienced approximately 10 dB of threshold shift that recovered within 24 hours. Channel catfish tested in October did not show any hearing loss. The investigators speculated that the difference in hearing loss between catfish groups might have been due to the difference in water temperature of the lake where all of the testing took place (Seneca Lake, New York) between October and December.

Alternatively, the observed hearing loss differences between the two catfish groups might have been due to differences between the two stocks of fish (Halvorsen et al. 2012). Any effects on hearing in channel catfish due to sound exposure appear to be (Halvorsen et al. 2012; Kane et al. 2010). Investigators observed no damage to cilliary bundles or other features indicative of hearing loss in any of the other fish tested including the catfish tested in October (Kane et al. 2010). Some studies have suggested that there may be some loss of sensory hair cells due to high intensity sources; however, none of these studies concurrently investigated effects on hearing. Enger (1981) found loss of cilliary bundles of the sensory cells in the inner ears of Atlantic cod following 1 to 5 hours of exposure to pure tone sounds between 50 and 400 Hz with a sound pressure level of 180 dB re 1 μPa.

Hastings (1995) found auditory hair-cell damage in a species with notable anatomical hearing specializations, the goldfish (*Carassius auratus*) exposed to 250 Hz and 500 Hz continuous tones with maximum peak levels of 204 dB re 1 μPa and 197 dB re 1 μPa, respectively, for about 2 hours. Similarly, Hastings et al. (1996) demonstrated damage to some sensory hair cells in oscars (*Astronotus ocellatus*) following a 1-hour exposure to a pure tone at 300 Hz with a peak pressure level of 180 dB re 1 μPa. In none of the studies was the hair cell loss more than a relatively small percent (less than a maximum of 15 percent) of the total sensory hair cells in the hearing organs.

In his sounds exposure guidelines for fishes, Popper et al. (2014) estimate exposure to low frequency sonar > 193 dB_{rms} re 1 μPa may lead to TTS in bony fish. The authors were not able to estimate a sound exposure level for mid-frequency sonar at which they would expect TTS. As discussed above, studies of the effects of long-duration sounds with sound pressure levels below 170 to 180 dB re 1 μPa indicate that there is little to no effect of long-term exposure on species such as the scalloped hammerhead shark that lack notable anatomical hearing specialization.

6.4.3.2 Explosions and Other Impulsive Sound Sources

Popper et al. (2005) examined the effects of a seismic airgun array on a fish with hearing specializations, the lake chub (*Couesius plumbeus*), and two species that lack notable specializations, the northern pike (*Esox lucius*) and the broad whitefish (*Coregonus nasus*) (a salmonid). In this study the average received exposure levels were a mean peak pressure level of 207 dB re 1 μPa; sound pressure level of 197 dB re 1 μPa; and single-shot sound exposure level of 177 dB re 1 μPa2-s. The results showed temporary hearing loss for both lake chub and northern pike to both 5 and 20 airgun shots, but not for the broad whitefish. Hearing loss was approximately 20 to 25 dB at some frequencies for both the northern pike and lake chub, and full recovery of hearing took place within 18 hours after sound exposure. Examination of the sensory surfaces of the ears by an expert on fish inner ear structure showed no damage to sensory hair cells in any of the fish from these exposures (Song et al. 2008).

McCauley et al. (2003) showed loss of a small percent of sensory hair cells in the inner ear of the pink snapper (*Pagrus auratus*) exposed to a moving airgun array for 1.5 hours. Maximum received levels exceeded 180 dB re 1 μPa2-s for a few shots. The loss of sensory hair cells continued to increase for up to at least 58 days post exposure to 2.7 percent of the total cells. It is not known if this hair cell loss would result in hearing loss since fish have tens or even hundreds of thousands of sensory hair cells in the inner ear (Lombarte and Popper 1994; Popper and Hoxter 1984) and only a small portion were affected by the sound. The question remains as to why McCauley et al. (2003) found damage to sensory hair cells while Popper et al. (2005) did not. There are many differences between the studies, including species, precise sound source, and spectrum of the sound that it is hard to speculate.

Hastings et al. (2008) exposed the pinecone soldierfish (*Myripristis murdjan*), a fish with anatomical specializations to enhance their hearing; and three species without notable specializations: the blue green damselfish (*Chromis viridis*), the saber squirrelfish (*Sargocentron spiniferum*), and the bluestripe seaperch (*Lutjanus kasmira*) to an airgun array. Fish in cages in 16 ft. (4.9 m) of water were exposed to multiple airgun shots with a cumulative sound exposure level of 190 dB re 1 μPa2-s. The authors found no hearing loss in any fish following exposures.

In his sound exposure guidelines for fishes, Popper et al. (2014) was not able to develop specific criteria (i.e., sound pressure levels that would result in an impact) for sound exposure from explosions that would lead to PTS, TTS, or behavioral responses. However, the authors did indicate that that individuals realtively close to the source would have a higher relative risk of being impacted than individuals further away.

6.4.4 Auditory Masking

Auditory masking refers to the presence of a noise that interferes with a fish’s ability to hear biologically relevant sounds. Fish use sounds to detect predators and prey, and for schooling, mating, and navigating, among other uses (Myrberg Jr. 1980; Popper et al. 2003). Masking of sounds associated with these behaviors could have impacts to fish by reducing their ability to perform these biological functions.

Any noise (i.e., unwanted or irrelevant sound, often of an anthropogenic nature) detectable by a fish can prevent the fish from hearing biologically important sounds including those produced by prey or predators (Myrberg Jr. 1980; Popper et al. 2003). Auditory masking may take place whenever the noise level heard by a fish exceeds ambient noise levels, the animal's hearing threshold, and the level of a biologically relevant sound. Masking is found among all vertebrate groups, and the auditory system in all vertebrates, including fish, is capable of limiting the effects of masking noise, especially when the frequency range of the noise and biologically relevant signal differ (Fay 1988; Fay and Simmons 1999).

The frequency of the sound is an important consideration for fish because many marine fish are limited to detection of the particle motion component of low frequency sounds at relatively high sound intensities (Amoser and Ladich 2005). The frequency of the acoustic stimuli must first be compared to the animal’s known or suspected hearing sensitivity to establish if the animal can potentially detect the sound.

One of the problems with existing fish auditory masking data is that the bulk of the studies have been done with goldfish, a freshwater fish with well-developed anatomical specializations that enhance hearing abilities. The data on other species are much less extensive. As a result, less is known about masking in marine species, many of which lack the notable anatomical hearing specializations. However, Wysocki and Ladich (2005) suggest that ambient sound regimes may limit acoustic communication and orientation, especially in animals with notable hearing specializations.

Tavolga studied the effects of noise on pure-tone detection in two species without notable anatomical hearing specializations, the pin fish (Lagodon rhomboids) and the African mouth-breeder (Tilapia macrocephala), and found that the masking effect was generally a linear function of masking level, independent of frequency (Tavolga 1974a; Tavolga 1974b). In addition, Buerkle studied five frequency bandwidths for Atlantic cod in the 20 to 340 Hz region and showed masking across all hearing ranges (Buerkle 1968; Buerkle 1969). Chapman and Hawkins (1973) found that ambient noise at higher sea states in the ocean has masking effects in cod, *Gadus morhua* (L.), haddock, *Melanogrammus aeglefinus* (L.), and pollock, *Pollochinus pollachinus* (L.), and similar results were suggested for several sciaenid species by Ramcharitar and Popper (2004). Thus, based on limited data, it appears that for fish, as for mammals, masking may be most problematic in the frequency region near the signal. There have been a few field studies that may suggest masking could have an impact on wild fish.

Gannon et al. (2005) showed that bottlenose dolphins (*Tursiops truncatus*) move toward acoustic playbacks of the vocalization of Gulf toadfish (*Opsanus beta*). Bottlenose dolphins employ a variety of vocalizations during social communication including low-frequency pops. Toadfish may be able to best detect the low-frequency pops since their hearing is best below 1 kHz, and there is some indication that toadfish have reduced levels of calling when bottlenose dolphins approach (Remage-Healey et al. 2006). Silver perch have also been shown to decrease calls when exposed to playbacks of dolphin whistles mixed with other biological sounds (Luczkovich et al. 2000). Results of the Luczkovich et al. (2000) study, however, must be viewed with caution because it is not clear what sound may have elicited the silver perch response (Ramcharitar et al. 2006). Astrup (1999) and Mann et al. (1998) hypothesized that high frequency detecting species (e.g., clupeids) may have developed sensitivity to high frequency sounds to avoid predation by odontocetes. Therefore, the presence of masking noise may hinder a fish's ability to detect predators and therefore increase predation.

Of considerable concern is that human-generated sounds could mask the ability of fish to use communication sounds, especially when the fish are communicating over some distance. In effect, the masking sound may limit the distance over which fish can communicate, thereby having an impact on important components of their behavior. For example, the sciaenids, which are primarily inshore species, are one of the most active sound producers among fish, and the sounds produced by males are used to "call" females to breeding sights (Ramcharitar et al. 2001) reviewed in Ramcharitar et al. (2006). If the females are not able to hear the reproductive sounds of the males, there could be a significant impact on the reproductive success of a population of sciaenids. Since most sound production in fish used for communication is generally below 500 Hz (Slabbekoorn et al. 2010), sources with significant low-frequency acoustic energy could affect communication in fish.

Also potentially vulnerable to masking is navigation by larval fish, although the data to support such an idea are still exceedingly limited. There is indication that larvae of some reef fish (species not identified in study) may have the potential to navigate to juvenile and adult habitat by listening for sounds emitted from a reef (either due to animal sounds or non-biological sources such as surf action)(e.g., (Higgs 2005)).

In a study of an Australian reef system, the sound signature emitted from fish choruses was between 0.8 and 1.6 kHz (Cato 1978) and could be detected by hydrophones 3 to 4 nm from the reef (McCauley and Cato 2000). This bandwidth is within the detectable bandwidth of adults and larvae of the few species of reef fish, such as the damselfish, *Pomacentrus partitus*, and bicolor damselfish, *Eupomacentrus partitus*, that have been studied (Kenyon 1996; Myrberg Jr. 1980). At the same time, it has not been demonstrated conclusively that sound, or sound alone, is an attractant of larval fish to a reef, and the number of species tested has been very limited. Moreover, there is also evidence that larval fish may be using other kinds of sensory cues, such as chemical signals, instead of, or alongside of, sound (Atema et al. 2002).

Popper et al. (2014) evaluated the potential for masking in fishes and concluded there is no reason to expect masking from explosions because while the detection of biologically important sounds would be reduced, this effect would only occur during the brief duration of the sound. The authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, it would only occur during the sonar transmissions and would result in a narrow range of frequencies being masked (Popper et al. 2014).

6.4.5 Physiological Stress

As with masking, a fish must first be able to detect a sound above its hearing threshold for that particular frequency and the ambient noise before a behavioral reaction or physiological stress can occur.

Stress refers to biochemical and physiological responses to increases in background sound. The initial response to an acute stimulus is a rapid release of stress hormones into the circulatory system, which may cause other responses such as elevated heart rate and blood chemistry changes. Although an increase in background sound has been shown to cause stress in humans, only a limited number of studies have measured biochemical responses by fish to acoustic stress (Remage-Healey et al. 2006; Smith et al. 2004; Wysocki et al. 2007) (Wysocki et al. 2006) and the results have varied. There is evidence that a sudden increase in sound pressure level or an increase in background noise levels can increase stress levels in fish (Popper and Hastings 2009a, 2009b). Exposure to acoustic energy has been shown to cause a change in hormone levels (physiological stress) and altered behavior in some species such as the goldfish *(Carassius auratus)* (Pickering 1981; Smith et al. 2004a, b), but not all species tested to date, such as the rainbow trout *(Oncorhynchus mykiss*) (Wysocki et al. 2007).

Remage-Healey et al. (2006) found elevated cortisol levels, a stress hormone, in Gulf toadfish exposed to low frequency bottlenose dolphin sounds. Additionally, the toadfish' call rates dropped by about 50 percent, presumably because the calls of the toadfish, a primary prey for bottlenose dolphins, give away the fish's location to the dolphin. The researchers observed none of these effects in toadfish exposed to an ambient control sound (i.e., low-frequency snapping shrimp 'pops').

Smith et al. (2004b) found no increase in corticosteroid, a stress hormone, in goldfish exposed to a continuous, band-limited noise (0.1 to 10 kHz) with a sound pressure level of 170 dB re 1 μPa for 1 month. Wysocki et al. (2007) exposed rainbow trout to continuous band-limited noise with a sound pressure level of about 150 dB re 1 μPa for 9 months with no observed stress effects. Growth rates and effects on the trout's immune system were not significantly different from control animals held at sound pressure level of 110 dB re 1 μPa.

6.4.6 Behavioral Reactions

There are little data available on the behavioral reactions of fish, and almost no research conducted on any long-term behavioral effects or the potential cumulative effects from repeated exposures to loud sounds (Popper and Hastings 2009a, 2009b). Behavioral effects to fish could include disruption or alteration of natural activities such as swimming, schooling, feeding, breeding, and migrating. Sudden changes in sound level can cause fish to dive, rise, or change swimming direction. There is a lack of studies that have investigated the behavioral reactions of unrestrained fish to anthropogenic sound, especially in the natural environment. Studies of caged fish have identified three basic behavioral reactions to sound: startle, alarm, and avoidance (McCauley et al. 2000; Pearson et al. 1992). Changes in sound intensity may be more important to a fish's behavior than the maximum sound level. Sounds that fluctuate in level tend to elicit stronger responses from fish than even stronger sounds with a continuous level (Schwartz 1985).

6.4.6.1 Non-impulsive Sound Sources

Gearin et al. (2000) studied responses of adult sockeye salmon (*Oncorhynchus nerka*) and sturgeon (*Acipenser* sp.) to pinger sounds produced by acoustic devices designed to deter marine mammals from gillnet fisheries. The pingers produced sounds with broadband energy with peaks at 2 kHz or 20 kHz. They found that fish did not exhibit any reaction or behavior change to the pingers, which demonstrated that the alarm was either inaudible to the salmon and sturgeon, or that neither species was disturbed by the mid-frequency sound (Gearin et al. 2000). Based on hearing threshold data, it is highly likely that the salmonids did not hear the sounds.

Culik et al. (2001) did a very limited number of experiments to determine the catch rate of herring in the presence of pingers producing sounds that overlapped with the frequency range of hearing for herring (2.7 kHz to over 160 kHz). They found no change in catch rates in gill nets with or without the higher frequency (greater than 20 kHz) sounds present, although there was an increase in the catch rate with the signals from 2.7 kHz to 19 kHz (a different source than the higher frequency source). The results could mean that the fish did not "pay attention" to the higher frequency sound or that they did not hear it, but that lower frequency sounds may be attractive to fish. At the same time, it should be noted that there were no behavioral observations on the fish, and so how the fish actually responded when they detected the sound is not known.

Doksaeter et al. (2009) studied the reactions of wild, overwintering herring to Royal Netherlands Navy experimental mid-frequency active sonar and killer whale feeding sounds. The behavior of the fish was monitored using upward looking echosounders. The received levels from the 1 to2 kHz and 6 to 7 kHz sonar signals ranged from 127 to 197 dB re 1 μPa and 139 to 209 dB re 1 μPa, respectively. Escape reactions were not observed upon the presentation of the mid-frequency active sonar signals; however, the playback of the killer whale sounds elicited an avoidance reaction. The authors concluded that mid-frequency sonar could be used in areas of overwintering herring without substantially affecting the fish.

There is evidence that elasmobranchs respond to human-generated sounds. Myrberg and colleagues did experiments in which they played back sounds and attracted a number of different shark species to the sound source (Myrberg et al. 1969; Myrberg et al. 1976; Myrberg et al. 1972; Nelson and Johnson 1972). The results of these studies showed that sharks were attracted to low-frequency sounds (below several hundred Hz), in the same frequency range of sounds that might be produced by struggling prey.

However, sharks are not known to be attracted by continuous signals or higher frequencies (which they presumably cannot hear). Studies documenting behavioral responses of fish to vessels show that Barents Sea capelin (*Mallotus villosus*) may exhibit avoidance responses to engine noise, sonar, depth finders, and fish finders (Jørgensen et al. 2004). Avoidance reactions are quite variable depending on the type of fish, its life history stage, behavior, time of day, and the sound propagation characteristics of the water (Schwartz 1985). Misund (1997) found that fish ahead of a ship that showed avoidance reactions did so at ranges of 160 to 490 ft. (48.8 to 149.4 m). When the vessel passed over them, some species of fish responded with sudden escape responses that included lateral avoidance or downward compression of the school.

In a study by Chapman and Hawkins (1973) the low-frequency sounds of large vessels or accelerating small vessels caused avoidance responses by herring. Avoidance ended within 10 seconds after the vessel departed. Twenty-five percent of the fish groups habituated to the sound of the large vessel and 75 percent of the responsive fish groups habituated to the sound of small boats.

Popper et al. (2014) concluded that the relative risk of a fish illiciting a behavioral reaction in response to low-frequency sonar was low, regardless of the distance from the sound source. The authors expected a limited number of fish species may respond to mid-frequency sonar since most fish do not have specializations that enable them to hear above 2,500 Hz (Popper et al. 2014, Halvorsen et al 2012).

6.4.6.2 Explosions and Other Impulsive Sound Sources

Pearson et al. (1992) exposed several species of rockfish (*Sebastes spp.*) to a seismic airgun. The investigators placed the rockfish in field enclosures and observed the fish's behavior while firing the airgun at various distances for 10 minute trials. Dependent upon the species, rockfish exhibited startle or alarm reactions between peak to peak sound pressure level of 180 dB re 1 μPa and 205 dB re 1 μPa. The authors reported the general sound level where behavioral alterations became evident was at about 161 dB re 1 μPa for all species. During all of the observations, the initial behavioral responses only lasted for a few minutes, ceasing before the end of the 10-minute trial.

Similarly, Skalski et al. (1992) showed a 52 percent decrease in rockfish (*Sebastes* sp.) caught with hook-and-line (as part of the study—fisheries independent) when the area of catch was exposed to a single airgun emission at 186 to 191 dB re 1 μPa (mean peak level) (See also (Pearson et al. 1987; Pearson et al. 1992)). They also demonstrated that fish would show a startle response to sounds as low as 160 dB re 1 μPa, but this level of sound did not appear to elicit decline in catch. Wright (1982) also observed changes in fish behavior as a result of the sound produced by an explosion, with effects intensified in areas of hard substrate.

Wardle et al. (2001) used a video system to examine the behaviors of fish and invertebrates on reefs in response to emissions from seismic airguns. The researchers carefully calibrated the airguns to have a peak level of 210 dB re 1 μPa at 16 m and 195 dB re 1 μPa at 109 m from the source. There was no indication of any observed damage to the marine organisms. They found no substantial or permanent changes in the behavior of the fish or invertebrates on the reef throughout the course of the study, and no marine organisms appeared to leave the reef.

Engås et al. (1996) and Engås and Løkkeborg (2002) examined movement of fish during and after a seismic airgun study by measuring catch rates of haddock (*Melanogrammus aeglefinus*) and Atlantic cod as an indicator of fish behavior using both trawls and long-lines as part of the experiment. These investigators found a significant decline in catch of both species that lasted for several days after termination of airgun use. Catch rate subsequently returned to normal. The conclusion reached by the investigators was that the decline in catch rate resulted from the fish moving away from the airgun sounds at the fishing site. However, the investigators did not actually observe behavior, and it is possible that the fish just changed depth.

The same research group showed, more recently, parallel results for several additional pelagic species including blue whiting and Norwegian spring spawning herring (Slotte et al. 2004). However, unlike earlier studies from this group, the researchers used fishing sonar to observe behavior of the local fish schools. They reported that fish in the area of the airguns appeared to go to greater depths after the airgun exposure compared to their vertical position prior to the airgun usage. Moreover, the abundance of animals 18 to 31 miles (29 to 50 km) away from the ensonification increased, suggesting that migrating fish would not enter the zone of seismic activity.

Alteration in natural behavior patterns due to exposure to impulsive noise (such as pile driving and explosions) has not been well studied. However, one study (Mueller-Blenkle et al. 2010), which took place with fish enclosed in a mesocosm (an enclosure providing a limited body of water with close to natural conditions), demonstrated behavioral reactions of cod and Dover sole (*Solea solea*) to impulsive sounds from pile driving. Sole showed a significant increase in swimming speed. Cod reacted, but not significantly, and both species showed directed movement away from the sources with signs of habituation after multiple exposures. For sole, reactions were seen with peak sound pressure levels of 144 to 156 dB re 1 μPa; and cod showed altered behavior at peak sound pressure levels of 140 to 161 dB re 1 μPa. For both species, this corresponds to a peak particle motion between 6.51×10^{-3} and 8.62×10^{-4} meters per second squared (m/s^2).

Popper et al. (2014) indicated very little is known about the effects of explosions of wild fish behavior, but suggested that startle responses, likely lasting less than a second, could occur from exposure to explosives and that such responses would not necessarily result in significant changes to subsequent behavior.

6.4.7 Repeated Exposures of Fish

As noted previously, there is almost no research on any long-term behavioral effects or the potential cumulative effects from repeated exposures of fish to loud sounds (Popper and Hastings 2009a, 2009b). 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; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Williams et al. 2014) (Read et al. 2014b). Within the ocean environment, aggregate anthropogenic impacts have to be considered in context of natural variation and climate change (Boyd and Hutchins 2012). To address aggregate impacts and responses from any changes due to processes such as habituation, tolerance, and sensitization, future experiments over an extended period of time still need further research (Bejder et al. 2009; Blickley et al. 2012) (Read et al. 2014b).

Most of the scalloped hammerhead sharks that may be exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training in the MITT Study Area. The majority of the proposed training and testing activities occur over a small spatial scale relative to the entire action area, have few participants, and are of a short duration (the order of a few hours or less). These periodic or episodic exposure and response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. For example, Navy sonar systems are generally deployed from highly mobile vessels or in-water devices which do not directly target ESA-listed resources. A typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min) (Navy 2013). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. For sonar devices that are stationary (e.g. dipped sonar), due to the duty cycle, duration of active transmission in a specific location, and mitigation measures (e.g. avoidance of visible ESA-listed resources), we would not expect repeated exposures.

Repeated exposure to impulsive acoustic stressors may be more likely as successive detonations could occur in close proximity to one another. However, with the exception of those individuals that are close enough to be killed or seriously injured, we would expect the effect of repeated expsoures under this scenario to not impact the fitness of individual sharks. For example, anecdotal evidence indicates scalloped hammerhead sharks, particularly juveniles, may spend a considerable amount of time in Apra Harbor, a relatively small area where underwater detonations are expected to occur. As suggested by Popper et al. (2014b), most responses to detonations are expected to be temporary startle responses with the animal resuming normal activity shortly after exposure. The same would be expected if multiple startle responses occur resulting from successive detonations. It also may be possible for detonations to temporarily affect juvenile or adult shark refugia or foraging habitat selection. However, given these animal’s mobility, we would expect individuals to temporarily select alternative refuge or forage sites nearby until the exposure levels in their initially selected location have decreased. To result in significant fitness consequences we would have to assume that an individual shark could not compensate for lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Similarly, we would have to assume individuals could not find alternative refuge habitat nearby. There is no indication this is the case. Therefore, even temporary displacement from initially selected habitat is not expected to impact the fitness of any individual animals because we would expect equivalent habitat to be available in close proximity. Further, foraging generally occurs at night, when underwater detonations will not take place, suggesting feeding behavior is not likely to be affected. As noted previously, scalloped hammerhead sharks may also experience hearing loss if an individual is in close proximity to an underwater detonation. However, hearing loss would be temporary because unlike marine mammals, fish are capable of regenerating sensory hairs and no permanent hearing loss has ever been reported in fish (Lombarte et al. 1993) (Smith et al. 2006b). Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these responses.

Multiple stressors may also have synergistic effects. For example, fish that experience temporary hearing loss or injury from acoustic stressors could be more susceptible to physical strike and disturbance stressors via a decreased ability to detect and avoid threats. These interactions are speculative, and without data on the combination of multiple Navy stressors, the synergistic impacts from the combination of Navy stressors are difficult to predict in any meaningful way.

6.5 Delineation of Individuals of *Acropora globiceps*

This section provides a summary of the available information used to assess the impacts of the proposed Navy activities on ESA-listed corals. A significant portion of this information is derived from the Final Listing Determinations on Proposal To List 66 Reef-Building Coral Species and To Reclassify Elkhorn and Staghorn Corals (79 FR 53877).

Reef-building corals are clonal organisms. A single larva will develop into a discrete unit (the primary polyp) that then produces modular units (*i.e.*, genetically-identical copies of the primary polyp) of itself, which are connected seamlessly through tissue and skeleton. These modular units may be solitary (*e.g.*, fungiid corals) or colonial. Most reef-building coral species are colonial, including all species covered in the final rule (79 FR 53877). Colony growth is achieved mainly through the addition of more polyps, and colony growth is indeterminate (see Figure 27 for an example of a coral colony). The colony can continue to exist even if numerous polyps die, or if the colony is broken apart or otherwise damaged. The biology of such clonal, colonial species creates ambiguity with regard to delineation of the individual in reef-building corals, specifically: (1) polyps versus colonies; (2) sexually-produced versus asexually-produced colonies; and (3) difficulty determining colony boundaries (79 FR 53877).

Summaries of how NMFS addressed these sources of ambiguity during the listing process are provided below, leading to a conclusion regarding the delineation of the “individual” for the species covered by the final rule.

Figure 27. Example of a Coral Colony, *Acropora globiceps*. (Veron, 2000) Source: (http://www.aims.gov.au/)

The "polyp" could be considered the smallest unit of the individual for reef-building corals. Each polyp in a coral colony consists of a column of tissue with a mouth and tentacles on the upper side, growing in a cup-like skeletal structure (the corallite) made of calcium carbonate that the polyp produces through calcification. The polyps are the building blocks of the colony, and most colony growth occurs by increasing the number of polyps and supporting skeleton. Polyps carry out the biological functions of feeding, calcification, and reproduction. However, because the polyps within a colony are modular units, and connected to one another physiologically (*i.e.*, via nerve net and gastrovascular cavity, and are the same sex), single polyps within a colony were not considered by NMFS to be individuals for purposes of the final listing.

Colonies are founded by either sexually-produced larvae that settle and become the primary polyp of a colony, or asexually-produced fragments of pre-existing colonies that break off to form a new colony. Fragments from the same colony can fuse back together into the same colony if they are close enough to grow together. Fragmentation in branching species may lead to a large number of asexually-produced, genetically identical colonies, commonly resulting in a population made up of more asexually-produced colonies than sexually-produced colonies (Hughes 1984). Sexually-produced colonies are important to the population by increasing the genetic diversity of the population. Asexual reproduction, though it does not create new genetic individuals, is likely the more critical mode for some species, especially branching species, allowing them to grow, occupy space, and persist between relatively rare events of sexual reproduction. NMFS used the concept of the "physiological colony" as the entity considered to be an individual.

The physiological colony for reef-building colonial species is defined as any colony of the species, whether sexually or asexually produced (79 FR 53877). A physiological colony is generally autonomous from other colonies of the same species. However, colony morphology, partial colony mortality, and other colony growth characteristics (*e.g.*, formation of stands or thickets) can complicate the delineation of physiological colonies from one another in the field. In those cases, colony shape may not distinguish colonies from one another, and boundaries between separate encrusting colonies that have grown together may be difficult or impossible to make out visually. Partial mortality of colonies, especially larger colonies, can also mask the boundaries between colonies, because the algae-encrusted coral skeleton of a partially dead colony may appear to delineate two or more colonies. In addition, many reef-building coral species occur in stands or thickets that may be tens of meters or more in diameter (*e.g.*, some *Acropora* species), possibly consisting of multiple colonies or only one large colony, also masking the boundaries between colonies. In each of these instances, the actual number of genetically-distinct individuals can only be determined through genetic analysis.

NMFS’ final rule considered the “individual” for each of the proposed species including *Acropora globiceps*, to be the “physiological colony,” as defined above. That is, polyps are not considered individuals, but sexually- and asexually-produced colonies are considered individuals because they are a type of physiological colony and are the unit that can be identified in the field.

6.6 Marine Species Density Estimates

Marine mammal and sea turtle density estimates that were used in NAEMO modeling for acoustic effects and our risk analyses on the effects of various stressors from Navy training and testing activities are discussed below. Estimates of abundance or density for corals and scalloped hammerhead shark in the MITT action area were not available.

6.6.1 Marine Mammal Density Estimates

Esimates of marine mammal densities from the Pacific NMSDD (See Section 3.1.1) are summarized in Table 41 below.

Table 41. NMSDD Estimates for Marine Mammals in the MITT Study Area and Transit Corridor

Species	Location	Spring	Summer	Fall	Winter
Blue whale	MITT Study Area	0.00001	0.00001	0.00001	0.00001
	Transit Corridor	0.00001	0.00001	0.00001	0.00001
Fin whale	MITT Study Area	0.00001	0.00001	0.00001	0.00001
	Transit Corridor	0.00001	0.00001	0.00001	0.00001
Sei whale	MITT Study Area	0.00029	0	0.00029	0.00029
	Transit Corridor	0.00013	0	0.00013	0.00013

Humpback whale*	MITT Study Area	0.00089	0.00089	0.00089	0.00089
	Transit Corridor	0.00089	0.00089	0.00089	0.00089
Sperm whale	MITT Study Area	0.00291	0.00291	0.00291	0.00291
	Transit Corridor	0.00176	0.00176	0.00176	0.00176

The units for numerical values are animals/km^2. 0 = species is not expected to be present

*Note that the humpback whales occurring in the Transit Corridor may be from the endangered Western North Pacific DPS or the Hawaii DPS, which is not listed under the ESA. Because we do not have separate density information for the two DPSs in the transit corridor, we are unable to assign take occurring in the transit corridor to specific DPSs. To be conservative, we assign all takes occurring in the transit corridor to humpback whales from the endangered Western North Pacific DPS.

6.6.2 Sea Turtle Density Estimates

Species-specific sea turtle density estimates for the MITT action area are derived from scientific literature and Navy monitoring and surveying efforts. For more information regarding the specific data sources used to estimate density for each sea turtle species in the MITT action area, see the Pacific Navy Marine Species Density Database Technical Report (Hanser et al. 2013).

The species-specific density estimates used in the NAEMO model are detailed in Table 42. For green and hawksbill turtles, available information allowed the Navy to estimate density by geographic location or habitat type (e.g., nearshore Guam, Pelagic and transit corridor, etc). For green sea turtles, information was not available to estimate DPS-specific densities for the various locations in the action area. Green sea turtles from the various DPSs that may occur in the action area are not visually distinguishable from each other and genetic testing in the area has been limited. In general, in-water information on green sea turtles in the action area (which largely overlaps with the delineation of the Central West Pacific DPS of green sea turtles) is limited (80 FR 15271). Leatherback and loggerhead are much less common in the MITT action area and a low density was assumed to represent the occasional transit of the action area. The density values below represent year round estimates. Density estimates by season are not available for sea turtles. For the MITT action area, an area was selected along the transit corridor to represent the range of different habitats that could occur along the corridor, including both island (Wake Island) and open ocean habitats. This is referred to as the “Transit Corridor” and assigned specific density estimates.

Table 42. NMSDD Estimates for Sea Turtles in the MITT Action Area and Transit Corridor

Species	Location	Density
Hawksbill	Nearshore FDM	1.0734
	Nearshore other islands	0.1342
	Pelagic and Transit Corridor	0.000024
Green	Nearshore Guam	0.2968
	Nearshore Tinian and other Islands	11.8
	Pelagic and Transit Corridor	0.000391
Loggerhead	MITT Action Area	0.000022
Leatherback	MITT Action Area	0.00022

The units for numerical values are animals/km².

6.7 Stressors Not Likely to Adversely Affect ESA-listed Species

The following section discusses stressors that are not likely to adversely affect ESA-listed species. If a stressor is likely to adversely affect any of the ESA-listed species in the MITT action area, it is discussed further in Section 6.8 and carried forward in our effects analysis. We address the specific species that are not likely to be adversely affected by respective stressors in those sections.

6.7.1 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, sea turtles, and scalloped hammerhead sharks 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, sea turtles, and scalloped hammerhead sharks 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, sea turtles, or scalloped hammerhead sharks 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, sea turtles, or scalloped hammerhead sharks. Therefore, the effects of weapon firing, launching, and impact noise on ESA-listed marine mammals, sea turtles, and scalloped hammerhead sharks would not rise to the level of take and any behavioral impacts would be insignificant. Because the potential effects of weapons firing, launch, and impact noise are insignificant, these potential stressors are not likely to adversely affect ESA-listed species and will not be considered further in this opinion.

Corals are not biologically capable of detecting noise except as vibrations of water particles caused by low frequency sounds. Therefore, effects to ESA-listed corals from noise stressors associated with weapon firing and impacts are insignificant and not likely to adversely affect them.

6.7.2 Effects of Aircraft Noise

Many of the activities the U.S. Navy conducts in the MITT 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. 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.

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.

6.7.2.1 Cetaceans

Studies have shown that aircraft presence and operation can result in changes in behavior of cetaceans (Arcangeli and Crosti 2009; Holt et al. 2009; Luksenburg and Parsons 2009; Noren et al. 2009; Patenaude et al. 2002; Richter et al. 2006; Richter et al. 2003; Smultea et al. 2008). Several authors have reported that sperm whales did not react to fixed-wing aircraft or helicopters in some circumstances (Au and Perryman 1982, Clarke 1956b, Green et al. 1992) (Gambell 1968) and reacted in others (Clarke 1956b, Fritts et al. 1983, Patenaude et al. 2002, Richter et al. 2003, 2006, Smultea et al. 2008, Würsig et al. 1998) (Mullin et al. 1991). 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). Richardson et al. (1995c) 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.

Thorough reviews on the behavioral reactions of marine mammals to aircraft and missile overflight are presented in Richardson et al. (1995c), 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 (Holst et al. 2011; Manci et al. 1988). Richardson et al. (1995c) 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. 1995c). These factors could include aircraft type (single engine, multi-engine, jet turbine), flight path (centered on the animal, off to one side, circling, level and slow), environmental factors such as wind speed, sea state, cloud cover, and locations where native subsistence hunting continues.

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Efroymson et al. 2000; Koski et al. 1998). Richardson et al. (1995c) 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. 1995c). Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 492 ft. (150 m) or higher. It should be noted that bowhead whales may have more acute responses to anthropogenic activity than many other marine mammals since these animals are often presented with limited egress due to limited open water between ice floes. Additionally, many of these animals may be hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

Variable responses to aircraft have been observed in toothed whales, though overall little change in behavior has been observed during flyovers. Toothed whale responses to aircrafts include diving, slapping the water with their flukes or flippers, swimming away from the direction of the aircraft, or not visibly reacting (Richardson et al. 1995c).

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. 1992; Richter et al. 2006; Richter et al. 2003; Smultea et al. 2008; Wursig et al. 1998). In one study, sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al. 1995c). A group of sperm whales responded to a circling aircraft (altitude of 800 to 1,100 ft. [244 to 335 m]) by moving closer together and forming a defensive fan-shaped semicircle, with their heads facing outward. Several individuals in the group turned on their sides, apparently to look up toward the aircraft (Smultea et al. 2008). Whale-watching aircraft apparently caused sperm whales to turn more sharply but did not affect blow interval, surface time, time to first click, or the frequency of aerial behavior (Richter et al. 2003). Navy aircraft do not fly at low altitude, hover over, or follow whales and so are not expected to evoke this type of response.

Smaller delphinids generally react to overflights either neutrally or with a startle response (Wursig et al. 1998). The same species that show strong avoidance behavior to vessel traffic (*Kogia* species and beaked whales) also react to aircraft (Wursig et al. 1998). Beluga whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns to a greater extent than mysticetes in the same area (Patenaude et al. 2002). These reactions increased in frequency as the altitude of the helicopter dropped below 492 ft. (150 m).

Exposure to fixed-wing aircraft noise would be brief (seconds) as an aircraft quickly passes overhead at relatively high speeds. Exposure to helicopter overflights may last longer and would have a higher likelihood of causing a behavioral response from a cetacean due to the lower flight altitudes and longer duration the helicopter could be in proximity to an animal. Exposures to both sorts of aircraft would be infrequent based on the transitory and dispersed nature of the overflights; repeated exposure to individual animals over a short period of time (hours or days) is extremely unlikely. Furthermore, the sound exposure levels would be relatively low to marine mammals that spend the majority of their time underwater. Based on the literature described above (e.g., Nowacek et al. 2007) and the nature of Navy aircraft use in the MITT action area, cetaceans exposed to aircraft noise could exhibit a short-term behavioral response (if they respond at all), but we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore such reactions would not rise to the level of take. The effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed cetaceans considered in this opinion. The potential effect of aircraft noise on ESA-listed cetaceans will not be considered further in this opinion.

6.7.2.2 Sea turtles

Based on sea turtle sensory biology (Bartol et al. 1999b; Ketten and Bartol 2005; Ketten and Bartol 2006; Lenhardt et al. 1994b; 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. Exposure to fixed-wing aircraft noise would be brief (seconds) as an aircraft quickly passes overhead at relatively high speeds. Exposure to helicopter overflights may last longer and would have a higher likelihood of causing a behavioral response from a sea turtle due to the lower flight altitudes and longer duration the helicopter could be in proximity to an animal. Exposures to both sorts of aircraft would be infrequent based on the transitory and dispersed nature of the overflights; repeated exposure to individual animals over a short period of time (hours or days) is extremely unlikely. Furthermore, the sound exposure levels would be relatively low to sea turtles that spend the majority of their time underwater. Based on the information described above and the nature of Navy aircraft use in the MITT action area, sea turtles exposed to aircraft noise could exhibit a short-term behavioral response (if they respond at all), but we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, such reactions would not rise to the level of take. The effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed sea turtles considered in this opinion. The potential effect of aircraft noise on ESA-listed sea turtles will not be considered further in this opinion.

6.7.2.3 Scalloped hammerhead sharks

Based on limited morphological and scientific study, scalloped hammerhead sharks are likely less sensitive to above-water acoustic noise, such as that originating from aircraft, than marine mammals. This is because scalloped hammerheads do not spend time at the surface like marine mammals and sea turtles and a very limited amount of sound from aircraft would propogate to depths where scalloped hammerheads reside. Scalloped hammerhead sharks exposed to aircraft noise could exhibit a short-term, minor behavioral response (if they respond at all), but we do not expect these short term behavioral reactions to create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Therefore, such reactions would not rise to the level of take. The effect of aircraft noise that may result in behavioral reactions is insignificant and is not likely to adversely affect the ESA-listed scalloped hammerhead sharks considered in this opinion. The potential effect of aircraft noise on ESA-listed scalloped hammerhead sharks will not be considered further in this opinion.

6.7.2.4 Acropora globiceps

Adult coral colonies are not biologically capable of detecting noise except as vibrations of water particles. The only known auditory sensing capabilities known for coral is the response of free-swimming coral larvae to underwater sounds produced by reef fish and crustaceans, as reported by Vermeij et al. (2010). The authors reported that some species of coral larvae detect reef sounds and then show an attraction response to the sounds generated on the reefs. However, due to the low number of aircraft flights, typical altitudes of flights, sporadic occurrence of flights, limited duration of flights, and the lack of substantial sound propagation into the water column from aircraft, there is a low probability of exposing coral larvae to aircraft noise at perceivable levels. Therefore, the effects of aircraft noise on *Acropora globiceps* are insignificant and aircraft noise is not likely to adversely affect this species.

6.7.3 Effects of Vessel Noise

Sounds emitted by large vessels can be characterized as low-frequency, continuous, or tonal, and sound pressure levels at a source will vary according to speed, burden, capacity and length (Richardson et al. 1995c) (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.

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 Green 2000; Bain et al. 2006; Bauer 1986; Bejder et al. 1999; Bejder and Lusseau. 2008; Bejder et al. 2009; Bryant et al. 1984; Corkeron 1995; Erbe 2002b; Félix 2001; Goodwin and Cotton 2004; Lemon et al. 2006; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2003; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002b; Wursig et al. 1998). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jaakson 1994; Evans et al. 1992; Evans et al. 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Based on the suite of studies of cetacean behavior to vessel approaches (Au and Perryman 1982; Bain et al. 2006; Bauer and Herman 1986; Bejder et al. 1999; Bejder et al. 2006a; Bejder et al. 2006b; 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. 2003; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams and Ashe 2007; Williams et al. 2002b; Wursig et al. 1998) (Aguilar Soto et al. 2006; Arcangeli and Crosti 2009; Au and Green 2000; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams et al. 2009)(Acevedo 1991), 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 2007; Williams and Ashe 2007);

- *The distance between vessel and marine mammals* when the animal perceives that an approach has started and during the course of the interaction (Au and Perryman 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.

Most of the investigations reported that animals tended to reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies (Corkeron 1995; Lundquist et al. 2012; Lusseau 2003; Lusseau 2004; Nowacek et al. 2001; Van Parijs and Corkeron 2001; Williams et al. 2002a; Williams et al. 2002b). In the process, their dive times increased, vocalizations and jumping were reduced (with the exception of beaked whales), individuals in groups move closer together, swimming speeds increased, and their direction of travel took them away from the source of disturbance (Baker and Herman 1989; Edds and Macfarlane 1987; Evans et al. 1992; Kruse 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 repeated disturbance from tour vessel traffic may interrupt social interactions, and postulated that those repeated disturbances may carry energetic costs, or otherwise affect individual fitness. However, they were unable to determine if such disturbances were likely to cause long-term harm. For behavioral responses to result in energetic costs that result in long-term harm, such disturbances would likely need to be sustained for a significant duration or extent where individuals exposed would not be able to select alternate habitat to recover and feed. Typical Navy training and testing activities would not likely result in such prolonged exposures and preclusion of individuals from feeding, breeding, or sheltering habitat.

Würsig et al. (1998) studied the behavior of cetaceans in the northern Gulf of Mexico in response to survey vessels and aircraft. They reported that false killer whales either did not respond or approached the ship (most commonly to ride the bow). Four of 15 sperm whales avoided the ship while the remainder appeared to ignore its approach.

Because of the number of vessels involved in U.S. Navy training exercises and testing activities, the vessel speed, and the use of course changes as a tactical measure with the associated sounds, the available evidence leads us to expect marine mammals to treat Navy vessels as stressors. Further, without considering differences in sound fields associated with any active sonar that is used during these exercises, the available evidence suggests that major training exercises, unit- and intermediate-level exercises, and testing activities would represent different stress regimes because of differences in the number of vessels involved, vessel maneuvers, and vessel speeds.

Sound emitted from large vessels, such as shipping and cruise ships, is the principal source of low-frequency noise in the ocean today, and marine mammals are known to react to or be affected by that noise (Foote et al. 2004; Hatch and Wright 2007; Hildebrand 2005; Holt et al. 2008b; Kerosky et al. 2013; Melcon et al. 2012; Richardson et al. 1995c).

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 2000; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams and Noren 2009). Noren et al. (2009) conducted research in the San Juan Islands in 2005 and 2006 and their findings suggested that close approaches by vessels impacted the whales' behavior and that the whale-watching guideline minimum approach distance of 100 m may be insufficient in preventing behavioral responses. Most studies of this type are opportunistic and have only examined the short-term response to vessel sound and vessel traffic (Magalhaes et al. 2002; Noren et al. 2009; Richardson and Wursig 1995; Watkins 1981c).

Long-term and cumulative implications of vessel sound on marine mammals remains largely unknown. Clark et al. (2009) provided a discussion on calculating the cumulative impacts of anthropogenic noise on baleen whales and estimated the noise from the passage of two vessels could reduce the optimal communication space for North Atlantic right whales by 84 percent (see also (Hatch et al. 2012).

Navy combatant vessels have been designed to generate minimal noise and use ship quieting technology to elude detection by enemy passive acoustic devices (Mintz and Filadelfo 2011; Southall et al. 2005). Given this, and that they are much fewer in number than their commercial counterparts, Navy vessels are a small component of overall vessel traffic and vessel noise in most areas where they operate (see Mintz and Filadelfo (2011) for a general summary for the U.S. Exclusive Economic Zone).

6.7.3.1 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 1981a). Only when vessels approached more closely did the fin whales in this study alter their behavior by increasing time at the surface and exhibiting avoidance behaviors. Other studies have shown when vessels are near, some but not all fin whales change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Au and Green 2000; Castellote et al. 2012b; Richter et al. 2003; Williams et al. 2002b).

Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012b) demonstrated that fin whales’ songs had shortened duration and decreased bandwidth, center frequency, and peak frequency in the presence of high shipping noise levels. It is not known if these changes in vocal behavior corresponded to other behaviors.

In the Watkins (1981a) study, humpback whales did not exhibit any avoidance behavior but did exhibit minor behavioral reactions to vessel presence. In a study of regional vessel traffic, Baker et al. (1983) found that when vessels were in the area, the respiration patterns of the humpback whales changed. The whales also exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 1.24 and 2.48 mi. (2,000 and 4,000 m) away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were within approximately 1.2 mi. (2,000 m; (Baker and Herman 1983)). Similar findings were documented for humpback whales when approached by whale watch vessels in Hawaii (Au and Green 2000).

Gende et al. (2011) reported on observations of humpback whales in inland waters of Southeast Alaska subjected to frequent cruise ship transits (i.e., in excess of 400 transits in a 4-month season in 2009). The study was focused on determining if close encounter distance was a function of vessel speed. The reported observations, however, seem in conflict with other reports of avoidance at much greater distance so it may be that humpback whales in those waters are more tolerant of vessels (given their frequency) or are engaged in behaviors, such as feeding, that they are less willing to abandon. This example again highlights that context is critical for predicting and understanding behavioral reactions as concluded by Southall et al. (2007b) and Ellison et al. (2012b).

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. 2009). 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 MITT action area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Nowacek et al. 2004) and therefore might provide insight to behavioral responses of other baleen whales. North Atlantic right whales continue to use habitats in high vessel traffic areas (Nowacek et al. 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Nowacek et al. 2004; Terhune and Verboom 1999). Although this may minimize potential disturbance from passing ships, it does increase the whales’ vulnerability to potential ship strike. The regulated approach distance for North Atlantic right whales is 500 yards (yd.) (457 m) (NMFS 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). Melcon et al. (2012) also recently documented that blue whales increased the proportion of time spent producing certain types of calls when vessels were present. Conversely, decreases in singing activity by humpback whales have been noted near Brazil due to boat traffic (Sousa-Lima and Clark 2008). The Central North Pacific stock of humpback whales is the focus of whale-watching activities in both its feeding grounds (Alaska) and breeding grounds (Hawaii). Regulations addressing minimum approach distances and vessel operating procedures are in place in Hawaii, however, there is still concern that whales may abandon preferred habitats if the disturbance is too high (Allen and Angliss 2010b).

The available information suggests that ESA-listed mysticetes are either not likely to respond to vessel noise (e.g., Nowacek et al. 2004a, Watkins 1981a) or are expected to respond only briefly if exposed to noise from Navy vessels. Expected behavioral responses include startle responses, brief avoidance behavior (e.g., Jahoda et al. 2003), or changes in respiration rate (e.g., Baker et al. 1983). Most avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect mysticete reactions to vessel noise to have any measurable effects on any individual’s fitness and any such responses are not expected to rise to the level of a take. Therefore, the effects of vessel noise on ESA-listed mysticetes is insignificant and not likely to adversely affect them.

6.7.3.2 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.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity (Holt et al. 2008a) as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado and Wartzok 2008). Likewise, modification of multiple vocalization parameters has been shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al. 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al. 2005). Killer whales off the northwestern coast of the United States have been observed to increase the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which has been suggested as a response to increased masking noise produced by the vessels (Foote et al. 2004). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. For example, the source level of killer whale vocalizations has been shown to increase with higher background noise levels associated with vessel traffic (Hotchkin and Parks 2013). In addition, calls with a high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al. 2008a).

Similar to mysticetes, the available information suggests that ESA-listed odontocetes are either not likely to respond to vessel noise or are expected to respond only briefly if exposed to noise from Navy vessels. Expected behavioral responses include brief avoidance behavior (e.g., Magalhaes et al. 2002; Wursig et al. 1998) or changes in vocal patterns (e.g., Holt et al. 2008a, Lesage et al. 1999). However, we expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. For these reasons, and given the short duration of vessel noise stressors and the infrequency of this stressor, we do not expect odontocete reactions to vessel noise to have any measurable effects on any individual’s fitness and any such responses are not expected rise to the level of a take. Therefore, the effects of vessel noise on ESA-listed odontocetes is insignificant and not likely to adversely affect them.

6.7.3.3 Sea turtles

Limited information is available on how or if ESA-listed sea turtles may respond to noise from Navy vessels during MITT training and testing activities. Hazel et al. (2007) suggested that green turtles rely more on visual than auditory cues when reacting to approaching Navy vessels. This suggests that if sea turtles were to respond to a Navy vessel, the animal might not respond to the vessel based on noise alone. Popper et al. (2014c) stated that no data is available on the potential effect of vessel noise or other continuous sounds on sea turtles. The only potential effect Popper et al. (2014c) suggested could occur from vessel noise was masking or behavioral response, with a higher likelihood of a behavioral response occurring the closer the sea turtle is to the vessel. Masking is not expected to result from Navy vessels. Navy vessels are a small component of overall vessel traffic and vessel noise in most areas where they operate (see Mintz and Filadelfo (2011) concerning a general summary for the U.S. Exclusive Economic Zone). Therefore, we do not expect Navy vessels from MITT activities to significantly contribute to ambient sound levels in the action area. Any masking of biologically important sounds for sea turtles would be temporary, occurring only when a vessel and sea turtle are in close proximity to one another and we do not expect such an incident to have any measurable effects on an animal's fitness. If a sea turtle responded behaviorally to noise from a Navy vessel, most responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior shortly thereafter. Any behavioral responses to vessel noise are expected to be temporary (e.g., a startle response, brief avoidance behavior) and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. We do not expect these short term behavioral reactions to create the likelihood of injury to such an extent as to significantly disrupt normal behavioral patterns and therefore such reactions would not rise to the level of take. Therefore, the effect of vessel noise that may result in behavioral reactions or temporary periods of masking is insignificant and is not likely to adversely affect ESA-listed sea turtles.

6.7.3.4 Scalloped hammerhead sharks

According to Popper et al. (2014a), there is no direct evidence of mortality or injury to fish from vessel noise. Further, TTS from continuous sound sources (e.g., vessel noise) has only been documented in fish species that have specializations for enhanced sensitivity to sound. None of the ESA-listed salmonids considered in this opinion are known to have these specializations. Data for species which do not have these specializations have shown no TTS in response to long term exposure to continuous noise sources (Popper et al. 2014a). This includes a study of rainbow trout (*Oncorhynchus mykiss*) exposed to increased noise for nine months in an aquaculture facility. The study also did not document any negative effects on the health of the fish from this increased exposure to noise (Popper et al. 2014a; Wysocki et al. 2007b).

Popper et al. (2014a) suggest that low frequency vessel noise (primarily from shipping traffic) may mask sounds of biological importance. As described previously in this opinion, none of the ESA-listed salmonids considered in this opinion have hearing specializations (which would indicate they may rely heavily on hearing for essential life functions) and they are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, and orient in the water column (Popper et al. 2014a). Further, hearing is not thought to play a role in salmon migration (e.g., Putnam et al. 2013). Additionally, any potential masking would be temporary as both the fish and vessel would be transiting the action area (likely at different speeds and in different directions). For these reasons, we do not expect any short-term instances of masking to have any fitness consequences for any individual fish.

Vessel activity may result in changes in fish behavior (Popper et al. 2014a). However, any behavioral responses to vessel noise are expected to be temporary (e.g., a startle response, brief avoidance behavior) and we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the vessel noise. We do not expect these short term behavioral reactions to create the likelihood of injury to such an extent as to significantly disrupt normal behavioral patterns and therefore such reactions would not rise to the level of take. Therefore, the effect of vessel noise that may result in behavioral reactions is insignificant and is not likely to adversely affect scalloped hammerhead sharks.

6.7.3.5 Acropora globiceps

Adult coral colonies are not biologically capable of detecting noise except as vibrations of water particles. The only known auditory sensing capabilities known for coral is the response of free-swimming coral larvae to underwater sounds produced by reef fish and crustaceans, as reported by Vermeij et al. (2010). The authors reported that some species of coral larvae detect reef sounds and then show an attraction response to the sounds generated on the reefs. However, potential interference in the ability of coral larvae to detect reef sounds would be temporary, lasting only the duration that the vessel is in the immediate vicinity of the larval coral. Since Navy vessels are generally transiting during Navy training and testing, exposures and potential masking would be brief. We do not expect these brief interruptions to inhibit the ability of coral larvae to detect reef habitat. Therefore, the effects of vessel noise on *Acropora globiceps* are insignificant and vessel noise is not likely to adversely affect this species.

6.7.4 Effects of Fiber Optic Cables, Guidance Wires, and Decelarators/Parachutes from Entanglement

Expended materials from U.S. Navy training and testing may pose a risk of entanglement or ingestion to marine mammals, sea turtles, or scalloped hammerhead sharks in the MITT action area. These interactions could occur at the sea surface, in the water column, or on the seafloor. Similar to interactions with other types of marine debris (e.g., fishing gear, plastics), interactions with military expended materials could result in negative sub-lethal effects, mortality, or result in no impact.

Expended materials from Navy training and testing may include the following: fiber optic cables, guidance wires, parachutes, 55 gallon drums, sonobuoy tubes, unrecovered aircraft stores (e.g., fuel tanks, carriages, dispensers, racks), ship hulks, and targets. At-sea targets are usually remotely-operated airborne, surface, or subsurface traveling units, most of which are designed to be recovered for reuse. However, if they are used during activities that utilize high-explosives, they may fragment. Expendable targets that may fragment include air-launched decoys, surface targets (such as marine markers, paraflares, cardboard boxes, and 10 ft. diameter red balloons), and mine shapes. Most expended materials and target fragments are expected to sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time.

There has never been a reported or recorded instance of marine mammals, sea turtles, or scalloped hammerhead sharks becoming 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 low densities of threatened or endangered species in the MITT 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, sharks, and sea turtles. Bottom feeding animals have an increased likelihood of encountering expended materials because they may find them during feeding long after the training or testing event has occurred. If encountered, foraging animals may ingest the item or become entangled by it.

If encountered, it is unlikely that an animal would get entangled in a fiber optic cable, guidewire, parachute, or other expended material while it was sinking or upon settling to the seafloor. An animal would have to swim through loops or become twisted within the cable or wire to become entangled, and given the properties of the expended cables and wires (low breaking strength and sinking rates) the probability of injury or mortality from expended materials is low. Specifically, fiber optic cable is brittle and would be expected to break if kinked, twisted or sharply bent. Thus, the physical properties of the fiber optic cable would not allow the cable to loop, greatly reducing the likelihood of entanglement of ESA-listed species. Based on degradation times, guidance wires would break down within one to two years and no longer pose an entanglement risk. The length of the guidance wires vary, but greater lengths increase the likelihood that a marine mammal or sea turtle could become entangled. Parachutes used by the Navy range in size from 18 to 48 in. (46 to 122 cm), but the vast majority of expended decelerator/parachutes are small (18 in.) cruciform shaped decelerators used with sonobuoys. They have short attachment lines and, upon water impact, may remain at the surface for 5 to 15 seconds before the decelerator/parachute and its housing sink to the seafloor. Entanglement of an animal in a parachute assembly at the surface or within the water column would be unlikely, since the parachute would have to land directly on an animal, or an animal would have to swim into it before it sinks. It is possible that a bottom feeding animal (e.g., sperm whale, sea turtle) could become entangled when they are foraging in areas where parachutes have settled on the seafloor. For example, if bottom currents are present, the canopy may temporarily billow and pose a greater entanglement threat. However, the likelihood of curents causing a billowing of a parachute and being encountered by an ESA-listed species is so unlikely as to be considered discountable. In conclusion, based on their deep-water location of use, their sinking rate, their degredation rate, and the low density of ESA-listed marine mammals, sea turtles, and scalloped hammerhead sharks in the action area, the likelihood of these species becoming entangled in expended materials is so low as to be discountable. Therefore, entanglement is not likely to adversely affect ESA-listed marine mammals, sea turtles, or scalloped hammerhead sharks.

Fiber optic cables, guidance wires, and decelarators/parachutes will be used over deep water, long distances from habitat types where *Acropora globiceps* occur. For this reason, we consider the likelihood of *Acropora globiceps* colonies becoming entangled in fiber optic cables, guidance wires, decelarators/parachutes, or other expended material from Navy training and testing activities to be discountable. Therefore, fiber optic cables, guidance wires, decelarators/parachutes, and other expended material are not likely to adversely affect *Acropora globiceps*.

6.7.5 Effects of Munitions and Other Military Expended Materials from Ingestion

The only munitions and other materials small enough to be ingested by ESA-listed marine mammals, sea turtles, and scalloped hammerhead sharks are small- and medium-caliber projectiles, broken pieces of firing targets, chaff, flare caps, decelerators/parachutes, and shrapnel fragments from explosive ordnance. Most expendable materials will be used over deep water and these items will sink quickly and settle on the seafloor with the exception of chaff and some firing target materials. Given the limited time most items will spend in the water column it is not reasonably expected these items will be accidentally ingested by ESA-listed species not accustomed to foraging on the sea floor. The ESA-listed species potentially exposed to expended munitions and shrapnel fragments while foraging on the sea floor is limited to sperm whales and sea turtles, and the scalloped hammerhead shark. Although scalloped hammerhead sharks and green, loggerhead, and hawksbill sea turtles are known to forage along the sea floor they are restricted to doing so in relatively shallow, near shore areas where expendable material use is not normally conducted. In contrast, sperm whales are capable of foraging along the sea floor in deep water. However, the relatively low density of both sperm whales and expended materials along the vast sea floor suggests ingestion would be rare. In cases where sperm whales, sea turtles, and scalloped hammerhead sharks do accidentally ingest expended materials small enough to be eaten it is likely they will pass through the digestive tract and not result in an increased likelihood of injury from significant disruption of breeding, feeding, or sheltering. ESA-listed corals are likely not capable of ingesting expended materials due to their size and their exposure would be limited due to corals primarily existing in near shore areas. Therefore, ingestion of expended materials by corals is not reasonably expected to occur.

Chaff is composed of fine fibers of silicon dioxide coated with aluminum alloy. Due to their light weight and small size they float and can be carried great distances in both air and water currents. Their dispersal in wind and water results in chaff fibers likely occur in low densities on the ocean surface. Given the small size, low densities, and low toxicity of chaff, any accidental ingestion by ESA-listed species feeding at the ocean surface is not expected to result in an increased likelihood of injury from significant disruption of breeding, feeding, or sheltering. Firing target materials are normally retrieved before sinking and it is not reasonable to expect ingestion of these items to occur.

In conclusion, ESA-listed species are either so unlikely to ingest expended material as to be discountable, or in the case of bottom-feeding species, any ingested materials are likely to pass through the digestive tract without causing injury or any effects rising to the level of take. Therefore, the effects of ingested expended materials on ESA-listed species is either discountable, or insignificant, and not likely to adversely affect them.

6.7.6 Effects of Electromagnetic Devices

Normandeau et al. (Normandeau et al. 2011) concluded there was behavioral, anatomical, and theoretical evidence indicating cetaceans sense magnetic fields. Most of the evidence in this regard is indirect evidence from correlation of sighting and stranding locations suggesting that cetaceans may be influenced by local variation in the earth’s magnetic field (Kirschvink 1990) (Klinowska 1985) (Walker et al. 1992). Results from one study in particular showed that long-finned and short-finned pilot whales, striped dolphin, Atlantic spotted dolphin, Atlantic white-sided dolphin, fin whale, common dolphin, harbor porpoise, sperm whale, and pygmy sperm whale were found to strand in areas where the earth’s magnetic field was locally weaker than surrounding areas (negative magnetic anomaly) (Kirschvink 1990). Results also indicated that certain species may be able to detect total intensity changes of only 0.05 microtesla (Kirschvink et al. 1986). This gives insight into what changes in intensity levels some species are capable of detecting, but does not provide experimental evidence of levels to which animals may physiologically or behaviorally respond. Further, not all physiological or behavioral responses are biologically significant and rise to the level of take as defined in the ESA.

Impacts to marine mammals associated with electromagnetic fields are dependent on the animal’s proximity to the source and the strength of the magnetic field. Electromagnetic fields associated with naval training exercises and testing activities are relatively weak (only 10 percent of the earth’s magnetic field at 79 ft.), temporary, and localized. Once the source is turned off or moves from the location, the electromagnetic field is gone. A marine mammal would have to be present within the electromagnetic field (approximately 656 ft. [200 m] from the source) during the activity in order to detect it. Again, detection does not necessarily signify a significant biological response rising to the level of take as defined under the ESA. Given the small area associated with mine fields, the infrequency and short duration of magnetic energy use, the low intensity of electromagnetic energy sources, and the density of cetaceans in these areas, the likelihood of ESA-listed cetaceans being exposed to electromagnetic energy at sufficient intensities to create a biologically relevant response is so low as to be discountable. Therefore, electromagnetic energy from mine neutralization equipment is not likely to adversely affect ESA-listed cetaceans and will not be considered further in this opinion.

Sea turtles use geomagnetic fields to navigate at sea, and therefore changes in those fields could impact their movement patterns (Lohmann and Lohmann 1996a) (Lohmann and Lohmann 1996b) (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 1996a) (Lohmann and Lohmann 1996b) (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 1996a) (Lohmann and Lohmann 1996b) Lohmann et al. 1997). For example, Lohmann and Lohmann (Lohmann and Lohmann 1996a) (Lohmann and Lohmann 1996b) 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) (Lohmann et al. 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.

In a recent literature review on the effects of electromagnetic fields on marine species (Fisher and Slater 2010) elasmobranchs including scalloped hammerhead sharks were noted to have extreme sensitivity to low-frequency AC electric fields but little is known about their sensitivity to magnetic fields. Elasmobranchs, including scalloped hammerhead sharks, use natural electromagnetic fields in their daily lives and, as a result, are at a higher risk of influence from anthropogenic sources of electromagnetic energy. These species receive electrical information about the positions of their prey, the drift of ocean currents, and their magnetic compass headings. In general, elasmobranchs experience sensitivity to electromagnetic fields between 5 x 10-7 to 10-3 V/m (Fisher and Slater 2010). At this level, these species are generally attracted to the source. However, at 1 µV/cm or greater, elasmobranchs typically avoid the source (Kalmijn

1982) (Gill and Taylor 2002). There are discrepancies between the findings of Gill and Taylor (2002) and Kalmijn (1982) on the lower threshold for elasmobranchs sensitivity to electromagnetic fields. Gill and Taylor report this threshold at 5 x 10-7 V/m, while Kalmijn reports it to be 5 x 10-9 V/m. Elasmobranchs attacking submarine cables has been observed (Marra 1989). In 1982, off the coast of Massachusetts, an experiment determined the sensitivity of dogfish (*Mustelus canis*), stingray (*Urolophus halleri*), and blue shark (*P. glauca*) to electromagnetic fields. Each species attacked the electromagnetic field sources (Kalmijn 1982).

A CMACS (2003) discussion indicated that the strength of the electromagnetic fields emitted by submerged AC cables are substantially lower than those associated with the Earth's geomagnetic field. Therefore, they may be undetectable to magneto-sensitive species, such as scalloped hammerhead sharks, that are attuned to naturally occurring field strengths. While some elasmobranch species such as sharks can detect and respond to electromagnetic fields that are within the range induced by submerged power cables and other anthropogenic sources, no studies were found describing whether such levels affect the behavior of elasmobranchs under field conditions (Fisher and Slater 2010). We assume that scalloped hammerhead sharks most likely avoid sources of strong electromagnetic fields and that any potetial behavioral responses would be very minor and only occur very near the source. Therefore, we conclude the potential effect of electromagnetic fields produced during Navy training and testing is insignificant and this potential stressor is not likely to adversely affect scalloped hammerhead sharks during Navy training and testing activities.

Reef-building corals grown in an electromagnetic field generally have higher growth rates and less mortality as shown by experimental studies with *Acropora pulchra* and *Acropora yongei* (Borell et al. 2010). Electromagnetic fields presumeably aid in the accretion of calcium carbonate, allowing reef building corals to grow at a faster rate. ESA-listed corals are not expected to experience negative effects from the use of electromagnetic devices and any effects may be beneficial. For this reason, electromagnetic devices used during Navy training and testing are not likely to adversely affect ESA-listed corals.

6.7.7 Direct Physical Strike Effects from 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. Similarly, in-water devices have never been documented to strike ESA-listed corals. Therefore, the likelihood of striking a marine mammal, sea turtle, scalloped hammerhead shark, or ESA-listed coral colony is so unlikely as to be discountable. Any ESA-listed marine mammal, sea turtle, or shark species is likely to exhibit behavioral avoidance of in-water devices in the event they are within range to detect them. Any such avoidance behavior would be of short duration and intensity that it will not increase the likelihood of injury from disruption of breeding, feeding, or sheltering. The effect of any such avoidance behavior is insignificant. For these reasons, in-water devices are not likely to adversely affect ESA-listed marine mammals, sea turtles, scalloped hammerhead sharks, or colonies of *Acropora globiceps*.

Although the probability of a direct strike to reefs with ESA-listed corals is discountable, corals (including ESA-listed corals) broadcast spawn eggs and larvae into the water column where fertilization and early embryonic development occurs. The eggs, sperm, and larval stage of ESA-listed corals could remain in the water column for extended periods where they may be subjected to cavitation that are similar to natural processes. The effect from cavitation caused by in-water devices is discussed below.

Effects of Cavitation from Vessels and In-Water Devices on Juvenile Corals

Three ESA-listed species are known to occur within the immediate surroundings of Guam or the CNMI including *Acropora globiceps*, *Acropora retusa*, and *Seriatopora aculeata*. Although the direct strike of coral reefs with vessel hulls is rare and discountable, corals (including ESA-listed corals) broadcast spawn eggs and larvae into the water column where fertilization and early embryonic development occurs. In Section 6.5, we defined an individual coral as a colony and outlined the lifestages of coral that are pertinent to this assessement. Here, we address the potential effects to individuals from impacts to reproduction and recruitment from cavitation stressors.

Exposure of Juvenile Coral Life Stages to Cavitation

Three ESA-listed species are known to occur within the immediate surroundings of Guam or the CNMI including *Acropora globiceps*, *Acropora retusa*, and *Seriatopora aculeata*. Given the nature of ESA-listed coral spawning, it is reasonable to assume vessels and in-water devices will pass through water containing eggs, sperm, early embryonic stages, or planula larvae of the ESA-listed species occurring near Guam and the CNMI, where higher concentrations of vessel traffic and these life stages are likely to occur.

The eggs, sperm, and larval stage (Figure 17) of ESA-listed corals could remain in the water column for extended periods. Each individual polyp of an *Acropora* coral can produce 16 eggs and concentrations of sperm can be as high as 1 million per milliliter of seawater during spawning. Fertilized eggs develop into planula larvae within 5 days in *Acropora* species but these larvae can also remain in the water column over 200 days before settling. Although the reproductive effort and developmental timing of *Seriatopora* species are not as well known it is possible they are similar to *Acropora* species.

We assume that individuals in these life stages (eggs, sperm, early embryonic stages, or planula larvae) that occur offshore are less likely to come into contact with this stressor due to lower densities (greater volume of water), and a lower concentration of training and testing activities as compared to nearshore Guam and potentially the mouth of Apra Harbor.

Response of Corals to Cavitation

The eggs of *Acropora millepora*, a congeneric species to two of the ESA-listed species in the action area, are known to disintegrate into irregular groups or individual blastomeres when subjected to even very light shearing forces and turbulence (Heyward and Negri 2012). Under laboratory conditions these disintegrated cells commonly reorganized and continued development into eventual juveniles (Heyward and Negri 2012). Therefore, the disassociation of embryonic cells can be beneficial through the creation of more juveniles, although it is suspected others suffered direct mortality from being disassociated. In a manual for coral larvae rearing for reef rehabilitation, Guest et al. (2010), suggests rough handling of broadcast spawning coral embryos during early cell division stages (up to 36 hours post fertilization) will result in many embryo deaths or embryos being smaller than normal. Mead and Denny (1995) found turbulent water decreased successful fertilization of broadcast spawned eggs in the purple sea urchin, likely due to mechanical separation of eggs and sperm. The authors also found fertilized eggs exposed to high shear stresses of turbulent water showed abnormal development and low survival (Mead and Denny 1995). Shear stress from water turbulence has also been reported to cause increased mortality in fish eggs (Bunn et al. 2000; Eshenroder et al. 1994; Morgan et al. ; Sutherland and Ogle 1975).

Life stages subjected to cavitation from vessels and in-water devices could be deformed, die, or experience a decreased likelihood of fertilization. However, as described above, the reproductive biology of *Acropora globiceps*, and other coral species, results in prolific larval production and high natural mortality from a combination of factors including predation and dispersal to areas within the ocean without appropriate settlement habitat (e.g., deeper water, colder water, inappropriate substrate). Any anthropogenic mortality from the Navy's actions is likely to be infinitesimally small by comparison (L. Smith, personal communication, May 6, 2015) and biologically insignificant. Additionally, of the 19 threats to coral identified in the 2011 status review report of the 82 candidate coral species petitioned under the U.S. Endangered Species Act (Brainard et al. 2011a) and the top 9 threats to coral analyzed in the final rule (79 FR 53851), none include mortality of larvae by physical contact such as cavitation. While cavitation from vessels and in-water devices may result in the mortality of the developmental stages of *Acropora globiceps* (and other ESA-listed coral species), it likely would have an insignificant effect on the reproductive potential for an individual (i.e., colony) of the species or recruitment at the population level of this species. Since this level of effect is not expected to be signficant and detectable at the individual level (i.e., colony) we would not consider this effect to be a reduction in fitness of any colony of *Acropora globiceps* and thus we do not anticipate any population-level effects.

6.8 Stressors That Are Likely to Adversely Affect ESA-Listed Species

The following sections discuss stressors that are likely to adversely affect ESA-listed species.

6.8.1 Effects of Collisions with Vessels

Navy vessels could strike ESA-listed animals during the course of training and testing activities. For example, within the SOCAL Range Complex, Navy vessels conducting training and testing have struck three whales over the last 10 years and in the Hawaii Range Complex, the Navy struck two whales over the last ten years. However, these are areas where the number of military vessels is generally much higher and training and testing activities occur more often than in the MITT action area. The Navy has been training and testing in the MITT action area for many years, and there has never been a documented case of a vessel striking an ESA-listed animal.

Vessels strikes could occur from surface operations or 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 surface transits and slower for certain tactical maneuvers, and greater speeds when submerged. 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.The number of participating vessels in the action area varies based on training and testing schedules. Most activities include either one or two vessels, with an average of one vessel per activity, and last from a few hours up to 2 weeks. Multiple ships, however, can be involved with major training events, although ships can often operate for extended periods beyond the horizon and out of visual sight from each other. A majority of vessel transit in the action area will occur in concentrated locations near docks and ports, particularly in and around Apra Harbor.

The Navy employs several actions to minimize to minimize collisions between surface vessels and ESA-listed animals that might occur in the action area. These measures 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. Navy policy (Chief of Naval Operations Instruction 3100.6H) requires participating 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 cumulatively provided to NMFS on an annual basis. In addition, the Navy and NMFS have standardized regional reporting protocols for communicating to NMFS stranding coordinators information on any ship strikes as soon as possible. These communication procedures will remain in place as part of this proposed action.

6.8.1.1 Cetaceans

Vessel strikes from commercial, recreational, and Navy vessels are known to affect large whales and have resulted in serious injury and occasional fatalities to cetaceans (Lammers et al. 2003) (Laggner 2009) (Calambokidis 2012) (Douglas et al. 2008) (Berman-Kowalewski et al. 2010). Reviews of the literature on ship strikes mainly involve collisions between commercial vessels and whales (e.g., Laist et al. (2001) Jensen and Silber (2004)). The ability of any ship to detect a marine mammal and avoid a collision depends on a variety of factors, including environmental conditions, ship design, size, speed, and manning, as well as the behavior of the animal. Records of collisions date back to the early 17th century, and the worldwide number of collisions appears to have increased steadily during recent decades (Laist et al. 2001; Ritter 2012) (IWC, 2008).

Vessel speed, size and mass are all important factors in determining potential impacts of a vessel strike to marine mammals. For large vessels, speed and angle of approach can influence the severity of a strike. Based on modeling, Silber et al. (2010) found that whales at the surface experienced impacts that increased in magnitude with the ship's increasing speed. Results of the study also indicated that potential impacts were not dependent on the whale's orientation to the path of the ship, but that vessel speed may be an important factor. At ship speeds of 15 knots or higher (7.7 m/second), there was a marked increase in intensity of centerline impacts to whales. Results also indicated that when the whale was below the surface (about one to two times the vessel draft), there was a pronounced propeller suction effect. This suction effect may draw the whale into the hull of the ship, increasing the probability of propeller strikes (Silber et al. 2010).

Key points in discussion of participating vessels in relationship to potential ship strike include:

- Many military ships have their bridges positioned closer to the bow, offering better visibility ahead of the ship (compared to a commercial merchant vessel).
- There are often aircraft associated with the training or testing activity, which can more readily detect cetaceans in the vicinity of a vessel or ahead of a vessel's present course before crew on the vessel would be able to detect them.
- Military ships are generally more maneuverable than commercial merchant vessels, and if cetaceans are spotted in the path of the ship, would be capable of changing course more quickly. Military ships operate at the slowest speed possible consistent with either transit needs or training or testing needs. While minimum speed is intended as a fuel conservation measure particular to a certain ship class, secondary benefits include better ability to spot and avoid objects in the water including marine mammals. In addition, a standard operating procedure for Navy vessels is to maneuver the vessel to maintain a distance of at least 500 yd. (457 m) from any observed whale and to avoid approaching whales head-on, as long as safety of navigation is not imperiled.
- The crew size on military vessels is generally larger than merchant ships, allowing for the possibility of stationing more trained Lookouts on the bridge. At all times when vessels are underway, trained Lookouts and bridge navigation teams are used to detect objects on the surface of the water ahead of the ship, including cetaceans. Additional Lookouts, beyond those already stationed on the bridge and on navigation teams, are positioned as Lookouts during some training events.

- Lookouts receive extensive training including Marine Species Awareness Training, which instructs Lookouts to recognize marine species detection cues (e.g., floating vegetation or flocks of seabirds) as well as provides additional information to aid in the detection of cetaceans.

While it is possible for a Navy vessel to strike a cetacean during the course of training and testing activities in MITT action area, we do not believe this is reasonably certain to occur. As stated previously, the Navy has been training in the action area for years and no such incident has occurred. Additionally, the Navy employs minimization measures to reduce the likelihood for a surface vessel to strike a large whale. For these reasons, we consider the likelihood that a cetacean will be struck during Navy training and testing activities in the MITT action area so low as to be discountable. Therefore, this stressor is not likely to adversely affect ESA-listed cetaceans and will not be considered further in this opinion.

6.8.1.2 Sea turtles

Any of the sea turtles species present in the action area can occur at or near the surface of the water, and therefore may be susceptible to vessel strike. There are no reported cases of a sea turtle being struck by a Navy vessel in the MITT action area. However, 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.

As described in Table 42, densities of leatherback and loggerhead sea turtles are very low in the MITT action area (i.e., < 0.00022 animals/km^2), particularly in the inshore environment near docks and ports where most of the vessel traffic is concentrated. Hawksbill sea turtles also occur in relatively low densities (i.e., from 0.000024 to 0.1342 animals/km^2) in the MITT action area[8] (NAVFAC 2013). Given these low densities, the low amount of Navy vessel traffic that will occur in the MITT action area, and that there are no reported cases of a sea turtle being struck by a Navy vessel in the MITT action area, we consider the likelihood for an individual of any of these species to be struck during Navy training and testing activities in the MITT action area to be so low as to be discountable.

Green turtles are more common in the action area, particularly in the nearshore environment (Table 42). Therefore, this species has a much higher likelihood of encountering moving Navy vessels. It is possible that green turtles will exhibit avoidance behavior in the event they encounter a Navy vessel in transit. If this occurs, the turtle would be expected to resume normal activities shortly after the initial avoidance response. This brief avoidance behavior of green turtles will not increase the likelihood of injury from significant disruption of breeding, feeding,

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[8] According to the Navy's Marine Species Density Database, hawksbill sea turtle densities in the MITT action area range as high as 1.0734 animals/km^2, but this estimate is for nearshore areas around FDM where there is limited Navy vessel traffic.

or sheltering. Hazel et al. (2007) studied the behavioral responses of green turtles to vessels approaching at varying speeds. With the assumption that how quickly a turtle responds to an oncoming vessels is dependent on how soon the turtle can detect the oncoming vessel, they determined that the majority of turtles will likely not be able to avoid vessels traveling faster than 4 km/hr.

Navy vessel movement is expected to be most concentrated in and around Apra Harbor, where major docking facilities exist. Outside of Apra Harbor, including deep-water, offshore areas, vessel traffic is much less dense as vessels travel to locations throughout the wide action area to conduct training and testing activities. Additionally, green turtle density in offshore areas is significantly lower than in nearshore habitats. For example, per the Navy Marine Species Density Database (NAVFAC 2013), estimated green turtle density ranges as high 32.22 animals/km^2 in Apra Harbor. Alternatively, in deeper, offshore waters, green turtle occurrence is much lower, assumed to be approximately 0.39 animals/km^2, per the Navy Marine Species Density Database (NAVFAC 2013). Therefore, we expect Apra Harbor to be the only area in which sea turtle strikes are reasonably certain to occur. While Navy vessels in nearshore areas, particularly around docks and ports are expected to travel at slower speeds than in offshore environments, some green turtles in these nearshore areas are expected to be susceptible to vessel strike because some vessels in these areas are likely to be traveling at speeds greater than 4km/hr.

Previous NMFS biological opinions have relied on sea turtle stranding network data to estimate mortality from vessel strike (e.g., Hawaii Troll and Handline BiOp; Phase II Navy Hawaii-Southern California Training and Testing). Although little information exists to quantify this impact, vessel collision was implicated as the cause of three green turtle strandings in Apra Harbor between November 2002 and April 2008 (DAWR unpublished data). We are also aware of a sea turtle stranding that occurred in Apra Harbor in May 2016. The stranded turtle showed evidence of vessel strike, but did not occur in close temporal or spatial proximity to Navy training and testing activities (Julie Rivers personal communication to Eric MacMillan; March 3, 2017). Though available information does not allow us to estimate the percentage of vessel struck turtles that are observed stranded in Apra Harbor, we expect this observed stranding rate to be fairly high due to the human population along the Harbor and given that it 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), we assume 40 percent of vessel struck sea turtles will strand (NMFS 2008b). Based on the observed stranding rate in Apra Harbor from November 2002 to April 2008 (i.e., three observed strandings with evidence of vessel strike)[9], we estimate a total of 8 turtles

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[9] Note: Sea turtle stranding data for this region is not readily available. The sea turtle stranding data from November 2002 to April 2008 represents the longest-term sea turtle stranding dataset available to us at the time of this consultation. We are aware of the sea turtle vessel strike that occurred in Apra Harbor in 2016 (as stated above). However, because sea turtle stranding data from years 2009 to 2015 or for any other strandings that occurred in

were struck by vessels during this time period (3 turtles / 0.4 = 7.5 turtles). This comes to an annual strike rate of 1.36 green sea turtles per year (7.5 turtles / 5.5 years = 1.36 turtles per year).

Navy vessels account for a large percentage of the vessel traffic in Apra Harbor, but substantial traffic also occurs from recreational and commercial vessels. The of Port of Guam had 635 non-Navy vessel calls in 2013, while the Navy reports 389 Navy vessel calls in 2014 (1,024 total vessel calls). Therefore, Navy vessels account for approximately 38 percent of the large vessel traffic in and out of the Port of Guam on an annual basis. Since we do not have data to indicate how many smaller, recreational boats transit through Apra Harbor, we conservatively estimate that Navy vessel traffic accounts for 38 percent of the vessel movement within Apra Harbor. Therefore, we expect that on average, the Navy will strike up to 1 green turtle per year (1.36 turtles per year x 0.38 = 0.52; rounded up to 1 turtle).

Response of sea turtles to vessel strike

Ship strikes are known to injure and kill sea turtles (Work et al. 2010b). Stranding networks that keep track of sea turtles that wash up dead or injured have consistently recorded vessel propeller strikes as a cause or possible cause of death (Chaloupka et al. 2008a). We conclude that collisions with vessels would likely result in blunt trauma, lacerations, and mortality.

6.8.1.3 Scalloped Hammerhead Shark – Indo-West Pacific DPS

Scalloped hammerhead sharks in the action area may encounter moving Navy vessels. However, it is expected that this highly mobile and fast swimming species will exhibit evasive behavior in the event they encounter a Navy vessel at close range and will be able to avoid being struck. For this reason, the likelihood of a scalloped hammerhead shark being struck by a Navy vessel is discountable. It is expected that sharks will return to normal behavior shortly after avoiding the oncoming vessel as with other natural evasive responses where the perceived threat has passed. Any temporary avoidance behavior will not increase the likelihood of injury from significant disruption of breeding, feeding, or sheltering and the effect of such temporary avoidance is insignificant. For these reasons, scalloped hammerhead sharks are not likely to be adversely affected by vessel strike.

6.8.1.4 Corals

Though unlikely (with the exception of Amphibious Assault and Amphibious Raid Operations, as described below), Navy vessels may contact the benthos, including coral reefs, during training and testing activities. If a Navy vessel struck an ESA-listed coral colony, this could result in colony fragmentation, injury, or mortality. However, we do not believe vessel strike of coral colonies is reasonably certain to occur. In nearshore habitats, where a large percentage of the vessel traffic is concentrated, most Navy vessel movements are confined to established channels,

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2016 was not available and this analysis required a vessel strike rate over a period of time, we determined that the sea turtle stranding dataset from 2002 to 2008 represented the best available data for the analysis in this consultation.

ports, or transit lanes. These areas are frequently used by vessels (Navy and non-Navy), and have water deep enough to where vessels do not contact coral reef or other substrate types. For movement in nearshore habitats not in established channels or transit lanes, vessels are not expected to contact coral reef habitats because they are equipped with appropriate navigational equipment and vessel operators actively avoid running aground (this excludes Amphibious Assault and Amphibious Raid operations). Additionally, Navy vessels are not expected to strike coral reefs in offshore habitats because waters are deep enough to where vessels will not contact coral reef or other substrate types. For these reasons, we consider the likelihood of an ESA-listed coral being struck by a Navy vessel during training and testing activities (excluding Amphibious Assault and Amphibious Raid Operations, which is addressed below) to be so low as to be discountable.

Navy vessels could contact benthic habitat in areas adjacent to corals in the surf zone during Amphibious Assault and Amphibious Raid operations. However, due to the minimization measures described below, vessels are expected to avoid direct contact with coral colonies. Amphibious Assault and Amphibious Raid training activities would be conducted in the nearshore area, including the surf zone, up to the high tide line. Prior to conducting any amphibious over-the-beach training activity with larger amphibious vehicles such as Landing Craft Air Cushions (LCACs) or Amphibious Assault Vehicles (AAVs), surveys would be conducted to identify and designate boat lanes and beach landing areas that are clear of coral, hard bottom substrate, and obstructions. Based on the surveys, if the beach landing area and boat lane is clear, the activity could be conducted, and crews would follow procedures to avoid obstructions to navigation, including coral reefs. However, if the survey indicates there is any potential for impacts on corals or hard bottom substrate, the Navy will coordinate with applicable resource agencies before conducting the activity. Further, LCAC and AAV landing and departure activities would be scheduled at high tide, decreasing the likelihood that a vessel would contact coral colonies. If necessary to traverse over areas with coral or other hard-bottom substrate, LCACs and AAVs would hover over, but avoid direct contact with such habitats. Surveys would not be necessary for beach landings with small boats, such as RHIBs. We do not expect small boats to contact coral reefs during this activity because it would be a navigational and safety hazard for vessels of this size to strike exposed or near-surface substrate. For reasons described above, we consider the likelihood of an ESA-listed coral being struck by a Navy vessel during Amphibious Assault and Amphibious Raid operations to be so low as to be discountable.

Precision anchoring will occur in Apra Harbor and other existing anchorages in the Mariana Islands. Except in existing anchorages and near-shore training areas around Guam and within Apra Harbor, Navy vessels will not anchor within the anchor swing diameter of surveyed shallow coral reefs, live hardbottom, articificial reefs, or shipwrecks. Recent studies have identified evidence of anchor and/or anchor chain damage to coral in Apra Harbor, including the formation of a rubble field on the southern side of the floating dry dock. Movement of mooring chains on the southern side of the floating dry dock has produced a significant rubble field, although mooring chains on the northern (outer) side of the floating dry dock do not appear to have caused similar damage (DoN 2010b). Available data suggests ESA-listed corals do not occur at existing Navy anchorages in Apra Harbor or other locations. Therefore, we consider the likelihood of Navy anchorage impacting ESA-listed coral species to be so low as to be discountable.

6.8.2 Overview of Effects of Acoustic Stressors

The U.S. Navy grouped approximately 300 individual sources of underwater acoustic sound or explosive energy, into 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; and
- 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 takes remain within the overall analyzed and authorized limits. This flexibility is required to support evolving Navy training and testing requirements, which are linked to real world events.

There are two primary types of source classes: "Impulsive" and "Non-impulsive" acoustic. A description of each source classification analyzed for marine mammals is provided in Table 43 and Table 44 below. 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 kHz, 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).

Table 43. Training and Testing Non-Impulsive Acoustic Sources Analyzed for Marine Mammals

Source Class Category	Source Class	Description
Low-Frequency (LF): Sources that produce low-frequency (less than 1 kHz) signals	LF4	Low-frequency sources equal to 180 dB and up to 200 dB
	LF5	Low-frequency sources less than 180 dB
	LF6	Low-frequency sonar currently in development (e.g., ASW sonar associated with the LCS)
Mid-Frequency (MF): Tactical and non-tactical sources that produce mid-frequency (1–10 kHz) signals	MF1	Hull-mounted surface ship sonar (e.g., AN/SQS-53C and AN/SQS-60)
	MF2	Hull-mounted surface ship sonar (e.g., AN/SQS-56)
	MF3	Hull-mounted submarine sonar (e.g., AN/BQQ-10)
	MF4	Helicopter-deployed dipping sonar (e.g., AN/AQS-22 and AN/AQS-13)
	MF5	Active acoustic sonobuoys (e.g., DICASS)
	MF6	Active underwater sound signal devices (e.g., MK-84)
	MF8	Active sources (greater than 200 dB) not otherwise binned
	MF9	Active sources (equal to 180 dB and up to 200 dB)
	MF20	Active sources (greater than 160 dB, but less than 180 dB) not otherwise binned
	MF11	Hull-mounted surface ship sonars with an active duty cycle greater than 80%
	MF12	High duty cycle – variable depth sonar

High-Frequency (HF) and Very High-Frequency (VHF): Tactical and non-tactical sources that produce high-frequency (greater than 10 kHz but less than 200 kHz) signals	HF1	Hull-mounted submarine sonar (e.g., AN/BQQ-10)
	HF4	Mine detection, classification, and neutralization sonar (e.g., AN/SQS-20)
	HF5	Active sources (greater than 200 dB)
	HF6	Active sources (equal to 180 dB and up to 200 dB)
Anti-Submarine Warfare (ASW): Tactical sources such as active sonobuoys and acoustic countermeasures systems used during the conduct of anti-submarine warfare testing activities	ASW1	Mid-frequency DWADS
	ASW2	Mid-frequency MAC sonobuoy (e.g., AN/SSQ-125)
	ASW3	Mid-frequency towed active acoustic countermeasure systems (e.g., AN/SLQ-25)
Torpedoes (TORP): Source classes associated with the active acoustic signals produced by torpedoes	TORP1	Lightweight torpedo (e.g., MK-46, MK-54, or Anti-Torpedo Torpedo)
	TORP2	Heavyweight torpedo (e.g., MK-48)
Acoustic Modems (M): Systems used to transmit data acoustically through water	M3	Mid-frequency acoustic modems (greater than 190 dB)
Swimmer Detection Sonar (SD): Systems used to detect divers and submerged swimmers	SD1	High-frequency sources with short pulse lengths, used for the detection of swimmers and other objects for the purpose of port security.

Notes: dB = decibels, DICASS = Directional Command Activated Sonobuoy System, DWADS = Deep Water Active Distributed System, kHz = kilohertz, LCS = Littoral Combat Ship, MAC = Multi-static Active Coherent

Table 44. Training and Testing Explosive Sources Analyzed for Marine Mammals

Source Class	Representative Munitions	Net Explosive Weight (lb.)[1]
E1	Medium-caliber projectiles	0.1–0.25
E2	Medium-caliber projectiles	0.25–0.5
E3	Large-caliber projectiles	> 0.5–2.5
E4	Improved Extended Echo Ranging Sonobuoy	> 2.5–5.0
E5	5-inch projectiles	> 5–10
E6	15 lb. shaped charge	> 10–20
E8	250 lb. bomb	> 60–100
E9	500 lb. bomb	> 100–250
E10	1,000 lb. bomb	> 250–500
E11	650 lb. mine	> 500–650
E12	2,000 lb. bomb	> 650–1,000

[1] Net Explosive Weight refers to the amount of explosives; the actual weight of a munition may be larger due to other components

Activities and acoustic source classes modeled for sea turtles are provide in the Table 45 below:

Table 45. Activities and Active Acoustic Sources Modeled and Quantitatively Analyzed by the U.S. Navy for Acoustic Impacts on Sea Turtles (Reference FEIS/OEIS V2, June 2014)

Activity	Acoustic Source Class
Training Activity	
Joint Expeditionary Exercise	ASW2, ASW3, MF1, MF12, MF2, MF3, MF4, MF5
Joint Multi-Strike Group Exercise	ASW2, ASW3, ASW4, MF1, MF11, MF12, MF2, MF3, MF4, MF5
Marine Air Ground Task Force Exercise (Amphibious)	ASW3, MF1, MF12, MF2, MF3, MF4
TRACKEX/TORPEX – MPA Advanced Extended Echo Ranging Sonobuoys	ASW2
TRACKEX/TORPEX – Surface	ASW3, MF1, MF11, MF12, MF2, TORP1
Testing Activity	
Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft	ASW2, MF5, MF6, TORP1
ASW Mission Package Testing	ASW1, ASW3, LF6, MF12, MF4, MF5
At-Sea Sonar Testing	ASW1, LF5, M3, MF1, MF10, MF11, MF3, MF9
Pierside Integrated Swimmer Defense	LF4, MF8, SD1

There are also non-impulsive sources with characteristics that are not anticipated to result in takes of marine mammals under the MMPA or ESA. 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, but are qualitatively analyzed in the MITT FEIS/OEIS, May 2015.

Understanding the number of munitions detonating in water is critical to assessing potential impacts from acoustic stressors, potential strike and fragments resulting from exploded munitions. Table 46 below provides the number and source of these munitions.

Table 46. Proposed Annual Number of Impulsive Source Detonations During Training and Testing in the MITT Action Area

Explosive Class	Net Explosive Weight (NEW)	Annual In-Water Detonations
E1	(0.1 lb. – 0.25 lb.)	10,140
E2	(0.26 lb. – 0.5 lb.)	106
E3	(>0.5 lb. – 2.5 lb.)	932
E4	(>2.5 lb. – 5 lb.)	420
E5	(>5 lb. – 10 lb.)	684
E6	(>10 lb. – 20 lb.)	76
E8	(>60 lb. – 100 lb.)	16
E9	(>100 lb. – 250 lb.)	4
E10	(>250 lb. – 500 lb.)	12
E11	(>500 lb. – 650 lb.)	6
E12	(>650 lb. – 2,000 lb.)	184

Understanding the frequency and duration of active sonar sources is imperative in our risk analysis for stressors resulting from non-impulsive sound sources. Table 47 below provide the annual hours of these sources in the MITT action area.

Table 47. Annual hours and items of non-impulsive sources used during training and testing within the MITT Action Area

Source Class Category	Source Class	Average Annual Use
Low-Frequency (LF) Sources that produce signals less than 1 kilohertz (kHz)	LF4	123 hours
	LF5	11 hours
	LF6	40 hours
Mid-Frequency (MF) Active sources from 1 to 10 kHz	MF1	1,872 hours
	MF2	625 hours
	MF3	192 hours
	MF4	214 hours
	MF5	2,588 hours
	MF6	33 hours
	MF8	123 hours
	MF9	47 hours
	MF10	231 hours
	MF11	324 hours
	MF12	656 hours
High-Frequency (HF) and Very High-Frequency (VHF) Tactical and non-tactical sources that	HF1	113 hours
	HF4	1,060 hours

Source Class Category	Source Class	Average Annual Use
produce signals greater than 10kHz but less than 200kHz	HF5	336 hours
	HF6	1,173 hours
Anti-Submarine Warfare (ASW) Tactical sources used during anti-submarine warfare training and testing activities	ASW1	144 hours
	ASW2	660 hours
	ASW3	3,935 hours
	ASW4	32 hours
Torpedoes (TORP) Source classes associated with active acoustic signals produced by torpedoes	TORP1	115 hours
	TORP2	62 hours
Acoustic Modems (M) Transmit data acoustically through the water	M3	112 hours
Swimmer Detection Sonar (SD) Used to detect divers and submerged swimmers	SD1	2,341 hours
Airgun (AG)1	AG	308 hours

Notes: dB = decibels, DICASS = Directional Command Activated Sonobuoy System, DWADS = Deep Water Active Distributed System, kHz = kilohertz, LCS = Littoral Combat Ship, MAC = Multi-static Active Coherent

6.8.3 Effects of Non-Impulsive Acoustic Stressors - Cetaceans

For this consultation, we considered exposure estimates from the Navy Acoustic Effects Model at two output points for marine mammals. First, the total number of ESA-listed species (animats) that would be exposed to acoustic sources greater than 120dB prior to the application of a dose-response curve or criteria. We term these the “unprocessed” estimates. This estimate is the number of times individual animats or animals are likely to be exposed to the acoustic environment that is a result of training exercises and testing activities, regardless of whether they are “taken” as a result of that exposure. In most cases, the number of animals “taken” by an action would be a subset of the number of animals that are exposed to the action because (1) in some circumstances, animals might not respond to an exposure and (2) some responses may be negative for an individual animal without constituting a form of “take” (for example, some physiological stress responses only have fitness consequences when they are sustained and would only constitute a “take” as a result of cumulative exposure).

The second set (Table 50) of predicted exposures (“processed”) of listed species were generated and processed using dose-response curves and criteria for temporary and permanent threshold shift developed by the Navy and NMFS’ Permits Division for the purpose of identifying harassment pursuant to the MMPA. Neither sets of exposure estimates, the unprocessed or processed, consider standard mitigation actions that NMFS’ Permits Division would require under the MMPA rule to avoid marine mammals or that the Navy proposes for marine mammals, nor did the estimates consider any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft.

Our estimation of take is derived from the processed exposure estimates on an annual basis, cumulatively over the five-year period, and cumulatively for the reasonably foreseeable future to derive a final estimate of anticipated levels of take by training activity and species.

6.8.3.1 Exposures to Non-impulsive Acoustic Stressors During Training

The following sections discuss the exposure of ESA-listed species to non-impulsive acoustic stressors.

6.8.3.1.1 Unprocessed Exposures to Non-Impulsive Acoustic Stressors During Training

Navy model output estimates of whales that will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities above received levels of 120 dB throughout the year are summarized in Table 48 and discussed below. Exposures at levels above 120 dB may result in behavioral or physiological responses (see Section 6.2).

Table 48. Model-Predicted Exposures of Cetaceans to Non-Impulsive Sound During Training Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
blue whale	195
fin whale	208
humpback whale	10,862
sei whale	3,378
sperm whale	14,813

Blue whale. The model output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 195 blue whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Thirty-seven (37) percent of the estimated 195 exposures to blue whales during training activities will result from TRACKEX/TORPEX – Surface exercises while 13 percent will result from Joint Expeditionary Exercises and 10 percent result from Joint Multi-Strike Group Exercises.

Fin whale. The model output estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 208 fin whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Forty (40) percent of the estimated 208 exposures to fin whales during training activities will result from TRACKEX/TORPEX – Surface exercises while 12 percent will result from Joint Expeditionary Exercises and 12 percent result from Joint Multi-Strike Group Exercises.

Humpback whale. The model output estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 10,862 humpback whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Forty-nine (49) percent of the estimated 10,862 exposures to humpback whales during training activities will result from TRACKEX/TORPEX – Surface exercises while 10 percent will result from Joint Expeditionary Exercises and 12 percent result from Joint Multi-Strike Group Exercises.

Sei whale. The model output estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 3,378 sei whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Forty-seven (47) percent of the estimated 3,378 exposures to sei whales during training activities will result from TRACKEX/TORPEX – Surface exercises while 10 percent will result from Joint Expeditionary Exercises and 13 percent result from Joint Multi-Strike Group Exercises.

Sperm whale. The model output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 14,813 sperm whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Forty-nine (49) percent of the estimated 14,813 exposures to sperm whales during training activities will result from TRACKEX/TORPEX – Surface exercises while 11 percent will result from Joint Expeditionary Exercises and 13 percent result from Joint Multi-Strike Group Exercises.

6.8.3.1.2 Unprocessed Exposures to Non-Implusive Acoustic Stressors During Testing

Navy model output estimates of whales that will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year are summarized in Table 49 and discussed below.

Table 49. Model-Predicted Exposures of Cetaceans to Non-Impulsive Sound During Testing Actvities

Species	Number of Model-Predicted Exposures >120 dB Per Year
blue whale	102
fin whale	115
humpback whale	4,179
sei whale	1,330
sperm whale	5,941

Blue whale. The model output estimates that blue whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 102 blue whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Forty-seven (47) percent of the estimated 102 exposures to blue whales during testing activities will result from Anti-submarine Warfare Tracking Test – Marine Patrol Aircraft while 33 percent will result from ASW Mission Package Testing and 15 percent result from At-Sea Sonar Testing.

Fin whale. The model output estimates that fin whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 115 fin whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Fifty (50) percent of the estimated 115 exposures to fin whales during training activities will result from Anti-submarine Warfare Tracking Test – Marine Patrol Aircraft while 32 percent will result from ASW Mission Package Testing and 13 percent result from At-Sea Sonar Testing.

Humpback whale. The model output estimates that humpback whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 4,179 humpback whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Thirty-seven (37) percent of the estimated 4,179 exposures to humpback whales during training activities will result from Anti-submarine Warfare Tracking Test – Marine Patrol Aircraft while 46 percent will result from ASW Mission Package Testing and 14 percent result from At-Sea Sonar Testing.

Sei whale. The model output estimates that sei whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 1,330 sei whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Thirty-seven (37) percent of the estimated 1,330 exposures to sei whales during training activities will result result from Anti-submarine Warfare Tracking Test – Marine Patrol Aircraft while 46 percent will result from ASW Mission Package Testing and 13 percent result from At-Sea Sonar Testing.

Sperm whale. The model output estimates that sperm whales will be exposed to sonar and other non-impulsive acoustic stressors associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 5,941 sperm whale exposure events annually to non-impulsive sounds associated with annual training at levels above 120 dB SPL. Thirty-two (32) percent of the estimated 5,941 exposures to sperm whales during training activities will result result from Anti-submarine Warfare Tracking Test – Marine Patrol Aircraft while 50 percent will result from ASW Mission Package Testing and 14 percent result from At-Sea Sonar Testing.

6.8.3.1.3 Processed Exposures to Non-Impulsive Acoustic Stressors During Training and Testing

Humpback whale, blue whale, fin whale, and sei whale are all mysticetes. Research and observations show that if mysticetes are exposed to sonar or other active acoustic sources they may react in a number of ways depending on the characteristics of the sound source, their experience with the sound source, and whether they are migrating or on seasonal grounds (i.e., breeding or feeding grounds). Reactions may include alerting, breaking off feeding dives and surfacing, diving or swimming away, or no response at all (Southall et al. 2007b) (Southall et al. 2012) (Goldbogen et al. 2013; Melcon et al. 2012). Additionally, migrating mysticetes (such as humpback whales moving through the action area) may divert around sound sources that are located within their path or may ignore a sound source depending on the context of the exposure.

Animals that experience TTS may have reduced ability to detect relevant sounds such as predators, prey, or social vocalizations until their hearing recovers. Recovery from a threshold shift (i.e., TTS; temporary partial hearing loss) can take a few minutes to a few days depending on the severity of the initial shift. Threshold shifts do not necessarily affect all hearing frequencies equally, so some threshold shifts may not interfere with an animal's ability to hear biologically relevant sounds. For exposures resulting in TTS, long-term consequences for individuals or populations would not be expected.

Predicted effects (processed values) from sonar and other active acoustic sources on ESA-listed marine mammals are shown in Table 50. Only behavioral responses (non-TTS effects) and TTS effects are anticipated. Based on the exposure levels modeled, no PTS level detections are expected.

Table 50. Predicted Effects (processed values) on Endangered Species Act-Listed Marine Mammals from Annual Training and Testing Use of Sonar and Other Active Acoustic Sources

Species	Non-TTS (Behavioral)	TTS	PTS
Mysticetes			
Blue whale	3	25	0
Fin whale	4	24	0
Humpback whale	181	679	0
Sei whale	61	258	0
Odontocetes			
Sperm whale	452	54	0

6.8.3.2 Response of ESA-Listed Species to Non-Impulsive Acoustic Stressors

The following sections discuss the response of ESA-listed species to non-impulsive acoustic stressors.

6.8.3.2.1 Blue Whale

Temporary Threshold Shift - 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. 2007b)) suggest that most TTS occurs in the frequency

range of the source up to one octave higher than the source (with the maximum TTS at ½ octave above). The more powerful mid-frequency sources used have center frequencies between 3.5 kHz and 8 kHz and the other unidentified mid-frequency sources are, by definition, less than 10 kHz, which suggests that TTS induced by any of these mid frequency sources would be in a frequency band somewhere between approximately 2 and 20 kHz. As discussed above, blue whales are low-frequency hearing specialists, producing low-frequency vocalizations that include moans in the range from 12.5 Hz to 400 Hz, with dominant frequencies from 16 Hz to 25 Hz. Therefore, even though recent studies (discussed below) indicate that blue whales hear and respond to mid-frequency sounds, it is unlikely that TTS caused by mid-frequency active sonar transmissions would interfere with an animal's ability to hear vocalizations or most other biologically important sounds. There are fewer hours of high frequency source use and the sounds would attenuate more quickly, plus they have lower source levels, but if an animal were to incur TTS from these sources, it would cover a higher frequency range (sources are between 20 kHz and 100 kHz, which means that TTS could range up to 200 kHz; however, high frequency systems are typically used less frequently and for shorter time periods than surface ship and aircraft mid-frequency systems, so TTS from these sources is even less likely).

2. Degree of the shift (i.e., how many dB is the sensitivity of the hearing reduced by)—Generally, both the degree of TTS and the duration of TTS will be greater if the marine mammal is exposed to a higher level of energy (which would occur when the peak dB level is higher or the duration is longer). The threshold for the onset of TTS was discussed previously in this document. An animal would have to approach closer to the source or remain in the vicinity of the sound source appreciably longer to increase the received SEL, which would be difficult considering the lookouts, the sightability of blue whales and other listed mysticetes, and the nominal speed of an active sonar vessel (10 to 15 knots). In the TTS studies, some using exposures of almost an hour in duration or up to 217 SEL, most of the TTS induced was 15 dB or less, though Finneran et al. (2007) induced 43 dB of TTS with a 64-second exposure to a 20 kHz source. However, mid-frequency active sonar emits a nominal ping every 50 seconds, and incurring those levels of TTS is highly unlikely.

3. Duration of TTS (recovery time)—In the TTS laboratory studies, some using exposures of almost an hour in duration or up to 217 SEL, almost all individuals recovered within 1 day (or less, often in minutes), though in one study (Finneran et al. 2007), recovery took 4 days. Based on the range of degree and duration of TTS reportedly induced by exposures to non-pulse sounds of energy higher than that to which freeswimming marine mammals in the field are likely to be exposed during mid-frequency/high-frequency active sonar training exercises in the action area, it is unlikely that blue whales or other listed cetaceans would ever sustain a TTS from MFAS that alters their sensitivity by more than 20 dB for more than a few days (and any incident of TTS would likely be far less severe due to the short duration of the majority of the exercises and the speed of a typical vessel). Also, because of the short distance within which animals would need to approach the sound source, it is unlikely that animals would be exposed to the levels necessary to induce TTS in subsequent time periods such that their recovery is impeded.

In addition, the mitigation zones prescribed in the final MMPA rule encompass the ranges to PTS for the most sensitive marine mammal functional hearing group, which is usually the high-frequency cetacean hearing group. Therefore, the mitigation zones are even more protective for blue whales and other low-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 to occur.

Therefore, for blue whales and other listed mysticetes analyzed in this opinion, even though the modeled TTS estimates may change from year to year over estimates provided here, 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 MITT 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. Additionally, though the frequency range of TTS that marine mammals might sustain would overlap with some of the frequency ranges of their vocalization types, the frequency range of TTS from mid-frequency active sonar (the source from which TTS would most likely be sustained because the higher source level and slower attenuation make it more likely that an animal would be exposed to a higher received level) would not usually span the entire frequency range of one vocalization type, much less span all types of vocalizations or other critical auditory cues. If impaired, marine mammals would typically be aware of their impairment and would likely be able to implement behaviors to compensate. Given the brief amount of time blue whales are expected to experience TTS, it is unlikely to significantly impair their ability to communicate, forage, or breed and will not have fitness level consequences at the individual or population level. Navy monitoring of Navy-wide activities since 2006 has documented hundreds of thousands of marine mammals on the range complexes and there are only two instances of overt behavioral change that have been observed and there have been no demonstrable instances of injury to marine mammals as a result of non-impulsive acoustic sources.

Behavioral Responses - As discussed, the Navy uses the behavioral response function to quantify the number of behavioral responses that would qualify as Level B behavioral harassment under the MMPA. As the statutory definition is currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated using the behavioral response function do not differentiate between the different types of potential reactions. Nor do the estimates provide information regarding the potential fitness or other biological consequences of the reactions on the affected individuals. We therefore consider the available scientific evidence to determine the likely nature of the modeled blue whale behavioral responses and the potential fitness consequences for affected individuals.

While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on blue whale hearing so we assume that blue whale vocalizations are partially representative of their hearing sensitivities. Blue whales are not likely to respond to high-frequency sound sources associated with the proposed training exercises and testing activities because of their hearing sensitivities. Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007b; Stafford and Moore 2005a), recent research has indicated blue whales not only hear mid-frequency active sonar transmissions, in some cases they respond to those transmissions (Goldbogen et al. 2013; Melcon et al. 2012; Southall et al. 2011a).

As summarized in Section 6.2.10 and Section 6.2.10.2, blue whales may hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context (Goldbogen et al. 2013; Melcon et al. 2012). However, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational MFA sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). As stated in Goldbogen et al. (2013) when summarizing the response of blue whales to simulated MFA sonar, "We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure."

Melcon et al. (2012) reported that blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low frequency calls (D calls) usually associated with feeding behavior. However, they they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior and indicated that implications of the documented responses are unknown. Goldbogen et al. (2013) speculated that if the documented temporary behavioral responses interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since unconsumed prey would likely still be available in the environment in most cases following the cessation of acoustic exposure. Additionally, it is worth noting here that most instances of exposure would be brief.

Moreover, there are important differences between the acoustic sources used in the author's Controlled Exposure Experiments and Navy sonar. As discussed previously, 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. (Southall et al. 2007b), 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 MITT 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. 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 MITT action area for decades, and there is no evidence that the activities have caused blue whales or other listed species evaluated in this opinion to abandon important habitat or any other severe adverse impacts.

Further, recent evidence indicates that the Eastern North Pacific blue whale population, which includes blue whales in Southern California, has likely reached carrying capacity (Monnahan et al. 2014). As stated previously, Navy training and testing activities in Southern California, as with similar activities in the MITT action area, include the use of MFA sonar, and have been ongoing for decades. Therefore, any potential temporary behavioral impacts on blue whales from the use of MFA during Navy training and testing activities in the MITT action area are also likely to have not prevented the recovery of the blue whales throughout its range. The TTS and behavioral response estimates may change from year to year, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the 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) (Read et al. 2014b). It is difficult to assess the effects of sounds individually and cumulatively on marine species because a number of factors can influence these effects including: indirect effects, age class, prior experience, behavioral state at the time of exposure, and that responses may be influenced by other non-sound related factors (Ellison et al. 2011; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Williams et al. 2014) (Read et al. 2014b). 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.

Non-impulsive acoustic stressors from Navy training activities conducted during five-year period of proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of blue whale populations. We anticipate temporary behavioral responses and TTS, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. 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 (i.e., training activities similar to those conducted in the MITT action area including the use of MFA sonar) in the Hawaii and Southern California Training Ranges.

6.8.3.2.2 Fin Whale

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to fin whales. Fin whales are not likely to respond to high-frequency sound sources associated with the proposed training and testing activities because of their hearing sensitivities. While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on fin whale hearing so we assume that fin whale vocalizations are partially representative of their hearing sensitivities. Those vocalizations include a variety of sounds described as low frequency moans or long pulses in the 10 Hz to 100 Hz band (Edds 1988; Thompson and Friedl 1982b; Watkins 1981a). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15 Hz to 40 Hz range. Ketten (1997a) reports the frequencies of maximum energy between 12 Hz and 18 Hz. Short sequences of rapid calls in the 30 to 90 Hz band are associated with animals in social groups (Clark personal observation and McDonald personal communication cited in Ketten 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. 2013) (Southall et al. 2011a), which have similar auditory physiology compared to fin whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. However, as described in Section 6.8.3.2.1, regarding the response of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual whales. Without more specific information, we assume the response of fin whales to TTS to be similar to that of blue whales. Additionally, several fin whales were tagged during the Southern California-10 BRS and no obvious responses to the controlled exposure to a mid-frequency sound source were detected by the visual observers or in the initial tag analysis (Southall et al. 2011a).

Further, work by Moore and Barlow (2011) indicate that since 1991, there is strong evidence of increasing fin whale abudance in the California Current area, which includes the Southern California Range Complex, an area in which Navy training and testing activities, including the use of MFA sonar, have been ongoing for decades. The authors predict continued increases in fin whale numbers over the next decade in that area, and that perhaps fin whale densities are reaching “current ecosystem limits.” This indicates that any potential temporary behavioral impacts on fin whales from the use of MFA sonar during Navy training and testing activities have not prevented fin whale numbers from increasing and potentially nearing ecosystem limits in Southern California. Since training and testing activities in the MITT action area are similar to those in Southern California, we would expect the effects to be similar in the MITT action area and thus not a limiting factor in the recovery of fin whales. The TTS and behavioral response estimates may change over baseline conditions from year to year, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitiness of affected individuals, is not expected to change.

Non-impulsive acoustic stressors from Navy training exercises and testing activities conducted during the remainder of the five-year year period of the MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability or recovery of fin whale populations. We anticipate temporary behavioral responses and instances of TTS, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.8.3.2.3 Humpback Whale

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 from the Western North Pacific DPS, 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. (2006b) conducted field investigations of humpback whale songs that led these investigators to conclude that humpback whales have an upper frequency limit reaching as high as 24 kHz. Based on this information, it is reasonable to assume that the MFA sonar the U.S. Navy uses during active sonar training exercises and testing activities in the MITT action area is 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 6.8.3.2.1 regarding responses of blue whales to MFA sonar, we expect any behavioral response to be temporary and to not result in any fitness consequences to individual humpback whales. Without more specific information, we assume the response of humpback whales to TTS to be similar to that of blue whales.

Humpback whales have been known to react to low frequency industrial noises at estimated received levels of 115 to 124 dB (Malme et al. 1985), and to calls of other humpback whales at received levels as low as 102 dB (Frankel et al. 1995). Malme et al. (1985) found no clear response to playbacks of drill ship and oil production platform noises at received levels up to 116 dB re 1 Pa. Studies of reactions to airgun noises were inconclusive (Malme et al. 1985). Frankel and Clark (1998) showed that breeding humpbacks showed only a slight 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 Ocean Acoustic Waveguide Remote Sensing 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 TTS and behavioral response estimates may change from year to year over baseline conditions, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training and testing activities conducted during the five-year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability of Western North Pacific DPS humpack whales or adversely impact species recovery. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of

TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.8.3.2.4 Sei Whale

Except as provided herein, our analysis of the modeled TTS and behavioral responses for blue whales applies equally to sei whales. Like blue and fin whales, sei whales are not likely to respond to high-frequency sound sources associated with the proposed training exercises and testing activities because of their hearing sensitivities. As discussed in the *Status of Listed Resources* section of this opinion, we have no specific information on the hearing sensitivity of sei whales. Based on their anatomical and physiological similarities to both blue and fin whales, we assume that the hearing thresholds of sei whales will be similar as well and will be centered on low-frequencies in the 10 to 200 Hz.

Results of studies on blue whales (Goldbogen et al. 2013) (Southall et al. 2011a), which have similar auditory physiology compared to sei whales, indicate that some individuals hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context. However, as described in Section 6.8.3.2.1, 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 6.8.3.2.1). The TTS and behavioral response estimates may change from year to year over baseline conditions, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitiness of affected individuals, is not expected to change.

Acoustic stressors from Navy training exercises and testing activities conducted during the five-year period of the MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability of sei whale populations or their ability to recover. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.8.3.2.5 Sperm Whale

Temporary Threshold Shift (TTS) - As mentioned previously, TTS can last from a few minutes to days, be of varying degree, and occur across various frequency bandwidths, all of which determine the severity of the impacts on the affected individual, which can range from minor to more severe. The TTS sustained by an animal is primarily classified by three characteristics:

1. Frequency—Available data (of midfrequency hearing specialists exposed to mid- or high-frequency sounds; (Southall et al. 2007b)) suggest that most TTS occurs in the frequency

range of the source up to one octave higher than the source (with the maximum TTS at ½ octave above). The more powerful mid-frequency sources used have center frequencies between 3.5 kHz and 8 kHz and the other unidentified mid-frequency sources are, by definition, less than 10 kHz, which suggests that TTS induced by any of these mid frequency sources would be in a frequency band somewhere between approximately 2 and 20 kHz. 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 1993b). Ridgway and Carder (Ridgway and Carder 2001) measured low-frequency, high amplitude clicks with peak frequencies at 500 Hz to 3 kHz from a neonate sperm whale.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. 2007b). 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 may change from year to year over baseline conditions including potential increases of exposure, 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 MITT action area.

Behavioral Responses - 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 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. 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 1999a; Watkins 1985), pingers (Watkins and Schevill 1975a), the Heard Island Feasibility Test (Bowles et al. 1994), and the Acoustic Thermometry of Ocean Climate (Costa et al. 1998). Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders (Watkins and Schevill 1975a). Goold (1999a) reported six sperm whales that were driven through a narrow channel using ship noise, echosounder, and fish finder emissions from a flotilla of 10 vessels. Watkins and Schevill (1975a) showed that sperm whales interrupted click production in response to pinger (6 to 13 kHz) sounds. They also stopped vocalizing for brief periods when codas were being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

Sperm whales 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. (2000b) noted that sighting frequency did not differ significantly among the different acoustic levels examined in the northern Gulf of Mexico, contrary to what Mate et al. (1994) reported. Sperm whales may also have responded to seismic airgun sounds by ceasing to call during some (but not all) times when seismic pulses were received from an airgun array >300 km away (Bowles et al. 1994).

A study offshore of northern Norway indicated that sperm whales continued to call when exposed to pulses from a distant seismic vessel. Received levels of the seismic pulses were up to 146 dB re 1 μPa peak-to-peak (Madsen et al. 2002). Similarly, a study conducted off Nova Scotia that analyzed recordings of sperm whale sounds at various distances from an active seismic program did not detect any obvious changes in the distribution or behavior of sperm whales (McCall-Howard 1999). 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 1975a). They also stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995). Because they spend large amounts of time at depth and use low frequency sound sperm whales are likely to be susceptible to low frequency sound in the ocean (Croll et al. 2001b). 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. (1995c) 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 may change from year to year over baseline conditions, which could mean an increase in the number of individual animals exposed per year or an increase in the number of times per year some animals are exposed. However, the severity of individual responses, and the consequences of those responses on the fitiness of affected individuals, is not expected to change.

Non-Impulsive acoustic stressors from Navy training exercises and testing activities conducted during the five-year period of the proposed MMPA Rule and into the reasonably foreseeable future are not likely to reduce the viability of sperm populations or their ability to recovery. We anticipate behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses or instances of TTS to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.

6.8.4 Effects of Non-Impulsive Acoustic Stressors - Sea Turtles

For this consultation, we considered exposure estimates from the Navy Acoustic Effects Model at several output points for sea turtles.

6.8.4.1 Unprocessed Exposures to Non-Impulsive Acoustic Stressors During Training Activities

Sea turtles that reside in the action area may be exposed several times throughout the year to sound from sonar and other active acoustic sources. Exposures to sonar and other active acoustic sources in open water areas would be intermittent and geographically variable. The Navy model output indicates that green, hawksbill, loggerhead, and leatherback sea turtles would be exposed to sonar at levels that may rise to the level of behaviroral take and TTS.

Navy model output estimates of sea turtes that will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year are summarized in Table 51 and discussed below.

Table 51. Model-Predicted Exposures of Sea Turtles to Non-Impulsive Sound During Training Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
Green Turtle	12,564
Hawksbill Turtle	383
Leatherback Turtle	99
Loggerhead Turtle	94

Green sea turtle. The NAEMO provided an unprocessed estimate of 12,564 green sea turtle exposure events annually to non-impulsive sounds associated with annual training at levels greater than 120 dB SPL. Eighty-two (82) percent of the estimated 12,564 green sea turtle exposures will result from Marine Air Ground Task Force Exercise (Amphibious) while 11 percent will result from TRACKEX/TORPEX – Surface.

Hawksbill sea turtle. The NAEMO provided an unprocessed estimate of 383 hawksbill sea turtle exposure events annually to non-impulsive sounds associated with annual training at levels greater than 120 dB SPL. Forty-seven (47) percent of the estimated 383 hawksbill sea turtle exposures will result from Marine Air Ground Task Force Exercise (Amphibious) while 19 percent will result from TRACKEX/TORPEX – Surface, 17 percent from Joint Multi-Strike Group Exercises and 12 percent from Joint Expeditionary Exercises .

Leatherback sea turtle. The NAEMO provided an unprocessed estimate of 99 leatherback sea turtle exposure events annually to non-impulsive sounds associated with annual training at levels greater than 120 dB SPL. Thirty (30) percent of the estimated 99 leatherback sea turtle exposures will result from Joint Multi-Strike Group Exercises while 28 percent will result from Joint Expeditionary Exercises, 20 percent from TRACKEX/TORPEX – Surface and 15 percent from Marine Air Ground Task Force Exercise (Amphibious).

Loggerhead sea turtle. The NAEMO provided an unprocessed estimate of 94 loggerhead sea turtle exposure events annually to non-impulsive sounds associated with annual training at levels greater than 120 dB SPL. Twenty-six (26) percent of the estimated 94 loggerhead sea turtle exposures will result from Joint Multi-Strike Group Exercises while 26 percent will result from Joint Expeditionary Exercises, 22 percent from TRACKEX/TORPEX – Surface and 19 percent from Marine Air Ground Task Force Exercise (Amphibious).

6.8.4.2 Unprocessed Exposures to Non-impulsive Acoustic Stressors During Testing Activities

Navy model output estimates of sea turtes that will be exposed to sonar and other non-impulsive acoustic stressors associated with training activities throughout the year are summarized in Table 52 and discussed below.

Table 52. Model-Predicted Exposures of Sea Turtles to Non-Impulsive Sound During Testing Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
Green Turtle	20,848
Hawksbill Turtle	800
Leatherback Turtle	144
Loggerhead Turtle	131

Green sea turtle. The NAEMO provided an unprocessed estimate of 20,848 green sea turtle exposure events annually to non-impulsive sounds associated with annual testing activities at levels greater than 120 dB SPL. Eighty-five (85) percent of the estimated 20,848 green sea turtle exposures will result from ASW Mission Package Testing while 7 percent will result from At-Sea Sonar Testing.

Hawksbill sea turtle. The NAEMO provided an unprocessed estimate of 800 hawksbill sea turtle exposure events annually to non-impulsive sounds associated with annual testing activities at levels greater than 120 dB SPL. Sixty-nine (69) percent of the estimated 800 hawksbill sea turtle exposures will result from ASW Mission Package Testing while 10 percent will result from At-Sea Sonar Testing and 15 percent from Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft.

Leatherback sea turtle. The NAEMO provided an unprocessed estimate of 144 leatherback sea turtle exposure events annually to non-impulsive sounds associated with annual testing activities at levels greater than 120 dB SPL. Fifty-five (55) percent of the estimated 144 leatherback sea turtle exposures will result from ASW Mission Package Testing while 39 percent will result from Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft.

Loggerhead sea turtle. The NAEMO provided an unprocessed estimate of 131 loggerhead sea turtle exposure events annually to non-impulsive sounds associated with annual testing activities at levels greater than 120 dB SPL. Sixty (60) percent of the estimated 131 loggerhead sea turtle exposures will result from ASW Mission Package Testing while 34 percent will result from Anti-Submarine Warfare Tracking Test – Maritime Patrol Aircraft.

6.8.4.3 Processed Exposures to Non-Impulsive Acoustic Stressors During Training and Testing Activities

Processed exposures of sea turtles to acoustic stressors from training and testing activities predicted by the NAEMO are shown below in Table 53. The exposure estimates represent the total number of exposures and not necessarily the number of individuals exposed, as a single individual may be exposed multiple times over the course of a year. The predicted acoustic impacts do not take into account avoidance behavior or mitigation measures, such as establishing shut-down zones for certain sonar systems. As described in section 6.3.8, the TTS and PTS criteria used for sea turtles in the NAEMO modeling was 17 dB below what it was supposed to be. Therefore, the estimates provided are overestimates. Additionally, the Navy model may over-predict acoustic impacts because it does not consider avoidance and the criteria to predict impacts.

Table 53. Annual Total Model-Predicted Impacts on Sea Turtles from Training and Testing Activities Using Sonar and Other Active Acoustic Sources

Sea Turtle Species	Temporary Threshold Shift	Permanent Threshold Shift
Green sea turtle*	251	0
Hawksbill sea turtle	17	0
Leatherback sea turtle	12	0
Loggerhead sea turtle	15	0

*See section 6.10.6.1 for discussion on allocating these estimated instances of TTS to turtles from specific green sea turtle DPSs.

6.8.4.4 Response of Sea Turtles to Non-impulsive Acoustic Stressors During Training and Testing Activities

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Ketten 2006) (Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a) (Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003a). Additionally, they are not known to produce sounds underwater for communication.

Further, although the information on the hearing capabilities of sea turtles is limited, the information available suggests that the auditory capabilities of sea turtles are centered in the low-frequency range (<2 kHz) (Bartol et al. 1999b; Dow Piniak et al. 2012; Lenhardt et al. 1983; Lenhardt et al. 1994b; 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. 2014b).

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 MITT 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 (as defined in section 3.3 of this opinion) we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for the energy expended during evasion and any potential lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Similarly, to result in significant fitness consequences we would have to assume that an individual turtle detects and responds to the acoustic source, and that it could not compensate for the energy expended during evasion. 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 are unlikely to lead to fitness consequences or implications for sea turtle populations.

Although we assume that sea turtles in the vicinity of sonar might experience a TTS, 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. (Piniak et al. 2012) 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. 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 will not result in fitness consequences to individuals or the populations to which they belong.

Non-impulsive acoustic stressors from Navy training and testing activities conducted during the five-year period of the MMPA rule and into the reasonably foreseeable future are likely to cause TTS to sea turtles in the MITT action area. However, there is no evidence that TTS results in energetic effects to individual sea turtles or would be likely to significantly reduce the viability of the population these individuals represent. Additionally, based on the analysis presented above, we do not anticipate any PTS, injury or mortality of sea turtles from non-impulsive acoustic stressors.

6.8.5 Effects of Non-Impulsive Acoustic Stressors - Scalloped Hammerhead Shark – Indo-Pacific DPS

This section discusses the effects of non-impulsive acoustic stressors on scalloped hammerhead sharks. Exposure of scalloped hammerhead sharks to acoustic stressors could not be quantitatively assessed due to limited information on species distribution and density in the action area. Scalloped hammerhead sharks are likely only capable of detecting sounds between 20 Hz and 1 kHz with the highest sensitivity in the lower end of this spectrum (Casper et al. 2003; Casper and Mann 2006; Myrberg Jr. 2001). As such, this species is only anticipated to detect and potentially be exposed to non-impulsive acoustic stressors below 1 kHZ, which the Navy defines as low-frequency sources. No low frequency sources are used during training activities. As part of training activities, in port maintenance of sonar systems occurs within Inner Apra Harbor where anecdotal evidence suggests a scalloped hammerhead nursery may be present. However, the in-port and at-sea maintenance of surface ship sonar systems training are mid-frequency hull-mounted sonar systems. The in-port and at-sea maintenance of submarine sonar systems are mid- and high-frequency sonar systems. Therefore, sonar from these activities is unlikely to be detected by scalloped hammerhead sharks.

A combined total of 174 hours of low-frequency acoustic sources will be used per year during testing activities. Low-frequency sonar use may be part of the following testing activities: at-sea sonar testing, pierside integrated swimmer defense testing, and anti-submarine warfare mission package testing. Of these activities, the pierside integrated swimmer defense testing will occur in Inner Apra Harbor, where exposure to scalloped hammerhead sharks is most likely. Pierside integrated swimmer defense testing includes non-impulsive acoustic energy from 180 to 200 dB at frequencies less than 1 kHz, suggesting that if juvenile or adult scalloped hammerhead sharks are present within Inner Apra Harbor, they could be exposed to non-impulsive acoustic energy at frequencies they can detect. Pierside integrated swimmer defense testing will occur for 28 days per year with intermittent periods of use of low-frequency sound.

The duration and intensity of low-frequency non-impulsive acoustic stressors and the lack of a swim bladder will likely minimize the effect this stressor has on scalloped hammerhead sharks. In studies of rainbow trout, low-frequency active sonar at 193 dB re 1 μPa^2 for up to 648 seconds did not cause mortality although up to a 25 dB threshold shift was observed when trout received a 400 Hz exposure (Popper et al. 2007). The threshold shift lasted at least 48 hours post-exposure. Exposure at this level also caused a rapid swimming behavioral response at the beginning of treatment. Although a threshold shift was observed, it is noteworthy no noticeable morphological effects were observed to ear structures and fish appeared healthy and active throughout the end of the experimental period (1 week) (Popper et al. 2007).

Several shark species, including the oceanic silky shark (*Carcharhinus falciformis*) and coastal lemon shark (*Negaprion brevirostris*), have been observed withdrawing from pulsed low-frequency sounds played from an underwater speaker (Klimley and Myrberg 1979; Myrberg et al. 1978). Lemon sharks exhibited withdrawal responses to pulsed low to mid-frequency sounds (500 to 4,000 Hz) raised 18 dB at an onset rate of 96 dB/sec to a peak amplitude of 123 dB RL from a continuous level, just masking broadband ambient noise (Klimley and Myrberg 1979). In their study, lemon sharks withdrew from artificial sounds which included 10 pulses/second (continuous), 10 pulses/second (intermittent), and 15 to 7.5 decreasing pulses/second (intermittent). Myrberg et al. (1978) reported that a silky shark withdrew 10 m (33 ft) from a speaker broadcasting a 150 to 600 Hz sound with a sudden onset and a peak SL of 154 dB. These sharks avoided a pulsed low frequency attractive sound when its sound level was abruptly increased by more than 20 dB. Other factors enhancing withdrawal were sudden changes in the spectral or temporal qualities of the transmitted sound. These results do not rule out that such sounds may have been harmful to them after habituation; the tests were not designed to examine that point. Klimley (unpublished data) also noted the increase in tolerance of lemon sharks during successive sound playback tests. The pelagic whitetip (*Carcharhinus longimanus*) also showed a withdrawal response during limited tests (Myrberg et al. 1978).

Some sharks are attracted to pulsing low-frequency sounds. Myrberg (2001) stated that sharks have demonstrated highest sensitivity to low-frequency sound (40 to 800 Hz). Free-ranging sharks are attracted to sounds possessing specific characteristics including irregular pulsed, broadband frequencies below 80 Hz and transmitted suddenly without an increase in intensity thus resembling a struggling fish. However, these signals are substantially different from the low-frequency active sonar signals produced during Navy testing activities.

The precise expected response of scalloped hammerhead sharks to low-frequency acoustic energy is not completely understood due to a lack of sufficient experimental and observational data for this species. However, given the signal type and level of exposure to the low frequency signals used in pierside swimmer defense testing activities and the lack of swim bladders in scalloped hammerhead sharks as discussed above, we do not expect significant responses (including TTS, PTS, injury or mortality) to low-frequency acoustic energy from pierside integrated swimmer defense testing. The most likely response of scalloped hammerhead sharks exposed to low-frequency sonar are minor temporary changes in their behavior including increased swimming rate, avoidance of the sound source, attraction to the sound source, or changes in orientation to the sound source. The available information suggests that scalloped hammerhead sharks are either not likely to respond to low frequency sonar or are not likely to measureably respond in ways that would create the likelihood of injury by annoying the animal to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Any behavioral responses to low frequency active sonar are expected to be temporary (e.g., a startle response, brief avoidance behavior) and we do not expect these reactions to have any measurable effects on any individual’s fitness. We expect individuals that exhibit a temporary behavioral response will return to baseline behavior immediately following exposure to the sonar signal. Therefore, the effects of non-impulsive acoustic stressors on ESA-listed scalloped hammerhead sharks is insignificant and not likely to adversely affect them.

6.8.6 Effects of Non-Impulsive Acoustic Stressors – *Acropora globiceps*

Adult coral colonies are not biologically capable of detecting noise except as vibrations of water particles. The only known auditory sensing capabilities for coral is the response of free-swimming coral larvae to underwater sounds produced by reef fish and crustaceans, as reported by Vermeij et al. (2010). The authors reported that some species of coral larvae detect reef sounds and then show an attraction response to the sounds generated on the reefs. However, potential interference in the ability of coral larvae to detect reef sounds would be temporary, lasting only the duration that the sonar source is turned on and is in the vicinity of the larval coral. Exposures and potential masking would be brief. We do not expect these brief interruptions to inhibit the ability of coral larvae to detect reef habitat. Therefore, the effects of non-impulsive acoustic stressors on *Acropora globiceps* are insignificant and non-impulsive acoustic stressors are not likely to adversely affect this species.

6.8.7 Effects of Impulsive Acoustic Stressors – Cetaceans

As mentioned above, we considered exposure estimates from the Navy Acoustic Effects Model at two output points for marine mammals. First, the total number of ESA-listed species (animats) that would be exposed to acoustic sources greater than 120 dB prior to the application of a dose-response curve or criteria. We term these the “unprocessed” estimates. This estimate is the number of times individual animats or animals are likely to be exposed to the acoustic environment that is a result of training exercises and testing activities, regardless of whether they are “taken” as a result of that exposure. In most cases, the number of animals “taken” by an action would be a subset of the number of animals that are exposed to the action because (1) in some circumstances, animals might not respond to an exposure and (2) some responses may be negative for an individual animal without constituting a form of “take” (for example, some physiological stress responses only have fitness consequences when they are sustained and would only constitute a “take” as a result of cumulative exposure).

The second set of exposure estimates (“processed”) of listed species were generated and processed using dose-response curves and criteria for temporary and permanent threshold shift developed by the Navy and NMFS’ Permits Division for the purpose of identifying harassment pursuant to the MMPA and account for any mitigation and avoidance behaviors that may reduce instances of exposure at levels that might result in injury including PTS. The unprocessed impulsive sound exposure estimates do not consider standard mitigation actions that NMFS’ Permits Division would require under the MMPA rule to avoid marine mammals or that the Navy proposes for marine mammals, nor did the unprocessed estimates consider any avoidance responses that might be taken by individual animals once they sense the presence of Navy vessels or aircraft.

The exposure and response analysis presented in this opinion considers these exposure estimates on an annual basis, cumulatively over the five-year period, and cumulatively for the reasonably foreseeable future to derive a final estimate of anticipate levels of take by training activity and species.

Approximately 12,500 explosives are used during training and testing activities per year and occur in areas designated for use of explosives within the action area. Approximately 10,000 (or 80 percent) of the explosives used in the action area are in source class E1 (0.1 to 0.25 lb. NEW). Most detonations would occur beyond 12 nm from shore, minimizing impacts near nesting beaches or coastal habitats for sea turtles. A small number of near-shore (within 3 nm) training activities could occur, potentially exposing some sea turtles approaching nesting beaches to impulse sounds over a short duration if the training occurred during nesting season or close to sea turtles nearshore habitats. In water training and testing activities using relatively low NEW explosives (up to 10 lb. NEW) will occur at UNDET sites within Apra Harbor and the Piti Mine Neutralization Site and higher NEW explosives (up to 20 lb. NEW) at the Agat Bay Mine Neutralization Site. Additionally, explosives may impact nearshore habitats around FDM.

However, the terrain of FDM does not provide any nesting beaches, therefore, effects are not expected.

6.8.7.1 Unprocessed Exposure to Impulsive Acoustic Stressors During Training Activities

Navy model output estimates of cetaceans that will be exposed to impulsive acoustic stressors associated with training activities throughout the year are summarized in Table 54 and discussed below.

Table 54. Model-Predicted, Unprocessed Exposures of Cetaceans to Impulsive Sound During Training Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
blue whale	112
fin whale	76
humpback whale	1,704
sei whale	562
sperm whale	1,747

Blue whale. The model estimates that blue whales will be exposed to impulsive acoustic stressors from explosions associated with training activities throughout the year. The NAEMO provided an unprocessed estimate of 112 blue whale exposure events annually to impulsive sounds associated with annual training at levels above 120 dB SPL. Ninety-six (96) percent of the estimated 112 unprocessed exposures to blue whales during training activities will result from BOMBEX [A-S].

Fin whale. The model estimates that fin whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 76 fin whale exposure events annually to impulsive sounds associated with annual training at levels above 120 dB SPL. Eighty-eight (88) percent of the estimated 76 unprocessed exposures to fin whales during training activities will result from BOMBEX [A-S].

Humpback whale. The model estimates that humpback whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 1,704 humpback whale exposure events annually to impulsive sounds associated with annual training at levels above 120 dB SPL. Eighty-two (82) percent of the estimated 1,704 unprocessed exposures to humpback whales during training activities will result from BOMBEX [A-S].

Sei whale. The model estimates that sei whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 562 sei whale exposure events annually to impulsive sounds associated with annual training at levels above 120 dB. Seventy-nine (79) percent of the estimated 562 unprocessed exposures to sei whales during training activities will result from BOMBEX [A-S]. Approximately 12 percent of exposures will result from GUNEX [S-S] Ship – Medium Caliber.

Sperm whale. The model estimates that sperm whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of sperm 1,747 whale exposure events annually to impulsive sounds associated with annual training at levels above 120 dB. Fifty-two (52) percent of the estimated 1,747 unprocessed exposures to sperm whales during training activities will result from BOMBEX [A-S]. Approximately 21 percent of exposures will result from GUNEX [A-S] - Medium Caliber and 10 percent from GUNEX [S-S] Ship – Medium Caliber.

6.8.7.2 Processed Exposure to Impulsive Acoustic Stressors during Training Activities

There are limitations to the data used in the NAEMO, and the results must be interpreted with consideration for these known limitations. Output (Table 54 and Table 55) from the NAEMO relies heavily on the quality of both the input parameters and impact thresholds and criteria. When there was a lack of definitive data to support an aspect of the modeling (such as lack of well described diving behavior for all marine species), conservative assumptions believed to overestimate the number of exposures have been chosen:

- Marine mammals (virtual animals called “animats” in the model) are modeled as being underwater, facing the source and therefore always predicted to receive the maximum sound level at their position within the water column (e.g., the model does not account for conditions such as body shading, porpoising out of the water, or an animal raising its head 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 toward the rear or side of an animal (Kastelein et al. 2005; Mooney et al. 2008; Popov and Supin 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 due to 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 which are implemented during many training and testing activities were not considered in the model. In reality, sound-producing activities

would be reduced, stopped, or delayed if marine mammals are detected within the mitigation zones around sound sources.

Observation of marine mammals prior to or during a detonation (explosion) would stop or delay the event until the area was cleared as described the Navy's Standard Operating Procedures, Mitigation, and Monitoring. This delay of the sound-producing activity (explosion) would reduce actual marine mammal sound exposures to zero at the level that rises to take pursuant to the ESA. Exposure to sound at lower levels may result in minor behavioral responses.

While there are model-predicted unprocessed exposures (Table 54 and Table 55) of cetaceans to impulsive sound sources (explosions), Navy processing of these exposures reduced exposures to impulsive sources to zero. Therefore, any potential effects from impulsive sound sources on cetaceans are considered discountable.

6.8.7.3 Unprocessed Exposure to Impulsive Acoustic Stressors During Testing Activities

Navy model output estimates of cetaceans that will be exposed to impulsive acoustic stressors associated with testing activities throughout the year are summarized in Table 55 and discussed below.

Table 55. Model-Predicted, Unprocessed Exposures of Cetaceans to Impulsive Sound During Testing Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
blue whale	6
fin whale	7
humpback whale	213
sei whale	71
sperm whale	284

Blue whale. The model output estimates that blue whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of six blue whale exposure events annually to impulsive sounds associated with annual testing at levels above 120 dB. Sixty-nine (69) percent of the estimated 6 exposures to blue whales during testing activities will result from Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft while 19 percent will result from Torpedo (Explosive) Testing and 11 percent from ASUW Mission Package Testing.

Fin whale. The model output estimates that fin whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of seven fin whale exposure events annually to impulsive sounds associated with annual testing at levels above 120 dB. Forty-five (45) percent of the estimated 7 exposures to fin whales during testing activities will result from Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft while 31 percent will result from Torpedo (Explosive) Testing and 17 percent from ASUW Mission Package Testing.

Humpback whale. The model output estimates that humpback whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 213 humpback whale exposure events annually to impulsive sounds associated with annual testing at levels above 120 dB. Seventy-one (71) percent of the estimated 213 exposures to humpback whales during testing activities will result from Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft while 16 percent will result from Torpedo (Explosive) Testing and 6 percent from ASUW Mission Package Testing.

Sei whale. The model output estimates that sei whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of 71 sei whale exposure events annually to impulsive sounds associated with annual testing at levels above 120 Db SPL. Seventy-seven (77) percent of the estimated 71 exposures to sei whales during testing activities will result from Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft while 13 percent will result from Torpedo (Explosive) Testing and 8 percent from ASUW Mission Package Testing.

Sperm whale. The model output estimates that sperm whales will be exposed to impulsive acoustic stressors from explosions associated with testing activities throughout the year. The NAEMO provided an unprocessed estimate of sperm 284 whale exposure events annually to impulsive sounds associated with annual testing at levels above 120 dB SPL. Eighty-one (81) percent of the estimated 284 exposures to sperm whales during testing activities will result from Anti-Submarine Warfare Tracking Test - Maritime Patrol Aircraft while 7 percent will result from Torpedo (Explosive) Testing, 5 percent from MCM Mission Package Testing, and 4 percent from ASUW Mission Package Testing.

6.8.7.4 Processed Exposure to Impulsive Acoustic Stressors during Testing Activities

As discussed above for training activities, while there are model-predicted unprocessed exposures (Table 54 and Table 55) of cetaceans to impulsive sound sources (explosions), Navy processing of these exposures reduced exposures to impulsive sources to zero. Therefore, any potential effects from impulsive sound sources on cetaceans are considered discountable.

6.8.7.5 Response of Cetaceans to Impulsive Acoustic Stressors During Training and Testing

Since we do not anticipate exposures of cetaceans to impulsive acoustic stressors, we also do not anticipate any responses rising to the level of take. Therefore, we conclude that the potential for effects to cetaceans resulting from impulsive acoustic stressors is sufficiently low to be discountable and impulsive acoustic stressors are not likely to adversely affect ESA-listed cetaceans.

6.8.8 Effects of Impulsive Acoustic Stressors - Sea Turtles

For this consultation, we considered exposure estimates from the Navy Acoustic Effects Model at several output points for sea turtles.

6.8.8.1 Unprocessed Exposures to Impulsive Acoustic Stressors During Training Activities

Navy model estimates of sea turtles that will be exposed to impulsive acoustic stressors associated with training activities throughout the year are summarized in Table 56 and discussed below.

Table 56. Model-Predicted Unprocessed Exposures of Sea Turtles to Impulsive Sound During Training Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
Green Turtle	2,630
Hawksbill Turtle	210
Leatherback Turtle	67
Loggerhead Turtle	121

Green sea turtle. The NAEMO provided an unprocessed estimate of 2,360 green sea turtle exposure events annually to impulsive sounds associated with annual training activities at levels greater than 120 dB SPL. Thirty-two (32) percent of the estimated 2,630 green sea turtle exposures will result from BOMBEX [A-S] while 26 percent will result from Underwater Demolition Qualification / Certification, 17 percent from Mine Neutralization – EOD and 11 percent Maritime Security Operations.

Hawksbill sea turtle. The NAEMO provided an unprocessed estimate of 210 hawksbill sea turtle exposure events annually to impulsive sounds associated with annual training activities at levels greater than 120 dB SPL. Forty-five (45) percent of the estimated 210 hawksbill sea turtle exposures will result from BOMBEX [A-S] while 19 percent will result from Underwater Demolition Qualification / Certification, 13 percent from Mine Neutralization – EOD.

Leatherback sea turtle. The NAEMO provided an unprocessed estimate of 67 leatherback sea turtle exposure events annually to impulsive sounds associated with annual training activities at levels greater than 120 dB SPL. Eighty-five (85) percent of the estimated 67 leatherback sea turtle exposures will result from BOMBEX [A-S] while 7 percent will result from GUNEX [S-S] Ship – Medium Caliber.

Loggerhead sea turtle. The NAEMO provided an unprocessed estimate of 121 loggerhead sea turtle exposure events annually to impulsive sounds associated with annual training activities at levels greater than 120 dB SPL. Fifty-six (56) percent of the estimated 121 loggerhead sea turtle exposures will result from BOMBEX [A-S] while 28 percent will result from GUNEX [S-S] Ship – Medium Caliber.

6.8.8.2 Unprocessed Exposures to Impulsive Acoustic Stressors During Testing Activities

Navy model output estimates of sea turtes that will be exposed to impulsive acoustic stressors associated with testing activities throughout the year are summarized in Table 57 and discussed below.

Table 57. Model-Predicted Exposures of Sea Turtles to Impulsive Sound During Testing Activities

Species	Number of Model-Predicted Exposures >120 dB Per Year
Green Turtle	489
Hawksbill Turtle	52
Leatherback Turtle	29
Loggerhead Turtle	32

6.8.8.3 Processed Exposures to Impulsive Acoustic Stressors During Training and Testing

Processed exposures of sea turtles to impulsive acoustic stressors from training and testing activities predicted by the NAEMO are shown below in Table 58. The exposure estimates represent the total number of exposures and not necessarily the number of individuals exposed, as a single individual may be exposed multiple times over the course of a year. The predicted acoustic impacts do not take into account avoidance behavior or mitigation measures. Following processing, Navy modeling results (indicate 14 TTS-level exposures, 1 PTS-level exposure, 4 exposures resulting in slight lung injury, and 2 exposures resulting in mortality combined for green and hawksbill turtles. No TTS, PTS, injury or mortality was predicted by the NAEMO modeling for loggerheads, or leatherback turtles.

Table 58. Annual Model-Predicted Impacts on Sea Turtles from Explosives Used During Training and Testing Activities

Sea Turtle Species	TTS	PTS	GI Tract Injury	Slight Lung Injury	Mortality
Green sea turtle*	**11**	**1**	0	**3**	**1**
Hawksbill sea turtle	**3**	0	0	**1**	**1**
Loggerhead sea turtle	0	0	0	0	0
Leatherback sea turtle	0	0	0	0	0

*See section 6.10.6.1 for discussion on allocating these estimated instances of TTS to turtles from specific green sea turtle DPSs.

6.8.8.4 Estimated Instances of Behavioral Harrassment of Sea Turtles from Impulsive Acoustic Stressors

In addition to Navy-modeled estimates for TTS, PTS, slight lung injury, and mortality from impulsive acoustic stressors, we calculated the number of modeled (unprocessed) instances of behavioral harassment of sea turtle species. To do this, we summed the number of modeled exposures greater than 175 dB (See Section 6.3.6) and subtracted instances of estimated TTS, PTS, slight lung injury, and mortality to provide an estimate of exposures that may result in more than minor behavioral responses potentially rising to the level of “take” pursuant to the ESA. These unprocessed estimates do not take into account U.S. Navy standard operating procedures and mitigation activities that might reduce actual instances of behavioral response. We also acknowledge that individuals experiencing TTS, PTS, or slight lung injury likely also experience some form of behavioral response associated with these responses; however, we do not double count these instances to remain consistent with our analysis of TTS, PTS, injury where only the highest level response is counted towards take.

Table 59. Annual Estimated Behavioral Harrassment of Sea Turtles from Training and Testing Activities Using Impulsive Acoustic Sources

Sea Turtle Species	Behavioral Harrassment
Green sea turtle	1,837
Hawksbill sea turtle	129
Leatherback sea turtle	49
Loggerhead sea turtle	54

*See section 6.10.6.1 for discussion on allocating these estimated instances of behavioral harrassment to turtles from specific green sea turtle DPSs.

6.8.8.5 Response of Sea Turtles to Impulsive Acoustic Stressors During Training and Testing

Predicted impacts to sea turtles from impulsive acoustic stressors range from behavioral harassment to mortality. Most detonations would occur beyond 12 nm from shore, which minimizes the impacts to sea turtles near nesting beaches or coastal habitats.

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.

Slight lung injury was also predicted to occur. Slight lung injury is defined as having a zero percent mortality rate and being completely recoverable (Finneran and Jenkins 2012). 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 long-term fitness consequences to any individuals, we do not expect population-level consequences to result from instances of slight lung injury.

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. (Piniak et al. 2012) 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. 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 would be conservative for sea turtles.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003a). 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.

Some sea turtles beyond the ranges of the TTS or PTS-level effects are expected to behaviorally react if they hear a detonation. A range of effects could occur at these lower exposure levels including masking, temporary habitat displacement, or short term behavioral responses (e.g., a startle response, changes in respiration, alteration of swim speed, or direction). For example, if a sea turtle hears multiple detonations in a short period, such as during gunnery, firing, or sonobuoy exercises, it may react by temporarily avoiding the area. The response of a sea turtle to an explosion from Navy training and testing activities will depend on 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 explosion and whether it is perceived as approaching or moving away could also affect the way a sea turtle responds.

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 an intermittent sound source (e.g., explosives). 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. Intermittent explosive use will not result in prolonged periods of time where masking could occur, reducing the likelihood of the proposed action causing masking that could result in negative fitness impacts to ESA-listed sea turtles. For this reason, the effect of any masking that could be caused by explosive detonations is insignificant. Therefore, masking is not likely to adversely affect ESA-listed species.

In instances where a sea turtle avoids the area where detonations are occurring, this is expected to result in an energy expenditure to move away from the detonations, and the potential for lost feeding or resting opportunities. However, any instances of disturbance are expected to be temporary in nature, with the animal returning to the area shortly after detonations cease. Because most activities would consist of a limited number of detonations and exposures would not occur over long durations, there would be an opportunity to recover from any incurred energetic cost. Area avoidance could also result in lost feeding or resting opportunities. However, to result in fitness consequences for the animal, we would have to assume that an individual turtle 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, if an animal’s rest was disrupted, we would expect the individual would be able to resume resting immediately after the detonations ceased or rest in alternative locations once the animal moves from the area. For these reasons, temporary habitat displacement of sea turtles from impulsive acoustic stressors is unlikely to lead to fitness consequences to individual sea turtles or long-term consequences for the ESA-listed sea turtles considered in this opinion.

Similarly behavioral responses that do not result in an animal leaving the area could result in temporary disruptions to important behaviors including feeding and resting. However, most often these would be no more than startle responses with the animal resuming normal behaviors immediately following the sound exposure (i.e., seconds). To result in fitness consequences for the animal, we would have to assume that an individual turtle could not compensate for lost feeding opportunities by feeding shortly after cessation of acoustic exposure, or feeding at a later time. There is no indication this is the case, particularly since food sources would still be available in the environment immediately after the detonation occurs. Similarly, if an animal’s rest was disrupted, we would expect the individual would be able to resume resting immediately after the detonations ceased. For these reasons, behavioral responses of sea turtles to impulsive acoustic stressors are unlikely to lead to fitness consequences to individual sea turtles or long-term consequences for the population.

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 (e.g., gas embolism) in sea turtles following capture in trawls or gillnets, with a higher incidence of decompression sickness 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 decompression sickness-like symptoms and fitness consequences. However, it should be noted that this is the first, and to our knowledge, only study that has documented decompression sickness-like symptoms in sea turtles. Previous research has suggested sea turtles are protected against decompression sickness through anatomical, physiological, and behavioral adaptations (Berkson 1967; Castellini 2012; Fossette et al. 2010b; Lutcavage and Lutz 1997; Piantadosi and Thalmann 2004). Additionally, in Garcia-Parraga et al. (2014), the animals were unable to control their ascent. In a behavioral response to an acoustic stressor, the animal would be able to control its ascent. Given this uncertainty in the available literature and the lack of evidence that this sort of extreme behavioral avoidance reaction would be expected, we believe the likelihood of such an incident occurring is discountable. For this reason, decompression sickness is not likely to adversely affect ESA-listed sea turtles and we do not consider it further in this opinion.

6.8.9 Effects of Impulsive Acoustic Stressors - Scalloped Hammerhead Shark – Indo-West Pacific DPS

The following section discusses the effects of impulsive acoustic stressors on scalloped hammerhead sharks.

6.8.9.1 Exposure to Impulsive Acoustic Stressors During Training

Agat Bay Mine Neutralization Site and Outer Apra Harbor UNDET (Figure 5) support activities using explosives up to 20 lb. and 10 lb. NEW devices, respectively. Explosives used at the Piti Floating Mine Neutralization Site are also limited to 10 lb. NEW. In total, approximately 260 explosives are used annually in shallow water and coastal areas (e.g., Apra Harbor, Agat Bay, Piti Point) per year (Figure 5). The scalloped hammerhead shark is primarily a shallow water, coastal species. Underwater detonations will primarily occur during the day when this species is more likely to be closer to shore. Additionally, Navy divers involved with underwater detonation at these sites 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.

The density of scalloped hammerhead sharks in the shallow, nearshore waters of the action area are not well understood but anecdotal evidence suggests inner Apra Harbor may serve as nursery habitat for this species (Miller et al. 2014). If Apra Harbor acts as a nursery for this species, juveniles may seasonally occur in substantial densities and it can also be reasonably expected adult males and females would frequently move between Apra Harbor and nearshore areas outside the harbor. One of the underwater detonations sites is within Apra Harbor itself, Agat Bay is south of Apra Harbor, and the Piti site is immediately north of Apra Harbor.

Although sound thresholds capable of inducing behavioral responses and injury in scalloped hammerhead sharks have not been determined, it is expected they will be less susceptible than marine mammals and fish species with swim bladders. Using the formula Young (1991) for effects of underwater detonations to fish without a swim bladder, it is estimated 90 percent of scalloped hammerhead sharks would likely survive at a range of 2.2 meters (7.3 feet) or further from a 10 lb. NEW explosive underwater detonation and 2.8 meters (9.2 feet) or further from a 20 lb. NEW underwater detonation. Therefore, the mortality zone for a 10 lb. NEW charge would be approximately 15.21 m^2 (167.42 ft^2) and approximately 24.63 m^2 (267 ft^2) for a 20 lb. NEW charge. Navy divers observe the underwater detonation site prior to detonation, which should minimize the likelihood of detonating explosives while scalloped hammerhead sharks (or other ESA-listed species) are within the immediate vicinity, thereby minimizing injury or mortality. Since scalloped hammerhead sharks may occur at all three sites, and we do not have the breakdown of planned detonations at the three sites by NEW, we will assume a worst case scenario with all 260 planned detonations using 20 lb. NEW. We assume that mitigation measures including detection of scalloped hammerhead sharks (or any shark species) within the underwater detonation site exclusion zones prior to detonation and subsequent delay of detonation until the area is clear would effectively eliminate the potential for direct injury or mortality resulting from exposure of scalloped hammerhead sharks within 2.8 meters (9.2 feet) of a 20 lb. charge detonation.

While we do not expect detonations to result in any instances of shark mortality, sub-lethal effects (temporary hearing loss, behavioral response) could occur. Due to a lack of shark density information and hearing/behavioral threshold data, it is not feasible to estimate the number of scalloped hammerhead sharks exposed to sublethal effects from impulsive acoustic stressors associated with underwater detonations. Since estimating the number of individual exposed scalloped hammerhead sharks is not feasible, we use the amount of affected habitat as a surrogate to analyze the effects of impulsive acoustic stressors on the Indo-West Pacific DPS of scalloped hammerhead shark.

In the absence of information on range to significant behavioral effects to scalloped hammerhead sharks from underwater detonations and lack of data on the efficacy of Navy mitigation including the ability to detect scalloped hammerhead sharks during routine underwater detonation activities, we doubled the estimated mortality zones for 20 lb. NEW detonations to derive an area (displacement zone) where scalloped hammerhead sharks might experience behavioral responses, TTS, temporary disruption of feeding, breeding, or sheltering activities, or the reduced availability of preferred habitat. We determined that the affected habitat area for a 20 lb. NEW charge would be 99 m2 (1,064 ft^2) per detonation (based on a radius of impact of 5.6m). While we did not exclude the estimated mortality zone from this larger displacement area, we still assume the injury and mortality is not likely. We anticipate that up to 260 charges would be detonated annually at underwater detonation sites where scalloped hammerhead sharks may be present. In this scenario, any scalloped hammerhead sharks within the 99 m^2 area would be behaviorally-harassed, or up to 99 m^2 of their habitat would be temporarily displaced during each detonation. We expect the 260 underwater detonations and subsequent effects to be distributed across the Piti Point, Outer Apra Harbor, and Agat Bay sites throughout a given year, and we anticipate the number of charges that are routinely detonated in a day (no more than five) or week (no more than 20) would result in temporary effects that would not lead to a significant reduction in feeding, breeding, or sheltering opportunities. That is, we expect normal behavioral activities of scalloped hammerhead sharks and the availability of this habitat to sharks to resume soon after each of the 260 disturbances events have concluded.

Based on the ETOPO1 Global Relief Model Bathymetry data available from NOAA's National Geophysical Data Center there are approximately 19,462,434 km^2 of ocean with depths less than 1,000 m (where scalloped hammerhead sharks are most likely to occur) occurring within listed DPS segments. The Indo-West Pacific DPS contains 13,510,249 km^2 of ocean with depths less than 1,000 m (Figure 28).

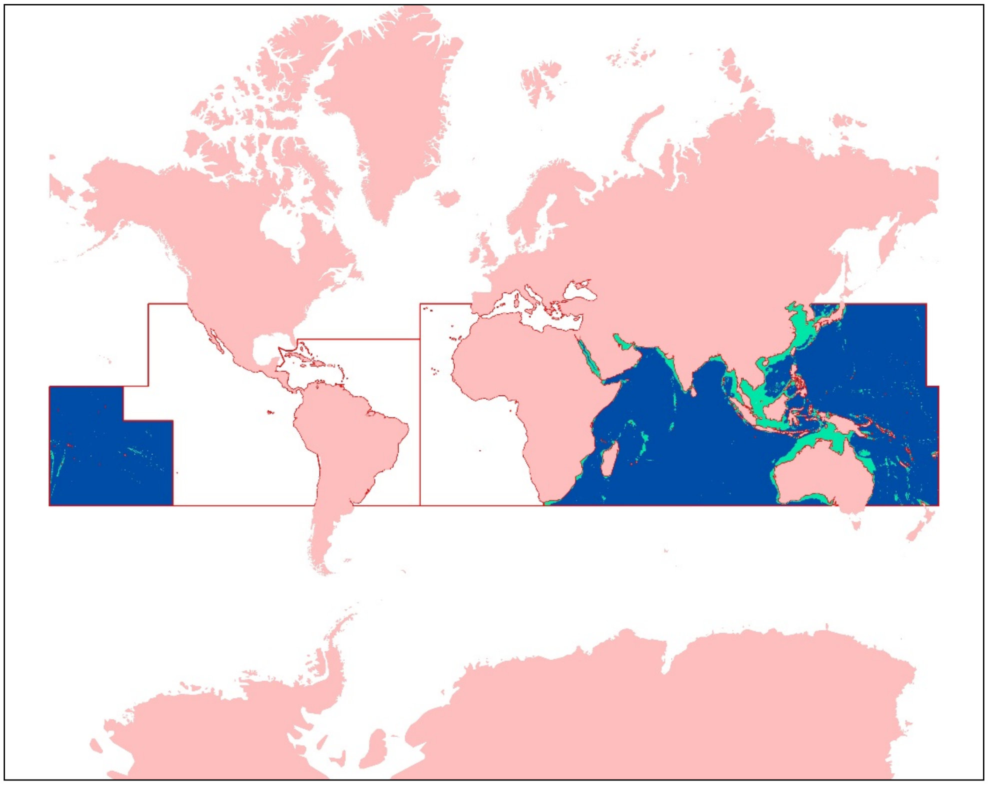


Figure 28. The Indo-West Pacifc DPS of scalloped hammerhead sharks. Dark blue areas are waters with depths greater than 1,000 m while light blue areas are

waters with depths less than 1,000 m. The total area of water with depths less than 1,000 m is 13,510,249 km^2.

6.8.9.2 Exposure to Impulsive Acoustic Stressors During Testing

No MITT testing activities include underwater detonations at the Apra Harbor, Agat Bay, or Piti Point sites. The use of impulsive airguns for pierside integrated swimmer defense testing activities occurring in Inner Apra Harbor will occur infrequently (no more than 11 activities totaling 308 hours per year) and will occur in a high-use area with high ambient noise levels. The small airguns used in pierside testing activities lack the strong shockwave and pressure increases associated with underwater detonations. These airguns produce 185 to 195 dB re 1 µPa^2 SEL at 1 m. The small ensonified area capable of eliciting an injury or behavioral response of scalloped hammerhead sharks and the limited duration of impulsive acoustic energy from swimmer defense airguns indicate the likelihood of exposure of scalloped hammerhead sharks to impulsive acoustic energy resulting in take from pierside integrated swimmer defense testing activities is so low as to be discountable. Therefore, no exposure of scalloped hammerhead sharks to impulsive acoustic stressors during testing is anticipated.

6.8.9.3 Response to Impulsive Acoustic Stressors During Training and Testing

Given the short distance from the detonation source that a shark would need to be for injury or mortality to occur and the mitigation procedures for underwater detonations, it is unlikely that scalloped hammerhead sharks would be injured or killed during underwater detonations. However, scalloped hammerhead sharks may experience hearing loss if within an unknown distance to the underwater detonation. Hearing loss would be temporary because unlike marine mammals, fish are capable of regenerating sensory hair cells and no permanent hearing loss has ever been reported in fish (Lombarte et al. 1993) (Smith et al. 2006b).

Additionally, the potential for behavioral harassment remains. As discussed above, we were unable to quantify the number of individual scalloped hammerhead sharks that could experience behavioral harassment. Instead, we estimated the area of affected habitat as a surrogate to define take. Because scalloped hammerheads are long-lived, highly mobile animals and alternate habitat is available outside of the habitat area expected to be effected during each underwater detonation event, sharks that are temporarily displaced from their preferred habitat would likely select adjacent habitat where forage, breeding habitat and refugia is available. Also, instances of temporary hearing loss are not expected to substantially impair individual scalloped hammerheads when considering overall lifetime fitness. We do not anticipate any responses of scalloped hammerhead sharks to integrated swimmer defense testing activities that would rise to the level of take. Because we do not expect fitness consequences to any individuals resulting from temporary displacement of habitat or TTS, we do not expect population-level consequences.

6.8.10 Effects of Impulsive Acoustic Stressors and Military Expended Materials

The effects of impulsive acoustic stressors on marine mammals, sea turtles, and scalloped hammerhead sharks are discussed in sections 6.8.7, 6.8.8, and 6.8.9, respectively. The effects of military expended materials on these species are discussed below in Section 6.8.10.1. In Section 6.8.10.2, we combined the discussion of the effects of impulsive acoustic stressors and military expended materials on *Acropora globiceps* because both stressors are related to the use of ordnance.

6.8.10.1 Marine mammals, sea turtles, and scalloped hammerhead sharks

Military expended materials include all sizes of non-explosive practice munitions, fragments from explosive munitions, and expended materials such as sonobuoys, ship hulks, and targets. The expected density of ESA-listed marine mammals, sea turtles, and scalloped hammerhead sharks in areas receiving a majority of munition, and ordnance testing that could potentially strike these species is very low. The impact site for munitions, dropped ordnance, and other expended materials have very small footprints. The density of marine mammals, sea turtles, and scalloped hammerhead sharks; the small footprint of expended materials; and the Navy's strike modeling efforts suggest direct strikes to these species from expended materials is so unlikely as to be discountable. According to the Navy's biological evaluation, there are no records of military expended materials striking marine mammals, sea turtles, or scalloped hammerhead sharks further supporting this conclusion.

6.8.10.2 Acropora globiceps

This section discusses the effects of impulsive acoustic stressors (i.e., explosives) and military expended materials on colonies of *Acropora globicep*s. As described previously in section 4.1 of this opinion, the available information regarding the distribution of *Acropora retusa* and *Seriatopora aculeata*, does not suggest these species are likely to occur near underwater detonation sites or at-sea target sites. Therefore, we only analyze effects to *Acropora globiceps* in this section.

Coral reef survival and recovery may be affected by acute single blasts as well as chronic blasting over greater spatial and temporal scales. Fox and Caldwell (2006) examined coral reef recovery following acute single blasts and following chronic blasting. Rubble resulting from single blasts slowly stabilized, and craters filled in with surrounding coral and new colonies. After five years, coral cover within craters formed by single blasts no longer differed significantly from control plots. In contrast, extensively bombed areas showed no significant recovery over the six years of the study, despite adequate supply of coral larvae. After extensive blasting, resulting coral rubble may shift in ocean currents, abrading or burying new coral larvae, thereby slowing reef recovery (Fox and Caldwell 2006). The effects of dynamite or "blast" fishing may help provide some insight to the potential effects of detonation of live military ordnance in and around coral colonies. The shock waves from blast fishing explosions break the coral's calcium carbonate skeleton into small pieces. Once broken, the coral/algal symbiotic relationship is disrupted and the coral begins to lose nourishment and starts to die. Blast fishers typically target clumps of corals which often suffer mortality within approximately 1-2 meter radius from the blast (McManus 1997).

A study at a former U.S. Navy range at Vieques Island (Puerto Rico, United States), which is now the largest national wildlife refuge in the Caribbean, investigated the geomorphology and benthic assemblage structure to understand the status of the coral reefs (Reigl et al. 2008). In that study, investigators found no differences in living benthic coral reef cover or composition of coral assemblages inside and outside the bombing range or in comparison to reefs investigated on St. Croix. Reigl et al. (2008) concluded that this may indicate not that zero impacts occurred but rather that natural disturbances appear to have altered the coral communities drastically, thus obscuring military impacts. Effects of natural disturbances were severe at Vieques, outweighing impacts of past military activity which were present but not quantitatively discernible at the scale of sampling. Disease and storms, rather than military expended ordnance, were seen to have taken the worst toll on corals at both Vieques and St. Croix (Reigl et al. 2008).

The US Navy conducted annual nearshore marine resource surveys around FDM from 1999 through 2012 (except 2011). A 2013 report presented the findings of the calendar year 2012 survey and compares findings from those of the previous 12 surveys (Smith et al. 2013a). The report indicates that despite ongoing use by the Department of Defense as a live and inert range, no significant impacts to the physical or biological environment were detected between 1999 and 2012. Direct ordnance impacts upon the submerged physical environment, which were clearly attributable to training activities, were detected in 2007, 2008, 2010, and 2012. Indirect impacts, such as ordnance that skipped or eroded off the island and rock and ordnance fragments blasted off the island were detected every year. The report indicated that very few areas of disturbance were detected, the size of any disturbed areas were generally less than 2 m^2, and substantial or complete recovery of these disturbed areas occurred within one year. Additionally, large numbers of one and two year old stony coral recruits were consistently observed, suggesting that coral recruitment is not a limiting factor around the island. The report also indicated that restricted access to FDM (because it is a DoD live and inert range) may have a conservation benefit to the reef ecosystem around the island, with marine resources at FDM comparable to or superior to those of any of the other islands within the Mariana Archipelago (Smith et al. 2013a). A publication using these data (years 2005 to 2012) supported this conclusion, finding that restricted access around FDM has “resulted in a de-facto preserve effect” (Smith and Marx Jr. 2016).

Potential impacts to ESA-listed corals from impulsive acoustic stressors around Guam

The vast majority of training and testing activities around Guam that use explosives occur in areas greater than 12 nm from shore. The potential impacts of these activities to ESA-listed corals are not considered further in this opinion because ESA-listed corals do not occur in water depths that occur this far from shore. Similarly, the Agat Bay Mine Neutralization Site is located beyond 3 nm of Guam and we do not expect ESA-listed coral to occur at this location because of the water depths this far from shore. The Outer Apra Harbor UNDET site and Piti Point Mine Neutralization site are within less than one kilometer of existing reef structures known to support coral growth (Figure 29). Explosives used at these sites are limited to 10 lbs NEW. Several surveys have been conducted within Apra Harbor (e.g., (Smith et al. 2009); (Starmer 2008)) and in only three instances (Lybolt 2015; Schils et al. 2011), was *Acropora globiceps* observed. However, the species was observed near Kilo Wharf and Spanish Steps, located across Apra Harbor from the UNDET site. The species has not been documented within a close enough range to the Outer Apra Harbor UNDET site to be adversely affected by Navy activities at this site. It is possible that *Acropora globiceps* is present at the Piti Point UNDET site, most likely very close to shore. However, no more than a 10 lb NEW detonation will be used at this site, resulting in a percussive effect of about a 10 foot radius. All UNDET locations are monitored for presence of corals and the Navy maintains a distance of 350 feet from hard bottom structures (e.g., coral reefs) during detonations, indicating that coral reefs (including those that may contain *Acropora globiceps*) are not expected to be affected. Effects from explosions and military expended materials at these locations are not reasonably expected to affect ESA-listed coral colonies due to the limited size of the explosives used, the distance between the sites and known locations of *Acropora globiceps,* as well as precautions taken by the Navy during these activities. For these reasons, the likelihood of adult *Acropora globiceps* colonies being exposed to impulsive acoustic stressors around Guam is discountable.

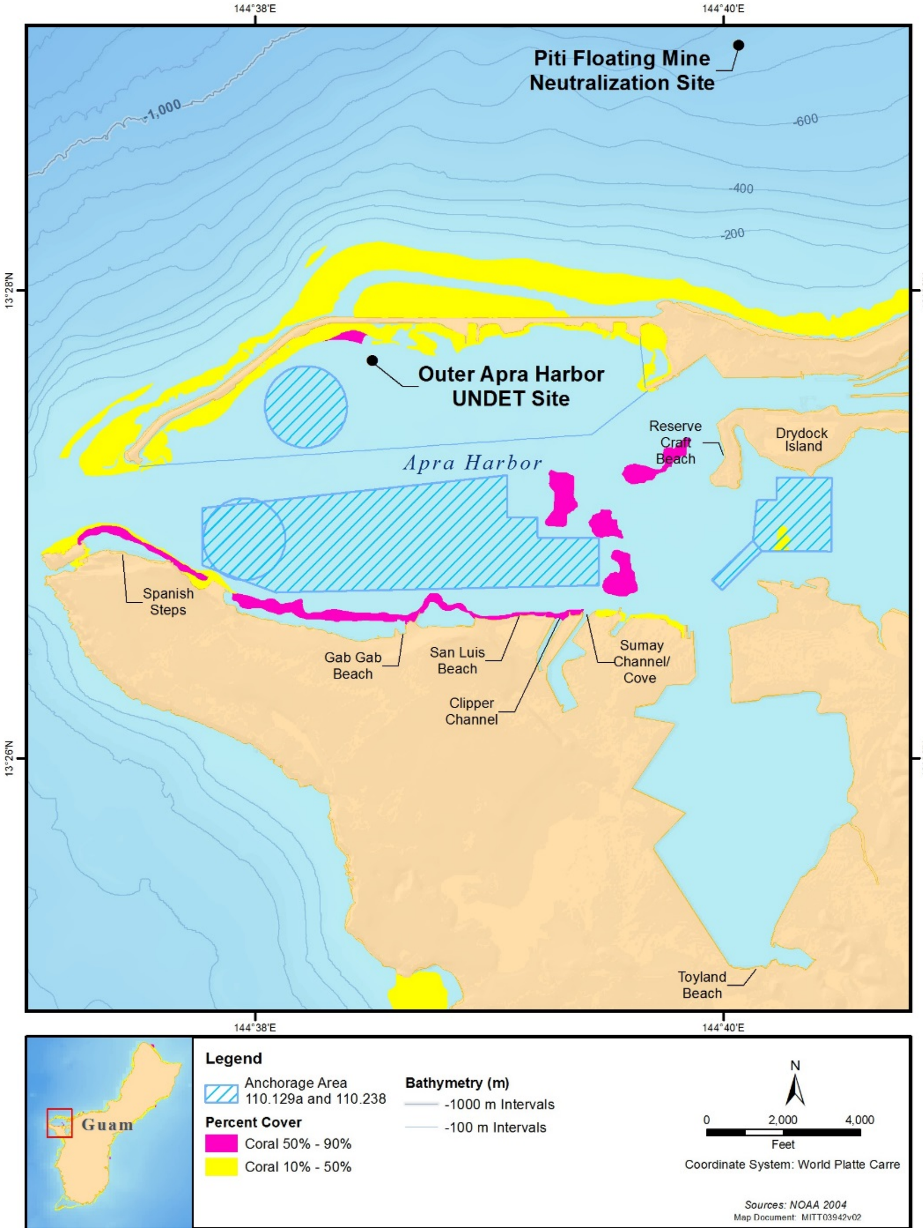
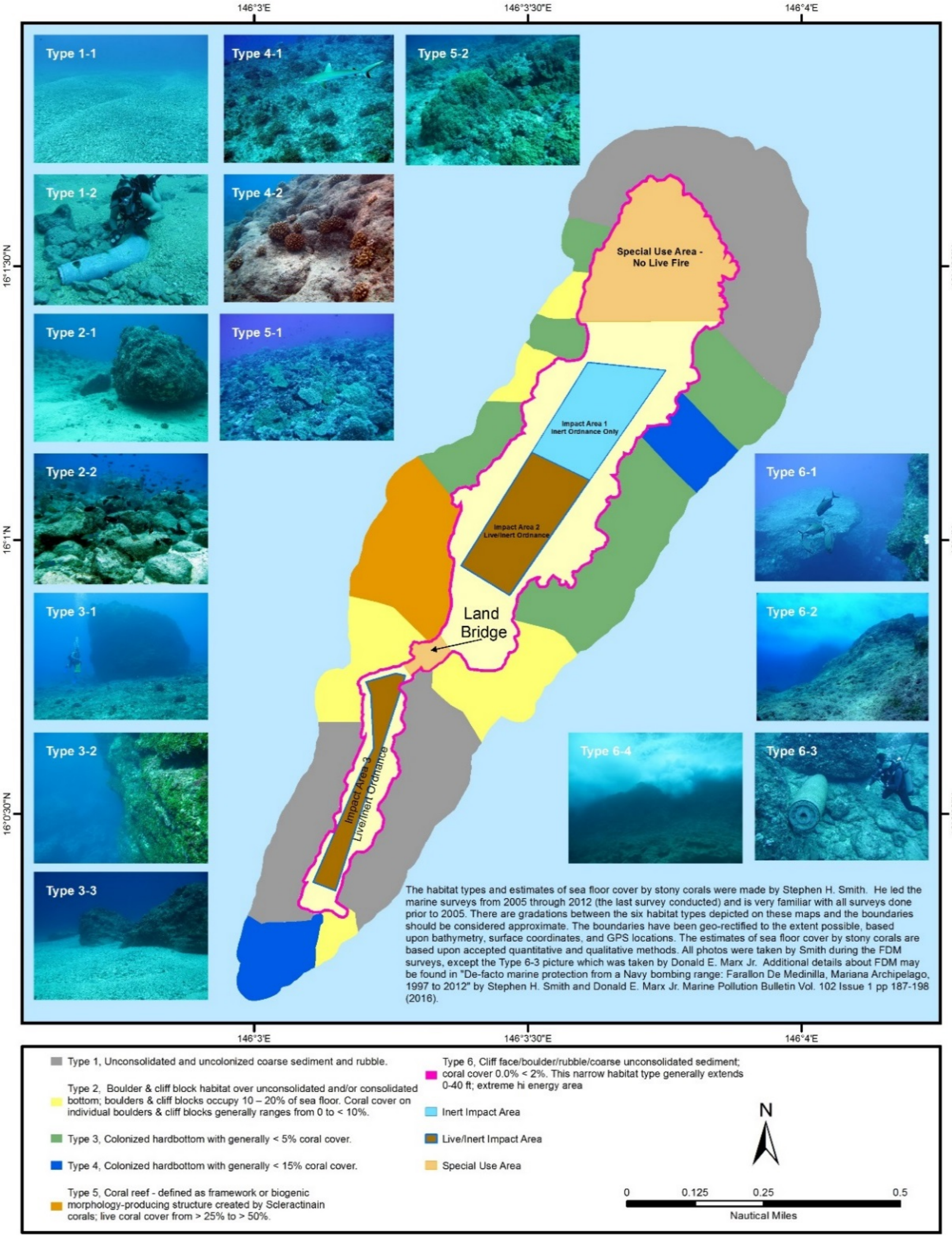


Figure 29. Underwater Detonation and Mine Neutralizaton Sites in and around Apra Harbor, Guam

Potential impacts to ESA-listed corals from impulsive acoustic stressors and expended materials around Farallon de Medinilla

FDM has been a target site for live-fire military exercises (ship-to-shore gunfire, aerial gunnery and bombing) since 1971 (Smith et al. 2013a). While the majority of live and inert ordnance strikes the island and does not impact the nearshore marine environment, there are known instances where bombs have missed the island, or where munition fragments have entered the nearshore environment following an impact to the island (e.g., Smith and Marx Jr. 2016). Therefore, ESA-listed corals that are present in the nearshore environment around FDM could be impacted by Navy training at FDM. If an inert munition lands in the nearshore environment, either as a result of missing the island or ricocheting off the island, and contacts any ESA-listed coral colonies, it could cause injury or mortality of those colonies. Similarly, if an on-island explosion ejects munition and rock fragments into the surrounding waters, and these fragments contact an ESA-listed coral, it could also cause injury. Should munitions land in the nearshore environment and explode, the explosion would occur at or near the water's surface. While most explosive energy would be reflected upward, any coral in the vicinity would be exposed to pressure waves, which could cause injury or mortality.

Data from underwater surveys around FDM indicate that *Acropora globiceps* is the only ESA-listed coral that is likely to inhabit waters around FDM (DoN 2005a; Smith et al. 2013b). While the species was field identified during most of the FDM surveys from 1999 to 2012, survey data from FDM indicate that *Acropora globiceps* is relatively rare compared to many of the other 80 species identified in the area (DoN 2005b). The most abundant coral species at FDM are *Pocillopora meandirna* and *Pocillopora eydouxi* (Smith et al. 2013b). Dive personnel observed most *Acropora globiceps* specimens at depths between 15 and 25 meters (Stephen H. Smith, personnel communication to NMFS; March 3, 2017). Without specific information on the exact locations of *Acropora globiceps* colonies, we assume that the species could occur in any of the areas with live coral cover within this depth range. As shown in Figure 30, live coral cover varies around the island, with some areas having zero percent cover and some areas having greater than 50 percent live coral cover.



The habitat types and estimates of sea floor cover by stony corals were made by Stephen H. Smith. He led the marine surveys from 2005 through 2012 (the last survey conducted) and is very familiar with all surveys done prior to 2005. There are gradations between the six habitat types depicted on these maps and the boundaries should be considered approximate. The boundaries have been geo-rectified to the extent possible, based upon bathymetry, surface coordinates, and GPS locations. The estimates of sea floor cover by stony corals are based upon accepted quantitative and qualitative methods. All photos were taken by Smith during the FDM surveys, except the Type 6-3 picture which was taken by Donald E. Marx Jr. Additional details about FDM may be found in "De-facto marine protection from a Navy bombing range: Farallon De Medinilla, Mariana Archipelago, 1997 to 2012" by Stephen H. Smith and Donald E. Marx Jr. Marine Pollution Bulletin Vol. 102 Issue 1 pp 187-198 (2016).

Figure 30. Habitat types around Farallon de Medinilla with representative photos. Impact areas are also presented.

As detailed in Table 60, FDM supports a full suite of munitions use, from the delivery of heavyweight explosive bombs dropped from fighter aircraft to small arms fire from helicopters. Up to 6,242 explosive bombs and 2,670 non-explosive bombs would be expended annually at

FDM (8,912 total). Bombs range in size from 25 pounds to 2,000 pounds. Up to 42,000 small-caliber projectiles would be expended annually at FDM. Small-caliber projectiles are those projectiles that are 50 caliber and below. Small caliber projectiles will lose most of their energy upon contact with the water surface and are not expected to impact *Acropora globiceps* (which have typically been observed at FDM in waters between 15 and 25 meters deep) with enough force to cause a measurable effect. Up to 17,350 explosive and 94,150 non-explosive projectiles would be expended annually. Medium caliber projectiles are those projectiles that are greater than 50 caliber, but less than 57mm. Up to 1,200 explosive, large-caliber projectiles and up to 1,800 non-explosive large-caliber projectiles would be expended annually. Large caliber includes 5-inch ship fired projectiles as well as mortars fired into the impact areas from the northern end of FDM. Up to 85 explosive missiles would be expended and up to 2,000 explosive rockets would be expended annually at FDM.

To estimate the potential impact resulting from physical disturbance, strike, and explosions, the Navy estimated the numerical quantity of explosive and non-explosive munition items, and munitions fragments which might enter the nearshore environment. While the maximum quantity of munitions to be used on FDM is known, they are intended to target FDM, not the surrounding waters. Thus, the number of munitions or fragments that enter the surrounding water, directly or indirectly, was estimated based on the percentage of those munition items which hit the island as intended and then enter the water due to ricochet or some other process, and the number of munitions that outright miss the island and land directly in the surrounding water. The Navy provided these estimates based on several data sources including range munition tracking information, after action reports, and Navy underwater dive studies of the waters surrounding FDM. For example, the Navy determined that at most, the number of non-explosive bombs which enter the near-shore environment is two percent of the non-explosive bombs expended and determined that the number of explosive bombs which enter the nearshore environment directly is at most one percent of the explosive bombs expended. More non-explosive bombs would be expected to enter the nearshore environment than explosive bombs because non-explosive bombs are much more likely to ricochet off the island and explosive bombs would only enter the nearshore environment as a result of missing FDM (Navy 2015). Table 60 provides the Navy's estimates for the percent of each type of munition that are expected to impact nearshore habitat around FDM. Further detail on how these estimates were derived is available in a memo to the file (Navy 2015).

For non-explosive munitions, munition fragments, and explosives with relatively small explosive weight (e.g., rockets and medium/large caliber projectiles) that may enter the nearshore environment, the Navy estimated an impact footprint based on the size of the munition or munition fragment. This information is provided in Table 60. Note that the rockets and medium/large caliber projectiles used at FDM have relatively small explosive weights and unlike bombs, the explosive effect of these munitions when detonated at the water's surface is not expected to impact coral (especially since ESA-listed corals at FDM were observed in 15-25 m depths).

For bombs with high explosives (HE) that may have greater impacts (see footnote in table), we also assessed the potential impact zone from a documented Navy observation of an underwater impact crater. During a 2010 survey, a fresh shallow crater was observed for the first time since 1999. The crater pit was 5 m across in its maximum dimension and the cratered portion of the seafloor was a maximum of 50 cm deeper than the surrounding sea floor. Water depth at the site was 12 m. Explosive Ordnance Disposal (EOD) Detachment Marianas personnel judged that a bomb had detonated at the water surface to produce the disturbance. This event occurred in an area that was dominated by relatively barren bedrock. No corals or any other sessile benthic invertebrates or the remains thereof were observed in the crater/blast pit or within a distance of approximately 4 m from the edge of the crater. Past the 4 m perimeter, (approximately 9 m from the center of the impact site) sea floor cover by corals was estimated to be less than 5 percent. This observation suggests that any coral within a circular area with a diameter of 13 meters (5m + 4m + 4m) could have been impacted, as coral beyond this range appear to be similar to undisturbed background locations. From this observation, we established an impact zone extending 6.5 meters from the center of the crater, equating to an area of 132.73 m^2.

The impact zone is based on a single observation of a surface detonation in 12 m of water with approximately 5 percent coral cover. We recognize that the range to effects for mortality and injury will likely increase with a decrease in water depth (less than 12 meters) and will likely decrease as water depth increases (greater than 12 meters). We also recognize the explosive weight of the bomb that created the crater was not known. However, we assume it was a relatively large bomb to create a crater of that size 12 meters below the water’s surface. As such, smaller bombs would be expected to create a smaller blast-impact area. Also, the magnitude of effects may vary depending on bottom type and bottom features (boulders, shelves, etc.). For example, the percent of hard bottom versus soft bottom environments might influence the amount of refraction of sound pressure waves. Nevertheless, we consider this zone of effect to be representative of a typical scenario at FDM with an average depth and bottom type similar to the 12 meter depth observation.

Table 60 provides estimates of the total nearshore habitat area affected by munitions at FDM. The total area impacted by each munition type is the product of the number of items that are expected to fall in the nearshore environment by the impact area per item. As described above, with the exception of high explosive bombs, the area of habitat affected for all munition types is based on the physical footprint of that particular munition (i.e., from direct strike). For explosive bombs, the area of habitat affected is based the impact area from an explosion.

Table 60. Estimated Area of Nearshore Habitat Impacted by Ordnance at Farrallon de Medinilla

Ordnance Items per the Proposed Action (FEIS/OEIS)	Number of Items Expended Annually	Percent of Items That Fall in the Nearshore Environment	Number of Items That Fall in Nearshore Environment	Impact Area Per Item (m2)	Total Nearshore Habitat Area (m2)
Bombs (HE)[1]	6,242	1	62.42	132.73	**8,285.01**
Bombs (NEPM)	2,670	2	53.4	3.0044	160.44
Bomb debris (end plates)	6,242	50	3,121	0.072929	227.61
Bomb Debris (ejecta)	6,242	50	3,121	0.072929	227.61
Small-caliber projectiles	42,000	5	2,100	0.0056	11.76
Medium-caliber projectiles (HE)[2]	17,350	5	867.5	0.0104	9.02
Medium-caliber projectiles (NEPM)	94,150	5	4,707.5	0.0104	48.96
Large-caliber projectiles (HE)[2]	1,200	5	60	0.1876	11.26
Large-caliber projectiles (NEPM)	1,800	20	360	0.1876	67.54
Missiles[3]	85	0	0	--	0
Rockets[2]	2,000	5	100	0.1484	14.84
				Total Habitat Area Affected	9,064.04

[1] Bombs (HE) is the only ordnance anticipated to result in mortality of coral colonies

[2] The rockets and medium/large caliber projectiles used at FDM have relatively small explosive weights and unlike bombs, the explosive effect of these munitions when detonated at the water's surface is not expected to impact coral (especially since ESA-listed corals at FDM were observed in 15-25 m depths).

[3] Missiles are precision-guided and therefore will not fall in the nearshore environment.

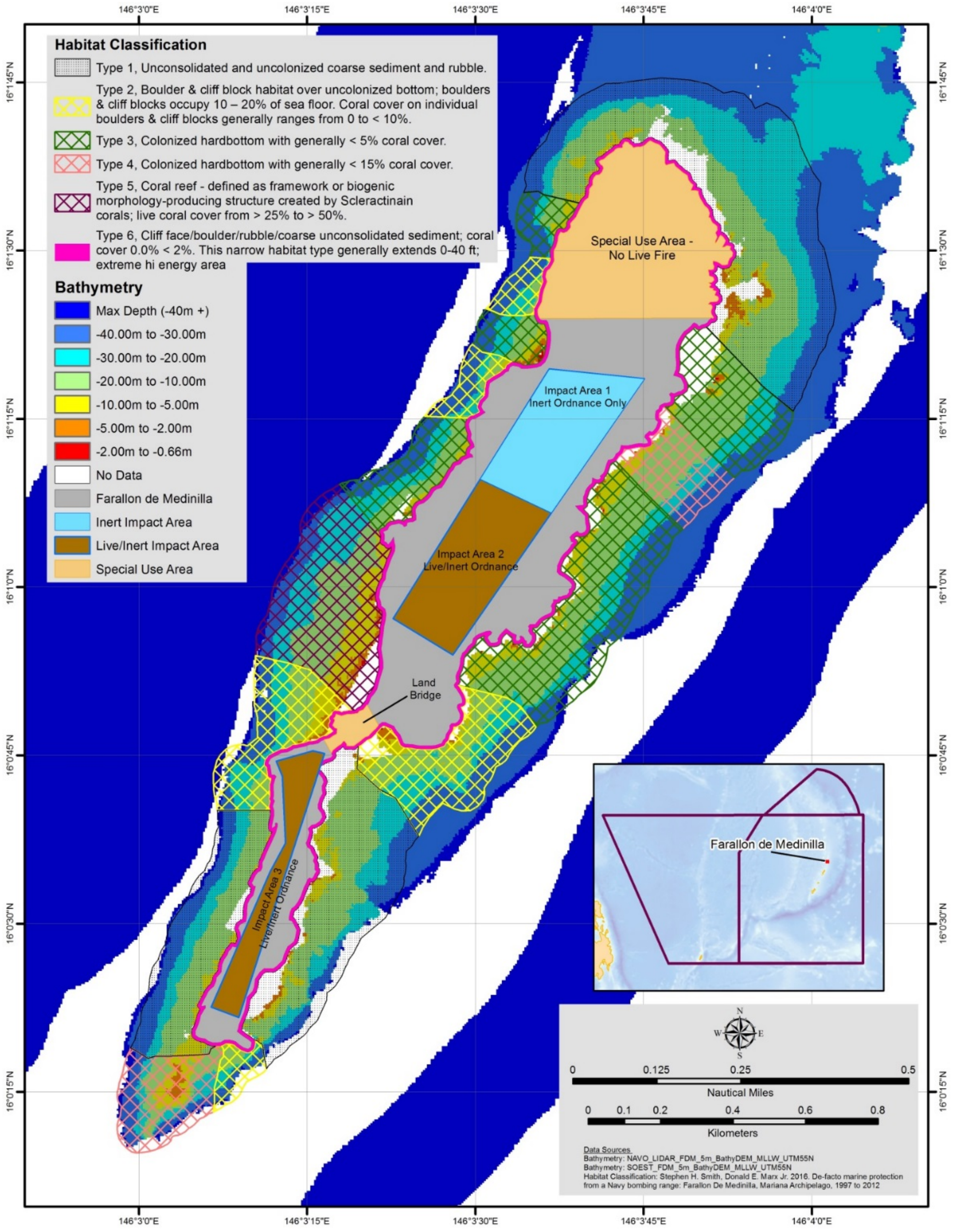
As detailed above and shown in Table 60, we estimate that the total area of nearshore habitat around FDM impacted annually is 9,064 square meters. The large majority of impacts are estimated to result from high explosive bombs that miss their intended on-shore target. We consider the estimated area of impact calculated above to be highly conservative because the crater our estimates are based on is the largest that has been observed in a decade of dive surveys. For example, Smith et al. (2013b) indicated that the size of any disturbed areas were generally less than two square meters.

We are unable to provide an estimate of the number of colonies of *Acropora globiceps* that may be within the area of nearshore habitat that we estimated will be affected annually. As noted in the final listing rule, Indo-Pacific reef-building coral species are generally difficult to identify, even by experts, because of: (1) The high biodiversity of reef-building corals; (2) the high morphological plasticity in many reef-building coral species; and (3) the different methods used for species identification (NMFS 2014, 79 FR 53852). For example, 13 of the 15 ESA-listed Indo-Pacific coral species, including *Acropora globiceps*, have a moderate or high level of species identification difficulty (Fenner 2014). Thus, even if experts can be hired to survey and monitor action area sites, the species-level data is likely to be confounded by species identification uncertainty.

In addition, coral reef communities are highly dynamic whether humans are present or not, with species presence/absence, colony density, colony size and morphology, and other factors varying over small spatial scales (e.g., a few meters separate forereef and backreef habitats, which can have radically different coral communities) and small temporal scales (e.g., seasonal and annual cycles of natural disturbance, like storms and predation events, can wipe out a species or community in a particular area, followed by species recovery). The spatial and temporal variability in coral habitat and species abundance is described in detail in the final rule's Corals and Coral Reefs section (NMFS 2014, 79 FR 53852). Thus, even if reliable species-level data can be produced for a given action area site, over time, any changes in species abundance caused by the proposed action are likely to be confounded by natural variability.

Thus, for the Navy’s action, it is not practical or possible to express the amount or extent of anticipated take of *Acropora globiceps*, or to monitor take-related impacts in terms of individuals of these species. Therefore, the incidental take of *Acropora globiceps* is expressed as a habitat area surrogate as prescribed by 50 CFR 402.14(i). In this case, we estimated that the total area of nearshore habitat around FDM impacted annually is 9,064 square meters (i.e., from direct strike and explosive effects). Though we are unable to provide a quantitative estimate of the number of *Acropora globiceps* colonies impacted by Navy activities at FDM, we can qualitatively assess the likelihood that impacts will occur to areas of nearshore habitat likely to contain colonies of the species. Areas of habitat with higher percent live coral cover between 15 and 25 meters (i.e., the depth range where the species was observed during coral reef surveys) would be most likely to contain *Acropora globiceps* colonies.

Figure 31 below shows the potential impact areas on FDM for live and inert ordnance, along with habitat type, percent live coral cover in each nearshore zone, and bathymetry. Impact areas one, two, and three are targets of inert ordnance whereas the use of live ordnance is limited to impact areas two and three. The majority of the in-water habitat adjacent to impact area three (i.e., on the east or west side of the island) is composed of unconsolidated and uncolonized course sediment and rubble with zero percent live coral cover. If ordnance were to impact nearshore habitat in these zones, we would not expect *Acropora globiceps* colonies to be impacted. These two zones are also the locations where we expect the majority of the nearshore impacts to occur because impact area three is a more narrow land mass than impact areas one and two, leaving pilots less margin for error. Much smaller zones primarily south of impact area three have live coral cover as high as 15 percent or live coral cover ranging between 0 and 10 percent. Adjacent to impact area two, most of the live coral cover is less than 5 percent, but a portion is composed of coral reef with live coral cover ranging from greater than 25 percent to greater than 50 percent. If ordnance were to land in the zone with the highest live coral cover, the likelihood of impacting a colony of *Acropora globiceps* would also be higher.



Data Sources
Bathymetry: NAVO_LIDAR_FDM_5m_BathyDEM_MLLW_UTM55N
Bathymetry: SOEST_FDM_5m_BathyDEM_MLLW_UTM55N
Habitat Classification: Stephen H. Smith, Donald E. Marx Jr. 2016. De-facto marine protection from a Navy bombing range: Farallon De Medinilla, Mariana Archipelago, 1997 to 2012

Figure 31. Coral habitat map around Farallon de Medinilla overlaid with bathymetry.

Similar to other areas in the Indo-Pacific, the coral community around FDM is characterized by high species diversity (79 FR 53852), with a low proportion of ESA-listed species (i.e., *Acropora globiceps*) mingled with many non-listed corals (DoN 2005b). *Acropora globiceps* is considered one of the more common ESA-listed corals in the Indo-Pacific, but Veron (2014) reported that the species only occurred at 3.2 percent of the 2,984 dive sites sampled. At each site where it was sampled, based on an abundance rating on a scale of 1 (low) to 5 (high) at each site where it occurred, this species had a mean abundance rating of 1.95. Based on this semi-quantitative system, the species' abundance was characterized as "uncommon" (Veron 2014). Similarly, while the species was field identified during most of the FDM surveys from 1999 to 2012, survey data from FDM indicate that *Acropora globiceps* is relatively rare compared to many of the other 80 species identified in the area (DoN 2005b). This, and the information presented above regarding the locations where the majority of nearshore habitat impacts are expected to occur (i.e., in areas with very low percent live coral cover), suggests that while a small area of habitat affected by Navy activities each year may contain a limited number of *Acropora globiceps* colonies, the majority of the area impacted will not.

Response and Risk Assessment

Any *Acropora globiceps* colonies that occur in the area of habitat impacted will be subject to a range of impacts. Even slight physical contact with a coral colony by an ordnance or ordnance fragment can crush and/or scrape off living polyps and interconnecting soft tissues in the area of contact, causing injured or dead tissue in the disturbed area. Additionally, direct trauma and mortality of corals may occur due to the rapid pressure changes associated with an explosion. Though most invertebrates lack air cavities that would respond to pressure waves, which typically causes the most damage in fish or marine mammals, a blast in the vicinity of hard corals (i.e., *Acropora globiceps*) could cause direct impact to coral polyps leading to coral colony death, or fragmentation and siltation of the corals.

The tissue thickness of *Acropora* species is 1 to 2 mm thick, considerably thinner than many coral species, which allows them to grow quicker than many other species (Loya et al. 2001). Therefore, injured *Acropora globiceps* colonies (i.e., colonies that are not completely destroyed) would likely be able to bud and develop new polyps to replace those lost in the injury. Fragmentation of the skeleton could result in the development of new, but genetically identical colonies. Bothwell (1981) reports that several *Acropora* species successfully colonize through fragmentation and translocation of fragments by storm-driven waves. Broken pieces may develop into new colonies over time, but re-growth of damaged tissue and skeleton has energetic costs that could slow other physiological processes such as reproduction. Fragmentation may lead to a large number of asexually-produced, genetically identical colonies, commonly resulting in a population made up of more asexually-produced colonies than sexually-produced colonies (Hughes 1984).

As described above, while a small area of habitat affected by Navy activities each year is likely to contain colonies of *Acropora globiceps* that could be injured or killed, the majority of the area impacted will not. Further, although individual colonies and clusters of colonies forming a small-scale reefscape are likely to be negatively impacted by impulsive explosions, underwater surveys of FDM reefs suggest significant population level impacts are not likely to occur, and colony repair or successful recruitment will likely occur within two to three years following disturbance

(Smith and Marx Jr. 2016). Smith and Marx Jr. (2016) documented that while impacts to reef habitat have occurred around FDM (i.e., from ordnance that skipped off the island, from ordnance fragments, and from an in-water detonation), no significant impacts to the physical or biological environment were detected between 2005 and 2012. Instead, the authors suggested that restricted access to FDM because it is a bombing range has resulted in a de-facto preserve effect. They noted that marine natural resources at FDM are "comparable or superior to" those at other locations within the Mariana Archipelago. This is despite FDM being used as a target site for live-fire military exercises (ship-to-shore gunfire, aerial gunnery and bombing) since 1971 (Smith et al. 2013a).

Additionally, the area of nearshore habitat that is expected to be affected by explosives and military expended material at FDM is infinitesimally small in relation to available habitat within this species' range. *Acropora globiceps*, and other ESA-listed corals in the Indo-Pacific, consist of at least millions of colonies, and occur across a range of thousands of miles. Because the species is sparsely populated across a wide range, localized impacts to potential coral reef habitat for this species are not expected to impact the species' ability to reproduce. Instead, other factors that affect corals over a broad geographic scale are larger drivers of the ability of *A. globiceps* to survive and recover. These factors include ocean warming, disease, ocean acidification, trophic effects of fishing, nutrients, and predation. Therefore, we do not believe annual impacts to this area of habitat at FDM will result in an appreciable reduction in the likelihood of the survival or recovery of *Acropora globiceps* in the wild.

Potential impacts from detonations on ESA-listed coral eggs, sperm, and larvae

Acropora globiceps broadcast spawn where fertilization and early embryonic development occurs (See section 4.2.11.3). The eggs, sperm, and larval stage of *Acropora globiceps* could remain in the water column for extended periods. Each individual polyp of an *Acropora* coral can produce 16 eggs and concentrations of sperm can be as high as 1 million per milliliter of seawater during spawning. Fertilized eggs develop into planula larvae within 5 days in *Acropora* species but these larvae can also remain in the water column over 200 days before settling. It is reasonable to assume in-water detonations occurring around FDM, at the Piti Point Floating Mine Neutralization site, the Agat Bay Mine Neutralization site, and the Outer Apra Harbor UNDET site could affect eggs, sperm, or planula larvae of *Acropora globiceps* (or other ESA-listed coral species) if their presence coincided with an explosion. Life stages subjected to the shearing forces of turbulent shockwaves from underwater detonations could be deformed, die, or experience a decreased likelihood of fertilization. Shock waves in the waters around explosions may reflect off of hard surfaces and the surface of the water, magnifying the exposure of nearby reefs. However, as described above, the reproductive biology of *Acropora globiceps*, and other coral species, results in prolific larval production and high natural mortality from a combination of factors including predation and dispersal to areas within the ocean without appropriate settlement habitat (e.g., deeper or colder water, unsuitable substrate). Any anthropogenic mortality from the Navy's actions is likely to be infinitesimally small by comparison (L. Smith, personal communication, May 6, 2015) and biologically insignificant to reproduction of corals. Additionally, of the 19 threats to coral identified in the 2011 status review report of the 82 candidate coral species petitioned under the U.S. Endangered Species Act (Brainard et al. 2011a) and the top 9 threats to coral analyzed in the final rule (79 FR 53851), none include mortality of larvae by physical contact such as cavitation or explosives, or acoustic effects. While detonations may result in the mortality of the developmental stages of *Acropora globiceps* (and other ESA-listed coral species), it likely would have an insignificant effect on the reproductive potential for an individual colony of the species or recruitment at the population level of this species. Since this level of effect is not expected to be significant and detectable at the individual level (i.e., colony) we would not consider this effect to be a reduction in fitness of any colony of *Acropora globiceps* and thus we do not anticipate any population-level effects.

6.9 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. Most of the action area includes federal military reserves or is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of business journals, trade journals, and newspapers using *First Search*, Google, and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require federal authorization or funding and is reasonably certain to occur. As a result, NMFS is not aware of any actions of this kind that are likely to occur in the action area during the reasonably foreseeable future.

6.10 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 6) to the environmental baseline (Section 5) and the cumulative effects (Section 6.9) to formulate the agency's biological opinion as to whether the action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 4).

The following discussions separately summarize the probable risks the proposed action poses to threatened and endangered species and critical habitat that are likely to be exposed. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this opinion.

Our effects analyses identified the probable risks the DoD and USCG training and testing activities and issuance of an MMPA rule and LOA to authorize take of marine mammals would pose to ESA-listed individuals that will be exposed to these actions. We measure risks to individuals of endangered or threatened species using changes in the individual's "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 DoD and USCG conducts in the MITT action area will continue to introduce a suite of stressors into the marine and coastal ecosystems of the Mariana Islands and the transit lanes to and from locations around Hawaii where Navy training and testing activities occur. The stressors include: low, mid, and high-frequency active sonar from surface vessels, torpedoes, and dipping sonar; shock waves and sound fields associated with underwater detonations, acoustic and visual cues from surface vessels as they move through the ocean's surface, and sounds transferred into the water column from fixed-wing aircraft and helicopters. Repeatedly exposing endangered and threatened marine animals to each of these individual stressors could pose additional risks as the exposures accumulate over time. Repeated exposures are discussed in more detail in section 6.2.11 and 6.3.7. 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 MITT action area would be exposed to combinations of stressors produced by DoD and USCG activities at the same time they are exposed to stressors from other human activities and natural phenomena. We recognize these interactions might have effects on endangered and threatened species that we have not considered; however, the data available do not allow us to do more than acknowledge the possibility.

For the purposes of this opinion, we assume that all of these activities in the MITT action area and associated impacts will continue into the reasonably foreseeable future at the levels set forth in the Final EIS/OEIS and MMPA rule. To address the likelihood of long-term additive or accumulative effects, we first considered (1) stressors that accumulate in the environment, and

(2) effects that represent either the response of individuals, populations, or species to that accumulation of stressors.

Sound does not permanently accumulate in the environment; therefore, an accumulative effects analysis on this stressor is not warranted. However, repeated exposure of individuals to acoustic stress can cause auditory fatigue and hearing loss. We expect ESA-listed species will not receive repeated exposures at a rate at which recovery between exposures would not occur because of the intermittent nature and duration of Navy acoustic sources. Navy activities in the MITT action area involving active sonar or underwater detonations are infrequent, short-term, and generally unit level. Unit level events occur over a small spatial scale (one to a few 10s of square miles) and with few participants (usually one or two). Single-unit unit level training would typically involve a few hours of sonar use, with a typical ping of every 50 seconds (duty cycle). Even though an animal's exposure to active sonar may be more than one time, the intermittent nature of the sonar signal, its low duty cycle, and the fact that both the vessel and animal are moving provide a very small chance that exposure to active sonar for individual animals and stocks would be repeated over extended periods of time. Consequently, the DoD and USCG's MITT activities do not create conditions of chronic, continuous underwater noise and are unlikely to lead to habitat abandonment or long-term hormonal or physiological stress responses in marine mammals.

Although Goldbogen et al. (2013) speculates that "frequent exposures to mid-frequency anthropogenic sounds may pose significant risk to the recovery rates of endangered blue whale populations," the authors acknowlege that the actual responses of individual blue whales to simulated mid-frequency sonar documented in the study "typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Moreover, the most significant response documented in the study occurred not as a result of exposure to simulated mid-frequency sonar but as a result of exposure to pseudo-random noise. Therefore, the overall weight of scientific evidence indicates that substantive behavioral responses by mysticetes, if any, from exposure to mid-frequency active sonar and other active acoustic sources evaluated in this opinion are likely to be temporary and are unlikely to have any long-term adverse impact on individual animals or affected populations. Even if sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks during major training exercises), we would expect that some individual whales would avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to 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 habitat to be available in close proximity. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

Further, and as described in section 6.2.13, establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013; Read et al. 2014a; Southall et al. 2016). To date, "we do not yet have the data to underpin the link between behavioral response and population consequences" (Harris et al. 2017). 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 (DeRuiter et al. 2017; Ellison et al. 2012a; Friedlaender et al. 2016; Goldbogen et al. 2013; Harris et al. 2017; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014b; Williams et al. 2014).

Our assessment that the continuation of the Navy activities into the reasonably foreseeable future is unlikely to have any adverse additive or long-term impacts on the affected threatened or endangered species (assuming current levels of activity and no significant changes in the status of species or to the *Environmental Baseline*) is also consistent with the absence of any documented population-level or adverse aggregate impacts resulting from DoD and USCG activities to date, despite decades of training in the MITT action area using many of the same systems. Most of the training activities the DoD and USCG conducts in the MITT action area are similar, if not identical, to activities that have been occurring in the same locations for decades.

Our regulations require us to consider, using the best available scientific data, effects of the action that are "likely" and "reasonably certain" to occur rather than effects that are speculative or uncertain. See 50 C.F.R. § 402.02 (defining to "jeopardize the continued existence of" and "effects of the action"). Our analysis and conclusions in this opinion are based on estimates of exposures and take assuming that the DoD and USCG conduct the maximum number of authorized training and testing activities for the maximum number of authorized hours. The effects of the action in relation to the *Status of Listed Resources* and the *Environmental Baseline* are presented by each species below.

6.10.1 Blue Whale

In determining whether U.S. Navy training and testing activities in the MITT 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any blue whales being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

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. The current abundance trend for blue whales rangewide including the MITT action area is not well understood. However, recent evidence indicates that some blue whale populations in the North Pacific may be increasing (Monnahan et al. 2014).

Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that in any given year during the five-year period (August 2015 through August 2020), blue whales could potentially experience up to 28 instances of take in the form of behavioral harassment and TTS resulting from non-impulsive acoustic stressors. Of those 28 instances of behavioral harassment, we anticipate no more than of 3 takes per year in the form of TTS. We do not anticipate any take in the form of injury from PTS or other injuries such as GI tract or lung injury during annual training or testing activities. We do not anticipate any mortality of blue whales from MITT stressors. The estimates of exposures to training and testing exercises that would result in behavioral responses (including instances of TTS) annually would not be expected to appreciably reduce the likelihood of the survival of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal’s hearing sensitivity from their vocalizations, we have no data on blue whale hearing so we assume that blue whale vocalizations are partially representative of their hearing sensitivities. Blue whales are not likely to respond to high-frequency sound sources associated with the proposed training exercises and testing activities because of their hearing sensitivities. Despite previous assumptions based on vocalizations and anatomy that blue whales predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b) (Oleson et al., 2007; Stafford and Moore, 2005), recent research has indicated blue whales not only hear mid-frequency active sonar transmissions, in some cases they respond to those transmissions (Southall et al. 2011a).

Blue whales may hear some sounds in the mid-frequency range and exhibit behavioral responses to sounds in this range depending on received level and context (Goldbogen et al. 2013 and Melcon et al. 2012). However, both Goldbogen et al (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational MFA sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). As stated in Goldbogen et al. (2013) when summarizing the response of blue whales to simulated MFA sonar, “We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Goldbogen et al. (2013) also speculated that if this temporary behavioral response interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. Additionally, in instances of TTS, individuals would likely fully recover within 24 hours of exposure and resume normal behavioral activities. There is no indication that this is the case, particularly since unconsumed prey would still be available in the environment following the cessation of acoustic exposure and resumption of normal behaviors following instances of behavioral response including responses associated with TTS.

During the Controlled Exposure Experiments (CEEs) used in Goldbogen et al. (2013), sound sources were deployed from a stationary source vessel positioned approximately 1,000 m from the focal animals, with one transmission onset every 25 seconds (Southall et al. 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 CEEs described above, transmitting about once per minute (DoN 2015). For example, a typical Navy vessel with hull mounted MFA sonar would travel over 0.3 kilometers between pings (based on a speed of 10 knots/hr and transmission rate of 1 ping/min). Based on this distance traveled and potential avoidance behavior of acoustically exposed animals, we expect repeat acoustic exposures capable of eliciting a behavioral response to an individual over a brief period of time to be rare. In the event an individual is exposed to multiple sound sources that elicit a behavioral response (e.g., disruption of feeding) in a short amount of time, including instances of TTS, we do not expect these exposures to have fitness consequences as individuals will resume feeding upon cessation of the sound exposure and unconsumed prey will still be available in the environment.

Even if sound exposure were to be concentrated in a relatively small geographic area over a long period of time (e.g., days or weeks during major training exercises), we would expect that some individual whales would avoid areas where exposures to acoustic stressors are at higher levels (e.g., greater than 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to 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.

The blue whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the MITT action area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including TTS, 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 6.2.11 and 6.8.3.2.1 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.

Further, recent evidence indicates that the Eastern North Pacific blue whale population, has likely reached carrying capacity (Monnahan et al. 2014). Navy training and testing activities, including the use of MFA sonar, have been ongoing in this area for decades. Therefore, any potential temporary behavioral impacts on blue whales from the use of MFA during Navy training and testing activities do not appear to have inhibited population growth of the Eastern North Pacific blue whale population in that area. We would expect similar effects to exist in the MITT action area.

The 1998 blue whale recovery plan does not outline downlisting or delisting criteria. The recovery plan does list several stressors potentially affecting the status of blue whales in the North Pacific Ocean that are relevant to MITT 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 MITT activities is not expected to impact the fitness of any individuals of this species. No mortality of blue whales is expected to occur from MITT activities.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of blue whales in the wild by reducing the reproduction, numbers, or distribution of that species. MITT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual blue whales in ways or to a degree that would reduce their fitness. We anticipate temporary behavioral responses, but do not anticipate any injury or mortality from acoustic stressors. We do not anticipate those behavioral responses to result in fitness consequences to any individuals and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.2 Fin Whale

In determining whether U.S. Navy training and testing activities in the MITT 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any fin whales being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that in any given year during the five-year period (August 2015 through August 2020), fin whales could potentially experience up to 28 instances of take in the form of behavioral harassment and TTS resulting from non-impulsive acoustic stressors. Of those 28 instances of behavioral harassment, we anticipate no more than 4 takes per year in the form of TTS resulting from non-impulsive sound stressors. We anticipate no take by injury in the form of permanent threshold shift (PTS) or other injuries such as GI tract or lung injury from annual training and testing activities. We do not anticipate any mortality of fin whales from MITT stressors. The estimates of annual exposures to training and testing exercises that would result in behavioral responses (including instances of TTS) would not be expected to appreciably reduce the likelihood of the survival of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of fin whales. However, recent observations of blue whale responses to the mid-frequency sonar sounds support the possibility that this ecologically, physiologically, and taxonomically similar species may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on fin whales. Considering information presented in this opinion, we consider fin whales to be able to hear and respond to mid-frequency sonar as blue whales appear to.

The fin whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the MITT action area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Individuals exposed to acoustic stressors at levels resulting in TTS will likely fully recover within 24 hours of the exposure and resume normal behaviors including feeding. As described previously, including in Sections 6.2.11 and 6.8.3.2.1 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.

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 MITT activities. As discussed previously, anthropogenic noise associated with MITT activities will not impact the fitness of any individuals of this species. Downlisting criteria for fin whales includes the maintenance of at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a 1 percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on fin whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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. MITT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual fin whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations).

We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.3 Humpback Whale – Western North Pacific DPS

In determining whether U.S. Navy training and testing activities in the MITT action area are likely to jeopardize the survival and recovery of humpback whales from the Western North Pacific DPS, 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any humpback whales being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that in any given year during the five year period (August 2015 through August 2020), humpback whales from the Western North Pacific DPS could experience up to 860 instances of behavioral harassment and TTS resulting from non-impulsive acoustic stressors. Of those 860 instances, we anticipate no more than 679 instances per year of harassment in the form of TTS resulting from non-impulsive sound stressors. We anticipate zero instances of injury in the form of PTS or other injuries such as GI tract or lung injury from annual training and testing activities. We do not anticipate any mortality of humpback whales from MITT stressors. The estimates of annual exposures to training exercises and exposures to testing exercises that would result in behavioral responses (including instances of TTS) would not be expected to appreciably reduce the likelihood of the survival of humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of humpback whales. However, recent observations of blue whale responses to the mid-frequency sonar sounds support the possibility that this ecologically, physiologically, and taxonomically similar species may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on humpback whales. Considering information presented in this opinion, we consider humpback whales to be able to hear and respond to mid-frequency sonar similar to blue whales.

The humpback whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the MITT action area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, 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 6.2.11 and 6.8.3.2.1 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.

There is no recovery plan specific to humpback whales from the Western North Pacific DPS. The 1991 humpback whale recovery plan (for the previous range-wide listing) 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 MITT activities. As described previously, anthropogenic noise associated with MITT activities will not impact the fitness of any individuals of this species and vessel strike of humpback whales are unlikely to occur from Navy training and testing activities.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of Western North Pacific DPS 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 DPS humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. MITT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual humpback whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.4 Sei Whale

In determining whether U.S. Navy training and testing activities in the MITT 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any sei whales being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

Assuming the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that in any given year during the five year period (August 2015 through August 2020), sei whales could experience up to 319 instances of behavioral harassment and TTS resulting from non-impulsive acoustic stressors. Of those 319 instances of behavioral harassment, we anticipate no more than 258 instances per year of harassment in the form of TTS resulting from non-impulsive sound stressors. We anticipate zero instances of injury in the form of PTS resulting from non-impulsive acoustic stressors or other injuries such as GI tract or lung injury from annual training and testing activities. We do not anticipate any mortality of sei whales from MITT stressors. The estimates of annual exposures to training exercises and exposures to testing exercises that would result in behavioral responses (including instances of TTS) would not be expected to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

Frequencies associated with mid-frequency sonar have generally been considered above the hearing range of sei whales. However, recent observations of blue whale responses to the mid-frequency sonar sounds support the possibility that this ecologically, physiologically, and taxonomically similar species may be capable of detecting and responding to them. Additional data are necessary to determine the potential impact that mid-frequency sonar may or may not have on sei whales. Considering information presented in this opinion, we consider sei whales to be able to hear and respond to mid-frequency sonar similar to blue whales.

The sei whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the MITT action area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Individuals experiencing TTS are likely to fully recover within 24 hours of exposure and return to normal behaviors. As described previously, including in Sections 6.2.11 and 6.8.3.2.1 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.

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 MITT activities. As described previously, anthropogenic noise associated with MITT activities will not impact the fitness of any individuals of this species. Downlisting criteria for fin whales includes the maintenance of 1,500 mature, reproductive individuals with at least 250 mature females and 250 mature males in each recovery population, which is already exceeded in the North Pacific. To qualify for downlisting, each recovery population must also have no more than a 1 percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a 10 percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sei whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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. MITT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual sei whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.5 Sperm Whale

In determining whether U.S. Navy training and testing activities in the MITT 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any sperm whales being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated that in any given year during the five-year period (August 2015 through August 2020), sperm whales could experience up to 506 instances of take in the form of behavioral harassment and TTS resulting from non-impulsive acoustic stressors. Of these 506 instances of behavioral harassment, we anticipate no more than 54 takes per year in the form of TTS resulting from non-impulsive sound stressors. We do not anticipate any take in the form of injury from PTS or other injuries such as GI tract or lung injury during annual training and testing activities. We do not anticipate any mortality of sperm whales from stressors associated with MITT activities. The estimates of exposures to training exercises and exposures to testing exercises that would result in behavioral responses (including instances of TTS) annually would not be expected to appreciably reduce the likelihood of the survival of sperm whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

The sperm whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the MITT action area. Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. Indivudals experiencing TTS will likely fully recover within 24 hours and resume normal behaviors. As described previously, including in Sections 6.2.11 and 6.8.3.2.5 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.

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 MITT activities. As discussed previously, anthropogenic noise associated with MITT 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.

Based on the evidence available, including the environmental baseline and cumulative effects, we conclude that impulsive and non-impulsive acoustic stressors resulting from training exercises and testing activities the DoD and USCG plans to conduct in the MITT action area on an annual basis, or cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the status of the species or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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. MITT stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual sperm whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.6 Sea Turtles

In determining whether U.S. Navy training and testing activities in the MITT 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any sea turtles being exposed to U.S. Navy vessels, sound fields associated with active sonar pings, or shock waves associated with underwater detonations. Those individuals that are exposed would only be so periodically or episodically.

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.

The predicted acoustic impacts on sea turtles from training and testing activities are shown in Table 53 and Table 58. The exposure estimates represent the total number of exposures and not necessarily the number of individuals exposed, as a single individual may be exposed multiple times over the course of a year. The predicted acoustic impacts do not take into account avoidance behavior or mitigation measures, such as establishing shut-down zones for certain sonar systems.

Approximately 12,500 explosives would be used during training and testing activities per year and would occur in areas designated for use of explosives within the action area. Approximately 10,000 (or 80 percent) of the explosives used in the action area are in source class E1 (0.1 to 0.25 lb. NEW). Other than those on FDM, most detonations would occur beyond approximately 12 nm from shore, minimizing impacts near nesting beaches or coastal habitats for sea turtles. A small number of near-shore (within 3 nm) training activities could occur, potentially exposing some sea turtles approaching nesting beaches to impulse sounds over a short duration if the training occurred during nesting season or close to sea turtles nearshore habitats. The terrain of FDM does not provide any nesting beaches, so effects are not expected at this location to nesting turtles.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Moein Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003b). Additionally, they are not known to produce sounds underwater for communication.

As described in sections 6.3.6, 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 to individual animals or long-term implications for the population.

We assume that acoustic stressors from Navy training and testing activities conducted during the five-year MMPA rule and into the reasonably foreseeable future are likely to cause TTS and PTS in ESA-listed sea turtles in the MITT action area. However, as described previously in sections 6.3.3 and 6.8.4.4, 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 MITT action area also have the ability to cause slight lung injury in ESA-listed sea turtles in the MITT action area. As described previously in section 6.8.4.4, 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 slight lung injury to substantially impact individual turtles when considering their overall lifetime fitness, we do not expect impacts to populations composed of those individual turtles.

The sea turtles that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons when the Navy is training and testing in the action area. These periodic or episodic exposure and response scenarios would allow sufficient time for the affected individuals to return to baseline conditions and resumption of normal behavioral activities. As described previously in Section 6.3.7, 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 in the sections below, along with lethal vessel strike.

6.10.6.1 Green Sea Turtle

Our analysis indicated that during annual training and testing activities in the MITT action area from August 2015 to August 2020, green sea turtles could experience take in the form of behavioral harassment, TTS, PTS, slight lung injury, or death. Based on the Navy's acoustic modeling, we anticipate approximately 1,837 instances of exposure to acoustic stressors that may result in behavioral harassment each year. Additionally, we expect 262 takes per year in the form of TTS resulting from impulsive (11 instances) and non-impulsive (251 instances) sound stressors each year. We anticipate one take each year during the five-year period in the form of PTS from impulsive sound sources. We also estimate approximately three (3) injuries in the form of slight lung injury and one (1) mortality from acoustic stressors will occur each year. We also anticipate the mortality of one green sea turtle from vessel strikes in a given year.

As stated in section 6.6.2, information was not available to estimate the abundance or density of each green sea turtle DPS in most portions of the action area. In general, in-water information on green sea turtles in the action area (which largely overlaps with the delineation of the Central West Pacific DPS of green sea turtles) is limited (80 FR 15271). Therefore, sufficient information is not available to quantitatively assign most green sea turtle take to specific DPSs (with the exception of the two estimated instances of mortality, as described below). Similarly, we do not have sufficient information to assign sea turtles estimated to be killed by DoD and USCG activities to individual nesting populations. Below, we present a qualitative assessment of the likelihood of turtles from a specific DPS being adversely affected by the proposed action in the form of sub-lethal take. As stated in Section 4.2.6.16, the vast majority of the green sea turtles that occur in the action area are likely from the Central West Pacific DPS. The majority of the action area overlaps with the nesting range of this DPS and the limited genetic testing that has occurred in the action area (in nearshore areas around CNMI) indicates that most green sea turtles are from this DPS. Therefore, the majority of green sea turtles that would be adversely affected by the Navy's activities in the form of sub-lethal take would likely be from the Central West Pacific DPS. A few green sea turtles from the East Indian-West Pacific DPS would also likely experience some form of sub-lethal take (most likely behavioral harassment) because the western portion of the action area extends into the area delineated for this DPS in the Final Rule to list 11 DPSs of green sea turtles under the ESA (81 FR 20057). Additionally, the oceanic range of this DPS may extend further east into other portions of the action area where Navy training and testing activities will occur. We would also expect a few green sea turtles from the Central North Pacific DPS to experience sub-lethal take because limited genetic sampling has indicated that approximately 3% of green sea turtles foraging in nearshore areas around CNMI are likely from this DPS (Peter Dutton, NMFS, personal communication to Eric MacMillan June 1, 2016).

Though in-water information on green sea turtles in the action area is limited (80 FR 15271), available data and subject matter expert opinion (T. Todd Jones personal communication to Eric MacMillan on May 24, 2016; Peter Dutton, NMFS, personal communication to Eric MacMillan June 1, 2016) suggest that the vast majority of green sea turtles in the MITT action area would be from the Central West Pacific DPS. Therefore, it is likely that the two instances of green sea turtle mortality (one from impulsive acoustic stressors and one from vessel strike) would be of turtles from the Central West Pacific DPS.

The potential fitness consequences of non-lethal instances of take (i.e., behavioral harassment, TTS, PTS, and slight lung injury) are described in the preceding section (Section 6.10.6). 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 from Navy activities could have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years (or ever, given high natural mortality rates of juvenile turtles).

In order to analyze the impact of these mortalities to the adult female population, we assume that 50 percent of those killed would be female nesting-age turtles (i.e., with two total turtle mortalities annually, one nesting female would be killed each year)[10]. As stated in Section 4.2.6.13, the Central West Pacific DPS of green sea turtles has an estimated 6,518 nesting females. Removing one female from this population in one year would reduce the reproductive potential of this population by 0.015 percent[11]. Because Navy training and testing activities in the MITT action area are expected to occur not just for a single year, but for the remainder of the five-year MMPA rule and into the reasonably foreseeable future, we also assess the potential impact of this level of annual mortality on the Central West Pacific DPS of green sea turtles over a longer, five-year time period[12] (1 annual nesting female mortality x 5 years = 5 nesting females). Removing five females from this population would reduce the reproductive potential of this population by 0.08 percent. We assume that if the status of the species and Navy activity levels remain the same, the estimated number of green turtles killed or injured by Navy activities annually, and in a 5-year period, will continue into the reasonably foreseeable future. We do not consider this an appreciable reduction in the numbers of female green turtles or the reproductive

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[10] Note that assuming 50 percent of the green turtles killed are nesting females is highly conservative, as explained in the following paragraph.

[11] As a worst case scenario for green sea turtles from the other DPSs which could occur in the action area, we also evaluated the potential for population level effects from removing one green turtle annually from the East Indian-West Pacific and Central North Pacific DPSs. The East Indian-West Pacific DPS has at least 61,000 nesting females. Removing one female from this population in one year would reduce the reproductive potential of this population by 0.002 percent, and removing five females from this population over a five-year time period would reduce the reproductive potential of this population by 0.008 percent. The Central North Pacific DPS has an estimated 3,846 nesting females. Removing one female from this population in one year would reduce the reproductive potential of this population by 0.03 percent, and removing five females from this population over a five-year time period would reduce the reproductive potential of this population by 0.13 percent. We do not consider this an appreciable reduction in the numbers of female green turtles or the reproductive rate of these DPSs, 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 green sea turtles from the East Indian-West Pacific and Central North Pacific DPSs, we do not expect this level of mortality to impact the survival or recovery of these DPSs.

[12] We believe that a five-year time period establishes a reasonable time frame to allow us to meaningfully evaluate the effects of mortality. If the status of the species and Navy activity levels remain the same, we anticipate this five-year mortality rate to continue into successive five year periods into the reasonably foreseeable future, and have considered this in our jeopardy analysis. We also note that because MMPA take authorization is limited to five years, any proposed new MMPA take authorization for Navy activities in the MITT action area would require consultation under the ESA prior to the expiration of the current take authorization in August 2020, and not less than every five years thereafter. These periodic reassessments will allow us to ensure that our assessment of mortality risk from the Navy's ongoing activities and the resultant impacts on listed sea turtle species is up-to-date and that no impacts are omitted from consideration.

rate of this DPS, 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 endangered Central West Pacific DPS of green sea turtles, we do not expect this level of mortality to impact the survival or recovery of this population.

Additionally, assuming that 50 percent of the green turtles killed are nesting-age females is a highly conservative assumption because most in-water green turtles in the Marianas Archipelago are juveniles (Kolinski et al. 2006; Palacios 2012; Pultz et al. 1999; Summers et al. 2017). If juveniles were to be killed, the potential for such mortalities to impact the population would be much less. For example, Van Houtan (2015) conducted a modeling exercise to calculate impacts to a sea turtle population from a fishery that incidentally captures primarily juvenile turtles in the Pacific Ocean. Since most of the sea turtles bycaught in the fishery were juveniles, the author estimated mortality from the fishery to adult nester equivalents, based the number and size of juvenile turtles killed in the fishery, demographic data, mortality rates, population sex ratios, and size-to-age models. For green turtles, Van Houtan (2015) estimated that by incidentally capturing and killing 18 juvenile green turtles annually, the fishery killed an adult to nester equivalent of 0.10 adult female green turtles annually, which is analogous to the fishery incurring a nester mortality every 10.1 years. This indicates that if the turtles killed by Navy activities (either explosives or vessel strike) were juveniles (which is the more likely scenario based on sea turtle survey data from the Mariana Archipelago), potential impacts to the reproductive potential of the population would be even lower than described in the previous paragraph.

Also important in this discussion is information on the status and trends of green turtle populations in the MITT action area. While, there is insufficient long-term and standardized monitoring information to describe abundance and population trends adequately for most areas of the MITT action area (e.g., Maison et al (2010)), trend information is available for sea turtles around Guam. Martin et al. (2016) analyzed data from five decades of marine megafauna surveys around Guam and found that since the 1960s, sea turtle abundance increased by 7%. This increase is despite the Navy conducting training and testing activities around Guam and in the MITT action area for decades. Martin et al. (2016) suggested that protections in the region may be working to recover turtle populations and noted that the observed increase in sea turtles in Guam is consistent with the historical shift from extraction to conservation protection.

Based on our analysis in this opinion, we conclude that effects from ongoing DoD and USCG 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 recovery of the Central West Pacific, East Indian-West Pacific, or Central North Pacific DPSs of green sea turtles in the wild by reducing the reproduction or distribution of those species. While a small number of individual Central West Pacific DPS green sea turtles are expected to die, we do not expect this level of mortality to result in population or species level effects that could impact the survival or recovery of this species. Green sea turtles from the Central West Pacific, East Indian-West Pacific, and Central North Pacific DPSs are expected to experience sub-lethal adverse affects, but these impacts are not expected to affect the fitness of individual turtles when considering their overall lifetime fitness. Because we do not expect consequences to the fitness of individual animals, we do not expect sub-lethal adverse impacts to affect the survival or recovery of any of these DPSs.

6.10.6.2 Hawksbill sea turtle

Our analysis indicated that during annual training and testing activities in the MITT action area from August 2015 to August 2020, hawksbill sea turtles could experience take in the form of behavioral harassment resulting from non-impulsive and impulsive acoustic stressors. As described in sections 6.3.6, 6.8.4.4, and 6.8.8.5, 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.

Based on the Navy’s acoustic modeling, we anticipate approximately 129 instances of exposure to impulsive acoustic stressors that may result in behavioral harassment. We do not anticipate any additional exposures to non-impulsive acoustic stressors that might result in purely behavioral responses that were not counted as TTS, PTS, or injury. Additionally, we anticipate no more than 20 takes per year in the form of harassment from TTS resulting from impulsive (3 instances) and non-impulsive (17 instances) sound stressors. We anticipate zero takes each year during the five-year period in the form of PTS from impulsive sound sources. We estimate that approximately one slight lung injury and one mortality from acoustic stressors could occur in a given year. We do not anticipate any mortality of this species from vessel strike.

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 from Navy activities will have immediate effects on recruitment while lost reproductive potential from mortality of a juvenile female might not be realized for several years (or ever, given high natural mortality rates of juvenile turtles).

As stated previously, the most recent estimate of the total annual number of nesting females for the Central Pacific hawksbill population was 950 to 1,185 females annually (NMFS and USFWS 2013). Hawksbill turtles do not nest annually, with remigration periods varying from one nesting site to another, averaging between every two to seven years (NMFS and USFWS 2013). To conservatively estimate the number of nesting females in the Central Pacific hawksbill population, we assume a remigration period of two years, and estimate that there are approximately 1,900 to 2,370 nesting females in the Central Pacific population. If Central Pacific hawksbills have longer remigration periods, the population estimate would be higher. Conservatively assuming there are 1,900 nesting females in the population, removing one female from this population in a given year would reduce the reproductive potential of this population by 0.05 percent. Because Navy training and testing activities in the MITT action area are expected to occur not just for a single year, but for the remainder of the five-year MMPA rule and into the reasonably foreseeable future, we also assess the potential impact of this level of annual mortality on the hawksbill turtle populations over a longer, five-year time period [13]. Over a five year period, we estimate five hawksbill turtles are likely to die from Navy acoustic stressors (1 annual hawksbill mortality x 5 years = 5 hawksbill mortalities). Assuming that 50 percent of hawksbill turtles are nesting females[14], we estimate that up to three female hawksbill turtles will die over a five year period. Removing three females from this population would reduce the reproductive potential of this population by 0.16 percent. We assume that if the status of the species and Navy activity levels remain the same, the estimated number of hawksbill turtles killed or injured by Navy activities annually, and in a 5-year period, will continue into the reasonably foreseeable future. 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.

We also consider how losing one female could impact the reproductive potential of Pacific hawksbill sea turtles ocean basin-wide or the rangewide population (i.e., the listed entity). The

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[13] We believe that a five-year time period establishes a reasonable time frame to allow us to meaningfully evaluate the effects of mortality. If the status of the species and Navy activity levels remain the same, we anticipate this five-year mortality rate to continue into successive five year periods into the reasonably foreseeable future, and have considered this in our jeopardy analysis. We also note that because MMPA take authorization is limited to five years, any proposed new MMPA take authorization for Navy activities in the MITT Action Area would require consultation under the ESA prior to the expiration of the current take authorization in August 2020, and not less than every five years thereafter. These periodic reassessments will allow us to ensure that our assessment of mortality risk from the Navy's ongoing activities and the resultant impacts on listed sea turtle species is up-to-date and that no impacts are omitted from consideration.

[14] Assuming that 50 percent of affected turtles are nesting females is highly conservative because Summers et al. (2017) found that the majority of hawksbill turtles in CNMI are juveniles. If juveniles were to be killed, the potential for such mortalities to impact the population would be much less (e.g., Van Houtan 2015).

most recent abundance estimate for nesting female hawksbill sea turtles in the Pacific Ocean is 10,194 to 12,770 nesting females per year and range wide is 22,004 to 29,035 nesting females per year (NMFS and USFWS 2013). Conservatively assuming a remigration period of two years, there are approximately 20,388 to 25,540 nesting females in the Pacific Ocean and 44,008 to 58,070 nesting females range wide. Removing one female from the Pacific Ocean and rangewide population in a given year would reduce the reproductive potential of these populations by 0.005 and 0.002 percent, respectively. Removing three females from the Pacific Ocean and rangewide population over a five-year period would reduce the reproductive potential of these populations by 0.014 and 0.007 percent, respectively. We do not consider this to be an appreciable reduction in the numbers of female hawksbill turtles or the reproductive rate of the Central West Pacific, Pacific Ocean (basin-wide), or rangewide population, either on an annual basis or continuing into the reasonably foreseeable future. We do not expect this level of mortality to result in an appreciable reduction in the numbers or reproductive rate of hawksbill sea turtles in the Central Pacific Ocean or rangewide. For this reason, we also do not expect this level of mortality to impact the survival or recovery of Central Pacific Ocean hawksbill sea turtles or the rangewide population.

As described in NMFS and the U.S. Fish and Wildlife Service's most recent (2013) five-year status review for hawksbill sea turtles, within the past 20 years, out of the 24 nesting sites evaluated in the Pacific Ocean, abundance was increasing at zero sites, stable at three sites, and decreasing at 28 sites (NMFS and USFWS 2013). Specific to the Marianas, the status review found that hawksbill populations were likely decreasing. On the other hand, Martin et al. (2016) analyzed data from five decades of marine megafauna surveys around Guam and found that since the 1960s, in-water sea turtle abundance increased by 7%. This increase is despite the Navy conducting training and testing activities around Guam and in the MITT action area for decades. Martin et al. (2016) suggested that protections in the region may be working to recover turtle populations and noted that the observed increase in sea turtles in Guam is consistent with the historical shift from extraction to conservation protection. Considering this information, and viewed within the context of the Status of Listed Resources and the Environmental Baseline (most notably including the numerous other anthropogenic stressors which result in impacts to hawksbill turtles in the Pacific Ocean including habitat loss, incidental capture in fisheries, and directed harvest), we believe the annual mortality of one hawksbill sea turtle caused by Navy training and testing activities in the MITT action area will not adversely affect the population viability of hawksbill sea turtles in the Central West Pacific, the Pacific Ocean as a whole, or rangewide.

Based on our analysis in this opinion, we conclude that effects from ongoing DoD and USCG 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 hawksbill sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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 hawksbill sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. That is, acoustic stressors will not have fitness consequences or 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 the species as listed pursuant to the ESA.

6.10.6.3 Leatherback Sea Turtle

Our analysis indicated that during annual training and testing activities in the MITT action area from August 2015 to August 2020, leatherback sea turtles could experience take in the form of behavioral harassment resulting from non-impulsive and impulsive acoustic stressors. As described in sections 6.3.6, 6.8.4.4, and 6.8.8.5, 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.

Based on the Navy’s acoustic modeling, we anticipate approximately 49 instances of exposure to impulsive acoustic stressors that may result in behavioral harassment. We do not anticipate any additional exposures to non-impulsive acoustic stressors that might result in purely behavioral responses that were not counted as TTS, PTS, or injury. Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated 12 takes per year in the form of harassment from TTS resulting from non-impulsive sound stressors. We anticipate zero takes each year during the five-year period in the form of PTS from impulsive sound sources or other forms of injury including GI tract, slight lung injury or mortality from acoustic stressors.

The 1998 recovery plan for the U.S. Pacific populations of leatherback sea turtles does not identify any major threats occurring in the action area, but sets criteria for the delisting the species. Delisting requires identifying regional stocks to source beaches, stability in the number of nesting females over 25 years, maintenance of at least 5,000 females in each stock over 6 years, maintenance of healthy foraging habitat, increases of foraging populations, completion of all priority one tasks, and the finalization of management plans. Any leatherbacks affected by the proposed action are not expected to experience fitness consequences because this species does not rely heavily on auditory cues from their environment for breeding, feeding, or sheltering. Because of a lack of fitness consequences and that acoustic stressors are not identified as a threat to leatherback recovery, we do not expect sub-lethal leatherback sea turtle take to impede recovery of this species.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of leatherback sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of leatherback sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. Therefore, we do not anticipate any measurable or detectable reductions in survival rate or trajectory of recovery of sub-populations in the North Pacific or to these species as listed pursuant to the ESA.

6.10.6.4 Loggerhead Sea Turtle – North Pacific DPS

Our analysis indicated that during annual training and testing activities in the MITT action area from August 2015 to August 2020, loggerhead sea turtles from the North Pacific DPS could experience take in the form of behavioral harassment resulting from non-impulsive and impulsive acoustic stressors. As described in sections 6.3.6, 6.8.4.4, and 6.8.8.5, 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.

Based on the Navy's acoustic modeling, we anticipate approximately 54 instances of exposure to impulsive acoustic stressors that may result in behavioral harassment. We do not anticipate any additional exposures to non-impulsive acoustic stressors that might result in behavioral responses that were not counted as TTS, PTS, or injury. Assuming that the Navy conducts the maximum number of authorized training and testing activities for the maximum number of authorized hours, we estimated 15 takes per year in the form of harassment from TTS resulting from non-impulsive sound stressors. We anticipate zero takes each year during the five-year period in the form of PTS from impulsive sound sources or other forms of injury including GI tract, slight lung injury or mortality from acoustic stressors.

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. 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 the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival or the recovery of loggerhead sea turtles from the North Pacific Ocean DPS in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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 loggerhead sea turtles in the wild by reducing the reproduction, numbers, or distribution of that species. That is, non-impulsive acoustic stressors will not have fitness consequences at the individual 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.

6.10.7 Scalloped Hammerhead Shark – Indo-West Pacific DPS

In determining whether U.S. Navy training and testing activities in the MITT action area are likely to jeopardize the survival and recovery of the Indo-West Pacific DPS of scalloped hammerhead sharks, 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. Many U.S. Navy training exercises and testing activities in the MITT action area over a five-year period and continuing into the reasonably foreseeable future will occur without any scalloped hammerhead sharks being exposed to shock waves associated with underwater detonations. Those individuals that are exposed to UNDETs in and around Apra Harbor, Guam would only be so periodically or episodically over certain months or seasons.

In section 6.8, we analyzed the effects of acoustic stressors on scalloped hammerhead sharks. There is no evidence of juvenile or adult scalloped hammerhead shark injury or mortality resulting from underwater detonations or other impulsive acoustic stressors during Navy training and testing. Additionally, the Navy has mitigation measures in place to reduce the likelihood of this occurring. Therefore do not anticipate injury or mortality of scalloped hammerhead sharks will occur. We estimated the affected habitat area for a 20 lb. NEW charge to be 99 m^2. In this scenario, any scalloped hammerhead sharks within the 99 m^2 area could be behaviorally-harassed, or up to 99 m^2 of their habitat would be temporarily displaced during each detonation. However, sharks would be expected to return to this habitat soon after each (up to five detonations per day or 20 per week) of the 260 disturbances events and behavioral responses have concluded. We concluded that 260 instances of habitat displacement (each displacing 99 m^2) would not significantly reduce the amount (13,510,249 km^2) of habitat available to the Indo-West Pacific DPS and that scalloped hammerhead sharks would quickly recover from instances of behavioral harassment and TTS within these areas. Therefore, these temporary disruptions of available habitat would not have fitness consequences to individual scalloped hammerhead sharks since they are highly mobile and could find suitable habitat in near proximity to the habitat they were displaced from. We also determined that any instances of temporary hearing loss would not be expected to have fitness consequences to individuals because scalloped hammerheads are long-lived animals and hearing would be expected to return within a short amount of time (hours). Given the nature of testing and training as described above, these periodic or episodic exposure and behavioral response scenarios, including responses to TTS, most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously, including in Section 6.4.7 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.

The Indo-West Pacific DPS of scalloped hammerhead sharks does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded temporary displacement of individuals belonging to this listed entity could occur and that sublethal effects not rising to the level of injury from acoustic stressors would be temporary and not impact the fitness of individuals or the population.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the DoD and USCG will conduct in the MITT action area on an annual basis, cumulatively over the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of the Indo-West Pacific DPS of scalloped hammerhead sharks in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing DoD and USCG 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 the Indo-West Pacific DPS of scalloped hammerhead sharks in the wild by reducing the reproduction, numbers, or distribution of that species. We do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that would be sufficient to be readily perceived or estimated.

6.10.8 Acropora globiceps

In determining whether U.S. Navy training and testing activities in the MITT action area are likely to jeopardize the survival and recovery of *Acropora globiceps*, 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. Our effects analysis determined that colonies of *Acropora globiceps* around FDM would likely be impacted from in-water explosions and direct strike from live and inert ordance. Though we are unable to provide a quantitative estimate of the number of *Acropora globiceps* colonies impacted by Navy activities at FDM (see section 6.8.10.2 for additional detail), we determined that 9,064 square meters of nearshore habitat around FDM could be impacted annually. The large majority of impacts are estimated to result from high explosive bombs that miss their intended on-shore target. We consider the estimated area of impact calculated above to be highly conservative because the crater our estimates are based on is the largest that has been observed in a decade of dive surveys. For example, Smith et al. (2013b) indicated that the size of any disturbed areas were generally less than two square meters.

While a small area of habitat affected by Navy activities each year is likely to contain colonies of *Acropora globiceps* that could be injured or killed, the majority of the area impacted will not. Further, although individual colonies and clusters of colonies forming a small-scale reefscape are likely to be negatively impacted by impulsive explosions, underwater surveys of FDM reefs suggest significant population level impacts are not likely to occur, and colony repair or successful recruitment will likely occur within two to three years following disturbance (Smith and Marx Jr. 2016). Smith and Marx Jr. (2016) documented that while impacts to reef habitat have occurred around FDM (i.e., from ordnance that skipped off the island, from ordnance fragments, and from an in-water detonation), no significant impacts to the physical or biological environment were detected between 2005 and 2012. Instead, the authors suggested that restricted access to FDM because it is a bombing range has resulted in a de-facto preserve effect. They noted that marine natural resources at FDM are "comparable or superior to" those at other locations within the Mariana Archipelago. This is despite FDM being used as a target site for live-fire military exercises (ship-to-shore gunfire, aerial gunnery and bombing) since 1971 (Smith et al. 2013a).

Additionally, the area of nearshore habitat that is expected to be affected by explosives and military expended material at FDM is infinitesimally small in relation to available habitat within this species' range. *Acropora globiceps*, and other ESA-listed corals in the Indo-Pacific, consist of at least millions of colonies, and occur across a range of thousands of miles. Because the species is sparsely populated across a wide range, localized impacts to potential coral reef habitat for this species are not expected to impact the species' ability to reproduce. Instead, other factors that affect corals over a broad geographic scale are larger drivers of the ability of *A. globiceps* to survive and recover. These factors include ocean warming, disease, ocean acidification, trophic effects of fishing, nutrients, and predation. Therefore, we do not believe annual impacts to this area of habitat at FDM will result in an appreciable reduction in the likelihood of the survival or recovery of *Acropora globiceps* in the wild.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from training and testing activities the Navy will conduct in the MITT action area on an annual basis, cumulatively over the reminader of the five year period from August 2015 through August 2020, or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected to appreciably reduce the likelihood of the survival of *Acropora globiceps* in the wild by appreciably reducing the reproduction, numbers, or distribution of that species. While a small number of individual colonies would be killed around FDM, this number is likely replaced with new colonies each year. Instances of recoverable injury such as breakage resulting from these stressors may also serve to propagate coral colonies in and around areas where mortality is estimated. We also conclude that effects from ongoing Navy training and testing activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of *Acropora globiceps* in the wild by reducing the reproduction, numbers, or distribution of that coral species.

7 CONCLUSION

During the consultation, we reviewed the current status of endangered blue whales, fin whales, Western North Pacific DPS humpback whales, sei whales, and sperm whales, green sea turtles (endangered Central West Pacific DPS, threatened East Indian-West Pacific and Central North

Pacific DPSs), endangered hawksbill sea turtles, endangered leatherback sea turtles, endangered North Pacific Ocean DPS loggerhead sea turtles, endangered olive ridley sea turtles, the threatened Indo-West Pacific DPS of scalloped hammerhead sharks, and threatened *Acropora globiceps*. We also assessed the *Environmental Baseline* for the MITT action area including ongoing DoD and USCG training and testing in the MITT action area along with the potential effects of DoD and USCG Mariana Islands Training and Testing Study from August 2015 through August 2020 along with the National Marine Fisheries Service's Permit Division's rule on the take of marine mammals incidental to training and testing activities and letter of authorization for the five-year period.

We conclude that Navy training and testing activities in the MITT action area and NMFS' MMPA regulations and LOA are likely to adversely affect but will not appreciably reduce the ability of these threatened and endangered species under NMFS' jurisdiction to survive and recover in the wild. Therefore, we conclude that these activities are not likely to jeopardize the continued existence of any endangered or threatened species. The actions also will not result in the destruction or adverse modification of critical habitat during the five-year period of the MMPA rule or continuing into the reasonably foreseeable future. These conclusions will remain valid assuming that the type, amount and extent of training and testing do not exceed levels assessed in this opinion and/or the status of the species affected by these actions does not change significantly from that assessed in this opinion.

8 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species without special exemption. ESA § 9 statutory prohibitions are limited to "endangered" species unless extended to "threatened" species. In the case of threatened species, section 4(d) of the ESA leaves it to the Secretary's discretion whether and to what extent to extend the statutory 9(a) ''take'' prohibitions, and directs the agency to issue regulations it considers necessary and advisable for the conservation of the species. At the time of this consultation, take prohibitions have not been extended to the threatened Indo-Pacific DPS of scalloped hammerhead shark or the threatened species of Indo-Pacific corals, including *Acropora globiceps*. However, consistent with *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012), we assessed the amount or extent of take to these threatened species that is anticipated incidental to Navy training and testing activities and include this information in the ITS. Inclusion of these species in the incidental take statement serves to assist the Action Agency with monitoring of take and provides a trigger for reinitiation if levels of estimated take are exceeded.

The ESA defines "take" as "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct." 16 U.S.C. § 1532(19). Harm is further defined by regulation to include "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering.” 50 C.F.R. 222.102. NMFS has not yet defined “harass” under the ESA in regulation. On December 21, 2016, NMFS issued interim guidance on the term “harass,” defining it as an action that “creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.” Under the MMPA, Level B harassment for military readiness activities, such as the activities analyzed in this opinion, is defined as “any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered.” 16 U.S.C. § 1362(18)(B)(ii). For purposes of this consultation, we relied on NMFS’ interim definition of harassment to evaluate whether the proposed activities are likely to harass ESA-listed fish, sea turtle, and coral species. For marine mammals, we relied on the MMPA definition of Level B harassment in the context of military readiness activities to estimate the number of instances of harassment because these estimates relied on the outputs of NAEMO modeling. For further explanation, see section 6 of the opinion.

Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

8.1 Amount or Extent of Take

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 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 could be incidentally taken during the course of an action (51 FR 19953). The amount of take resulting from the Navy’s activities was estimated based on the best information available.

In the following sections we summarize the anticipated take from annual training and testing activities by species and the interrelated and 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.

Section 7(b)(4)(C) of the ESA provides that if an endangered or threatened marine mammal is involved, the taking must first be authorized by Section 101(a)(5) of the MMPA. Accordingly, the terms of this incidental take statement and the exemption from Section 9 of the ESA become effective only upon the issuance of MMPA authorization (i.e., five year regulations and LOA) to take the marine mammals identified here. Absent such authorization, this statement is inoperative for marine mammals.

Table 61 provides the anticipated take incidental to training and testing activities of ESA-listed whale and sea turtle species in a given year where all possible activities are carried out in that year.

Table 61. Take Authorized Incidental to Training and Testing Activities, Issuance of the MMPA Regulation and Issuance of the LOA

ESA-Listed Species	**Impulsive and Non-Impulsive Acoustic Stressors**				**Vessel Strike**
	Behavioral Harassment (TTS)	**Harm (PTS)**	**Harm (Slight Lung Injury)**	**Mortality**	**Mortality**
Cetaceans					
Blue Whale	**28 (25)/yr**	0	0	0	-
Fin Whale	**28 (24)/yr**	0	0	0	-
Humpback Whale – Western North Pacific DPS	**860 (679)/yr**	0	0	0	-
Sei Whale	**319 (258)/yr**	0	0	0	-
Sperm Whale	**506 (54)/yr**	0	0	0	-
Sea Turtles					
Green Turtle – Central West Pacific, East Indian-West Pacific, and Central North Pacific DPSs*	**2,099 (262)/yr**	**1/yr**	**3/yr**	**1/yr**	**1/yr**
Hawksbill Turtle	**149 (20)/yr**	0	**1/yr**	**1/yr**	**-**
Leatherback Turtle	**61 (12)/yr**	0	0	0	-
Loggerhead Turtle – North Pacific DPS	**69 (15)/yr**	0	0	0	-

*The two instances of mortality (one from impulsive acoustic stressors and one from vessel strike) are assigned to the Central West Pacific DPS. Available information does not allow us to quantitatively assign sub-lethal take estimates to specific DPSs, though the vast majority are expected to occur to green sea turtles from the Central West Pacific DPS. See section 6.10.6.1 for further detail and analysis.

- Either exposures to the stressor are not expected or responses to exposures rising to the level of “take” are not expected.

Indo-West Pacific DPS scalloped hammerhead shark – It is not practical or possible to express the amount or extent of anticipated take of scalloped hammerhead shark (Indo-West Pacific DPS) or to monitor take-related impacts in terms of individuals of these species. Therefore, the incidental take (limited to harassment) of scalloped hammerhead sharks is expressed as a habitat area surrogate (i.e., area of affected habitat) as prescribed by 50 CFR 402.14(i). We estimate temporary effects to 99 m^2 of available habitat no more than 260 times per year (no more than five times per day or 20 times per week at all sites). Reinitation will be required if (1) effects to scalloped hammerhead sharks attributable to UNDET activities exceed temporary behavioral effects or temporary habitat displacement, (2) effects to scalloped hammerhead sharks attributable to UNDET activities occur beyond 99m^2 from the detonation location, or (3) more than 260 explosive detonations at Apra Harbor, Agat Bay, and Piti Point occur in a year (not to exceed five detonations in a day and not to exceed 20 detonations in a week).

Acropora globiceps – It is not practical or possible to express the amount or extent of anticipated take of *Acropora globiceps*, or to monitor take-related impacts in terms of individuals of these species. Therefore, the incidental take of *Acropora globiceps* (in the form of harm) is expressed as a habitat area surrogate as prescribed by 50 CFR 402.14(i). We estimated that the total area of nearshore habitat around FDM impacted annually is 9,064 square meters. Reinitiation will be required if the Navy: (1) exceeds the percent miss rates assumed in section 6.8.10.2 or (2) exceeds the number of items expended by type as presented in section 6.8.10.2. In either scenario, the estimated area of nearshore habitat impacted could be exceeded and reinitiation would be required.

8.2 Effects of the Take

In this opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

8.3 Reasonable and Prudent Measures

The measures described below are nondiscretionary, and must be undertaken by the U.S Navy so that they become binding conditions for the exemption in section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures, and terms and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

1. The Navy and NMFS' Permits Division shall have measures in place to limit the potential for interactions with ESA-listed species that may rise to the level of take as a result of the proposed actions described in this opinion. Standards and procedures should be incorporated into policy and guidance, directives, and standard operating procedures as appropriate.

2. The Navy and NMFS' Permits Division shall report all observed interactions resulting in take with any ESA-listed species (marine mammals, fish, adult corals (reefs, etc.) and sea turtles) resulting from the proposed actions that are observed during the course of Navy training and testing activities and while implementing monitoring requirements for marine mammals as required by the LOA.

3. The Navy shall monitor effects to coral reef habitat at FDM.

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 non-impulsive or impulsive acoustic activities 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 of marine mammals and sea turtles at the scale of MITT 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 (for marine mammals and sea turtles), and the link between sonar or explosive use and the level of take, to determine when anticipated take levels have been exceeded.

The take levels specified above also include one annual mortality each for green (Central West Pacific DPS) and hawksbill sea turtles from acoustic sources (based on Navy modeling) and one annual green sea turtle (Central West Pacific DPS) mortality resulting from a vessel strike. 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 most likely go undetected. Therefore, for these forms of take as well, we must 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.

The amount of anticipated take of the Indo-West Pacific DPS of scalloped hammerhead sharks and *Acropora globiceps* from acoustic stressors and military expended materials (*Acropora globiceps*) are expressed as habitat surrogates, consistent with 50 C.F.R. § 402.14(i).

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.

Reinitiation of consultation shall also be required if Navy monitoring programs detect any unanticipated form of take of ESA-listed species not specified above.

8.3.2 Reporting

The Navy and NMFS’ Permits and Conservation Division shall submit reports that identify the general location, timing, number of sonar hours and other aspects of the training exercises and testing activities, and any potential to exceed levels of training and testing analyzed in this opinion they conduct in the MITT 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.

The Navy shall also submit annual reports on the levels and types of ordnance used at FDM that impacts coral reef habitat, and shall provide summary reports for required coral surveys to the ESA Interagency Cooperation Division upon completion but not less than once every five years.

8.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the U.S. Navy must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above and outlines the mitigation, monitoring and reporting measures required by the section 7 regulations (50 CFR 402.14(i)). These terms and conditions are non-discretionary. If the U.S. Navy fails to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

1. To satisfy the reasonable and prudent measures above, the Navy must report to NMFS any exceedance of activity levels (e.g. sonar hours and the type and numbers of explosives used) of planned testing or training events specified in the preceding opinion and in the final MMPA rule before the exceedance occurs (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.

2. To satisfy reasonable and prudent measure number one in Section 8.3, the Navy shall implement all mitigation and monitoring measures as proposed in the action described in the final EIS/OEIS and consultation initiation package, as specified in the final MMPA rule and LOAs, and as described in this opinion in Section 2.9.

3. To satisfy reasonable and prudent measure number two in Section 8.3, the Navy shall notify NMFS if a dead or seriously injured sea turtle is observed during or following testing and training activities. The Navy shall notify NMFS when enough data are available to determine if the dead or seriously injured sea turtle may be 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.

4. To satisfy reasonable and prudent measure number three in Section 8.3, the Navy shall monitor and report annual numbers of ordnance by type (e.g., explosive bomb, non-explosve bomb, projectiles, missiles, rockets, etc.) expended at FDM. Navy will report all observed ricochets and misses that land in waters surrounding FDM occupied by corals. Additionally, the Navy shall provide reports of any observed in-water effects (e.g., crater size, observed mortality) to corals resulting from detonations of high-explosive ordnance as they are discovered incidental to routine operations or during coral reef surveys to confirm or to help revise assumptions on the effects of high-explosive bombs and other ordnance to corals at various depths.

5. To satisfy reasonable and prudent measure number three in Section 8.3, the Navy shall, no less than once every five years, survey coral reef habitat around FDM within 30 meters of water depth. These surveys shall be structured to confirm presence or absence and abundance of ESA-listed corals and to assess general trends in coral reef species composition, percent coral coverage, and condition (disease, predators, extent of breakage, etc.).

6. To satisfy reasonable and prudent measures number one and two in Section 8.3, NMFS' Permits Division shall ensure that all mitigation and monitoring measures as proposed by the Navy and as proposed by NMFS in the final MMPA rule and in Section 2.9 of this opinion are implemented by the U.S. Navy through the issuance of a final rule and subsequent letters of authorization (LOA) pursuant to the MMPA.

7. To satisfy reasonable and prudent measures number one and two in Section 8.3, the Navy as the Executive Agent for DoD and USCG activities, shall coordinate the execution of these required actions with other affected military services and the USCG and shall compile and summarize any and all annual monitoring and exercise reports and describe observed interactions with ESA-listed species [marine mammals, fish, adult corals (reefs, etc.) and sea turtles] that involve their actions.

9 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 CFR 402.02).

1. Monitor sighting, location, and stranding data for ESA-listed marine mammals, sea turtle species, scalloped hammerhead shark, and adult corals in the MITT action area.

2. As practicable, develop procedures to aid any individuals of an ESA-listed marine mammals, sea turtle species, scalloped hammerhead shark, and adult corals that have been impacted by DoD and USCG training and testing activities and is in a condition requiring assistance to increase likelihood of survival.

3. Continue to model potential impacts to ESA-listed species using NAEMO and other relevant models; validate assumptions used in risk analyses; and seek new information and higher quality data for use in such efforts.

4. Continue technical assistance/adaptive management efforts with NMFS to help inform future consultations on DoD and USCG training and testing in the MITT action area.

5. The Navy should coordinate with NMFS to monitor for presence of ESA-listed corals in and around the Piti Point Mine Neutralization site and Underwater Detonation Sites

within Apra Harbor, Guam to ensure the absence of these species and to avoid interactions.

6. The Navy should coordinate with NMFS to understand scalloped hammerhead shark abundance and density estimates for the MITT action area.
7. The Navy should explore methods to better quantify the risk of vessel strike to sea turtles.

In order for NMFS' Office of Protected Resources Endangered Species Act Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, the U.S. Navy should notify the Endangered Species Act Interagency Cooperation Division of any conservation recommendations they implement in their final action.

10 REINITIATION OF CONSULTATION

This concludes formal consultation on proposed Mariana Islands Training and Testing activities the DoD and USCG will conduct from August 2015 through August 2020. As 50 CFR 402.16 states, 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 ESA-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 ESA-listed species or critical habitat that was not considered in this opinion, or (4) a new species is ESA-listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, the U.S. Navy and NMFS' Permits Division must contact the ESA Interagency Cooperation Division, Office of Protected Resources immediately.

11 REFERENCES

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